# Interactions between climate change, fire regimes and biodiversity in Australia: A preliminary assessment

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Australian Government
Department of Climate Change
Department of the Environment, Water, Heritage and the Arts

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**Front Cover Image Credit**
Night fire in Spinifex (*Triodia* species) lit by traditional owners in arid, central Australian dune fields.
Source: Boyd Wright, University of New England

Severely burnt subalpine snowgum (*Eucalyptus pauciflora*) woodland with heathy understorey, on Spion Kopje ridge, Bogong High Plains region, Alpine National Park, Victoria, 2003.
Source: Henrik Wahren, La Trobe University.

Burnt tropical savanna woodland, showing post-fire regeneration of cycads (*Cycas* sp), Litchfield National Park, NT.
Source: Garry Cook

**Back Cover Image Credit**
Night fire in Spinifex (*Triodia* species) lit by traditional owners in arid, central Australian dune fields.
Source: Boyd Wright, University of New England

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## PREFACE BY THE AUSTRALIAN GOVERNMENT

Fire is a characteristic feature of many Australian landscapes: the flammable vegetation and prevalence of hot, dry seasons predispose these landscapes to burn whenever there is a source of ignition. And it has been this way for a very long time, so many Australian plants and animals have evolved to cope. However, as the climate becomes hotter and drier, and as elevated $CO_2$ levels change the rates of fuel accumulation, new and different fire regimes will come into play across these landscapes. How will the regional components of climate change affect fire regimes across Australia? And what are the implications for biodiversity conservation?

This report builds on knowledge of current fire regimes, forecasts changes to fire regimes from climate change, and disentangles the interactions between climate change, fire regimes and biodiversity across Australia's diverse landscapes. Understanding these complex interactions and developing tools to deal with uncertainties in a changing climate will be imperative for future fire management for biodiversity conservation.

These findings provide the foundation for a second phase of the project that is underway. The second phase will focus on the implications of climate change at regional scales, provide practical recommendations for regional fire managers, and build capacity to inform fire management strategies to aid in the conservation of Australia's biodiversity. The efficacy of fire management tools, especially prescribed burning practices to protect landscape values of people, property and biodiversity, will be evaluated using a cost-benefit analysis approach.

The Natural Resource Management Ministerial Council (NRMMC), in their Climate Change Adaptation Work Program 2006-2009, identified building of knowledge and capacity for fire managers as one of eleven priority actions to assist Australian natural resource management to adapt to the inevitable impacts of climate change. A number of other Australian Government and intergovernmental initiatives underway are building understanding of risks posed to biodiversity and to natural areas because of climate change, including to world heritage areas, protected areas, ecological water flows and biodiversity more generally. These initiatives will ensure that climate change becomes well integrated into policy decision-making relevant to biodiversity conservation and management in Australia.

# EXECUTIVE SUMMARY

**Summary of Key Findings:**

- Climate change will affect fire regimes in Australia through the effects of changes to temperature, rainfall, humidity, wind – the fire weather components - and through the effects of increases in atmospheric $CO_2$, and changes in moisture, on vegetation, and therefore fuels.
- Examination of weather data from south-eastern Australia over the period 1973-2007 shows that fire danger (as measured by the annual sum of the commonly-used Forest Fire Danger Index) rose by 10-40% at many sites from 2001-2007 relative to 1980-2000. Increases in fire danger have also been detected in some other parts of Australia.
- Climate change projections are for warming and drying over much of Australia, and hence an increased risk of severe fire weather, especially in south-eastern Australia. Modeling suggests an increase of 5 to 65 per cent in the incidence of extreme fire danger days by 2020 in this region.
- Climate change will have complex effects on fuels. On one hand, elevated $CO_2$ may enhance vegetation production and thereby increase fuel loads. On the other hand, drought may decrease long-term vegetation production (thereby decreasing fuel loads) and may decrease fuel moisture (thereby increasing potential rates of spread). The outcome of these processes on fuels, and hence fire regimes, are highly uncertain, and require further research.
- Fire regimes within Australia differ because of variation in key drivers such as fuel accumulation and drying, fire weather and ignitions. Climate change may be expected to affect fire regimes more in regions where the constraining factor(s) are fire weather-related (e.g. temperate forests of the south-east), than in places where the fire regimes are determined more by fuel or ignition rather than fire weather (e.g. tropical savannas of the north)
- Future fire regimes will also be affected by other agents of change, such as invasions of exotic species that may affect fuel loads. Simulation modeling of climate change impacts on fire regimes in the Australian Capital Territory (ACT) predicted that a 2°C increase in mean annual temperature would increase the landscape measure of fire intensity by 25%, increase the area burnt, and reduce intervals between fires.
- Climate change and changed fire regimes will have complex feedback (positive and negative) interactions with biodiversity, with different potential outcomes for different Australian biomes. There may be increased risks to both interval and intensity-sensitive species, as a consequence of changed climate and changed fire regimes. Climate change will probably have the most significant impacts on both the fire regimes and biodiversity of sclerophyll dominated vegetation such as the forests of south-eastern Australia and south-west Western Australia.
- Managing fire regimes to reduce risk to property, people and biodiversity under climate change will be increasingly challenging. In Australia, management of fire regimes for biodiversity conservation has variously emphasized fire detection and suppression, and fuel management. There needs to be an enhanced research effort on the complex interactions between fire, biodiversity, people, fuel management and land use change, to help meet these challenges.

Recent national analyses of the impact of climate change on Australia's biodiversity have stressed the need to consider changing patterns of disturbance in addition to the impacts of climate change itself. Although there is emerging evidence that climate change is causing shifts in fire regimes across the world, and causing changes to species distributions, there is a paucity of research on the interactive effects of climate change and disturbance regimes on biodiversity, and how biodiversity managers may adapt to changing circumstances.

This report investigates the possible future impacts of climate change on the frequency and intensity of fire in Australia, and the consequences of such change (i.e. fire frequency, spread and intensity) for Australia's biodiversity. It has been prepared by a multidisciplinary research team in consultation with state and federal agencies. The report identifies the key drivers of fire regimes and their potential to be affected by climate change. Four regional case studies, representing varied Australian landscapes with different current fire regimes and projected impacts from climate change, are used to illustrate the range of interactions that can occur between climate change, fire and biodiversity. Findings from these case studies are used to develop a national framework for management of fire to protect people, property and biodiversity.

Increasing our understanding of the interactions between climate change, individual fires and biodiversity in driving fire regimes is imperative if we are to manage fire effectively in Australian landscapes. The effects of climate change are likely to make fire management and biodiversity management more complex in the future, but also increasingly important if the objectives of biodiversity conservation and protection of people and property are to be achieved.

## Climate change, fire weather and fuels

Most of the observed increase in global average temperature since the mid-20th century is very likely due to the observed increase in anthropogenic greenhouse gases. Over the coming decades, climate change is forecast to result in a 'warmer and drier' scenario across much of Australia with impacts on biodiversity and fire regimes.

Climate change will affect individual fires through the effect of changes to temperature, rainfall, humidity and wind (the fire weather components); through increases in atmospheric $CO_2$; and changes in moisture, on vegetation and therefore fuels that feed fires. Fire regimes will also change, because individual fires are expected to change in their characteristics.

Fire regimes across Australia currently differ because of variation in four key drivers: (i) the rate of vegetation (and hence fuel) growth; (ii) the rate at which fuels dry; (iii) the occurrence of suitable fire weather for the spread of fire across the landscape; and (iv) ignition. Consequently, fire regimes in some regions are constrained primarily by availability of fuel, and in others by occurrence of periods of suitable weather. Climate change is expected to change fire regimes by its effects on these four key drivers.

It is already apparent that climate change is affecting fire weather. For example, in south-eastern Australia the severity of fire weather (as measured by the annual total Forest Fire Danger Index) has risen by 10 to 40% across most sites from 2001 to 2007 compared to the period 1980 to 2000. Climate change projections suggest the incidence of extreme fire-weather days will continue to increase by 5 to 65% by 2020, and by 100 to 300% by 2050 in south-eastern Australia, depending on warming scenarios. Similar upward trends in the severity of fire weather are also occurring in other areas of Australia, and fire seasons are likely to become longer and more intense. With increased incidence of severe fire-weather days as a consequence of climate change, we may generally expect a higher frequency of fire if the other conditions for fire (i.e. sufficient fuel loads and ignition) are met. Analyses using landscape fire models that can incorporate such climate change projections show that a moderate warming scenario (2°C increase in mean annual temperature) increased the area burnt and decreased the interval between fires in the Australian Capital Territory.

The effects of climate change on fuel loads are complex. The amount and type of fuel is determined by the balance between productivity, decomposition and consumption. Vegetation production (and hence fuel load) tends to be linked with annual rainfall. Alteration in rainfall patterns from climate change will influence the fuel load present and the rate at which the fuel dries. In areas projected to receive more rainfall, the fuel load may increase, such as in parts of northern Australia. In other regions, such as south-west Western Australia where rainfall is projected to be lower, it is anticipated that fuel loads may decrease.

In contrast to declining rainfall, elevated atmospheric $CO_2$ may increase productivity by its fertilisation effect on vegetation, thereby increasing fuel loads. It may also favour woody plants and shrubs over grasses, thereby altering the amount and distribution of fuel because woody plants, shrubs and their associated leaf litter will support a higher fuel load in landscapes. However, increasing aridity and drought across much of Australia may reduce the potential benefits of elevated $CO_2$ on fuels. The relative carbon and nitrogen content of leaf litter may also change, resulting in slower leaf litter decomposition or reduced palatability to herbivores. It is uncertain whether these factors will result in higher or lower fuel loads, but it is likely that the outcomes of these interactions will vary at landscape scales and in different regions of Australia.

## A national framework for evaluating climate change impacts on fire regimes – the 4 switch model

The key drivers of fire regimes can be incorporated into a national framework to evaluate the effects of climate change on fire regimes at landscape levels. The four key drivers of fire regimes are: (i) the rate of vegetation (and hence fuel) growth; (ii) the rate at which fuels dry; (iii) the occurrence of suitable fire weather for the spread of fire across the landscape; and (iv) ignition. These can be conceptualised as a sequence of conditional processes (or switches) that are required for fires to occur – the '4-switch' model. Each condition is a switch, and all four switches must be simultaneously 'on' in order for landscape fire to occur. Should any switch be 'off' in a particular locality, fire will not occur. Identification of the 'limiting' switch in different ecosystems is important because switches are likely to be differentially sensitive to climate change. The model explains why fire regimes at landscape scales vary in differing biomes.

This model is applied to four biomes, where suitable data and diagnostic capacity exist, to assess how fire regimes may respond to projected climate change: the tropical savannas of northern Australia, the arid hummock-grasslands and shrublands of central Australia, the semi-arid grassy woodlands of inland eastern Australia, and the temperate sclerophyll forests of south-eastern Australia. These biomes differ not only in vegetation and current fire regimes, but also in projected impacts of climate change on regional climate – particularly rainfall and temperature.

In tropical savannas, climate change may not have major effects on area burned and fire frequency. This is because the primary climatic and fuel drivers of fire (biomass, low moisture, spread capacity), as determined by the annual wet–dry climate, are non-limiting on an annual basis. Similarly, in arid regions, drought and fire weather are essentially non-limiting on an annual basis, and landscape fire will continue to be limited to periods following above average rainfall, even under climate change scenarios. In grassy woodlands, outcomes will depend on prognoses of available moisture and land use, and how these affect fuel amount and continuity. In southern sclerophyll-dominated vegetation, where both overstorey and understorey are dominated by woody plants, the primary effect of climate change on fire regimes will stem from the projected increase in the frequency of occurrence of days of extreme fire weather, which has the potential to increase area burnt and therefore reduce the intervals between fires.

## Climate change, fire regimes and biodiversity – four case studies

Impacts of climate change on biodiversity will be complex but there are several alternative outcomes: species may go extinct, they may persist in the landscape, or they may migrate. All of these potential responses to climate change will also interact with changing fire regimes.

Fire has been an integral part of the Australian landscape for millions of years and has been one factor that has driven the evolution of biota. A critical concept in relation to fire and biodiversity is that of the functional group – a group of species that share similar life history traits (e.g. age of sexual maturity, reproductive lifespan, longevity, productivity) that determine their response to the environment and fire. Species persist in the landscape in the face of recurrent fires by a variety of mechanisms. Different plant and animal species can be vulnerable to variation in the components of the regime – intensity, type, season or interval between fires. Some plant species resprout after fire ('resprouters'); others rely entirely on seed after fire ('seeders'). Similarly, animals persist in fire-prone landscapes by a variety of mechanisms. Some species are sensitive to variation in fire frequency, others to variation in fire intensity. Altered fire regimes are likely to contribute to changes in the composition of species assemblages. Changes to fire regimes may have different effects on different species, which lead to changes in ecosystems as we currently know them.

Positive and negative feedback interactions between climate change, biodiversity and fire regimes are likely. Changes in biodiversity composition resulting from the spread of exotic grasses, shifts in species distributions, and the effect of elevated atmospheric $CO_2$ on vegetation will affect the key drivers of fire regimes in Australia.

The interactions between climate change, fire regimes and biodiversity were explored in four case studies. The case studies were designed to illustrate the current understanding of climate, fire regimes and ecosystem dynamics in biogeographic regions characterised by different climate, vegetation and fire regimes, and where suitable data and diagnostic capacity are available to identify potential approaches to the problem.

## Case study 1: Alpine ash forests of the south-east highlands

The tall alpine ash (*Eucalyptus delegatensis*) forests of south-eastern Australia are relatively well-known ecologically. Alpine ash is a 'seeder' species – it can only propagate by seed that has been stored in the crown of the trees, and if the crown is completely scorched or flame-defoliated the entire population will be killed. Seed from the canopy is shed rapidly after a fire event to start the new cohort, and therefore stands of trees are often one age or have a limited series of ages connected to previous fires. Up to ten years are required for juvenile trees to reach maturity and produce seed. If the trees are killed before seed production begins, after it ends or during a year of very low seed production, then there is no seed for the next cohort and the species becomes locally extinct, unless recolonisation can occur from a nearby source population.

Elevated atmospheric $CO_2$ may enhance the productivity of alpine ash forests, resulting in higher fuel loads. With changes in weather towards drier and warmer conditions, the chance of fire occurrence would be expected to rise. All these changes point towards an increased risk of local extinction of alpine ash if fire intervals fall below the period needed for juveniles to flower and seed. However, the species may be able to migrate upward (altitudinally) in response to warming.

Alpine ash is an ideal target species for monitoring, and reacting to, any changes observed in its distribution as a consequence of changes to climate and fire regimes.

## Case study 2: Mediterranean forests and shrublands of south-west Western Australia

South-western Australia is one of 25 global biodiversity hotspots, and its unique flora and fauna are considered to be particularly vulnerable to the impacts of climate change. The vegetation in the south-west of Western Australia ranges from tall forest to shrubland depending on availability of moisture and soil nutrients. Climate change is already apparent in this region, with temperature increases and rainfall decreases in the latter part of the $20^{th}$ century amongst the fastest in Australia.

Continued warming and drying in this region will impact on fire regimes in several ways. A higher frequency of high to extreme fire weather days is likely to result in reduced intervals between fires. Effects of climate change on fuel loads (a key driver of fire regimes) are more complex. Reduced fuel loads and reduced post-recovery rates of plants will occur if drought retards the growth of plants. However, this may be offset by the stimulus to plant growth by the fertilisation effect of elevated atmospheric $CO_2$. In dry woodlands and shrublands the effects of drought might more than offset any potential $CO_2$ fertilisation effects. In contrast, in wetter forest areas, plant growth and hence fuel conditions might be maintained because of greater water use efficiency of vegetation brought on by $CO_2$ fertilisation.

A warmer, drier climate will make sensitive and restricted habitat types – including granite outcrops, riparian zones and wetlands – more vulnerable to fire. These habitat patches support a high proportion of species with fire-sensitive populations of plants (i.e. obligate seeders) and habitat-dependent fauna (e.g. quokkas and some ground-nesting bird species). Projected climate shifts are also likely to increase the time to reproductive maturity for many perennial plant species (by slowing growth rates), so that the estimated minimum safe fire interval may increase. However, with likely reduced intervals between fires, the greatest impacts are likely to be on obligate seeder (fire interval-sensitive) plant species, although even resprouter species (fire-tolerant) may also show gradual population decline.

Given the high biodiversity values of the south-west of Western Australia, further research is required to investigate the interactive effects of climate change and fire regime change as threats to the biodiversity of this globally important region.

## Case Study 3: Tropical savannas of northern Australia

The vast, sparsely populated landscapes of northern Australia are dominated by savannas that have a more or less continuous cover of grasses and a variable cover of trees. Fire regimes in the savannas are driven by the monsoonal, wet–dry tropical climate. During the wet season, plant growth is high and fuel accumulates that dries off each dry season. Conditions of low humidity, high temperature and continuous fuels cause extensive fires in the dry season.

Current fire regimes in savannas differ depending on the type of savanna, dominant species of grass and land use. The mesic savannas of the Kimberley, the Top End of the Northern Territory and Cape York Peninsula have an annual abundance of tall tropical grasses – where neither fuel nor fire weather is limiting – and average fire frequency is about one year in two. In the semi-arid savannas of Western Australia and the Northern Territory, fire frequency is about one year in four. In the semi-arid savannas of Queensland – where properties are smaller and land-use intensive – fire frequency is less than one year in ten. In arid savannas, dominated by hummock and spinifex grasses, fires are infrequent, and occur primarily in or following years of above-average rainfall. Across the savannas, land use change (e.g. intensification of grazing, spread of exotic grasses) may exert a stronger influence on future fire regimes than climate change. For example, exotic grasses such as gamba grass (*Andropogon gayanus*) can increase fine fuel loads by five times in tropical savannas.

While there is general resilience in the vegetation to variation in fire regimes, there are also some plant species and vegetation types that are highly sensitive to variation in fire intensity and/or interval between fires (e.g. native cypress (*Callitris sp.*), some monsoon thickets, heathlands of the Arnhem Land Plateau). Certain faunal groups, such as small mammals, are also sensitive to short fire intervals. Biodiversity monitoring programs have detected declines in the ranges of some faunal groups over the past decade; increased fire frequency and area burnt have been linked to this phenomenon, and are predicted to worsen with climate change. Targeted, active fire management strategies and programs for mitigation of fire and biodiversity monitoring programs will become even more important under a regime of changing climate.

## Case Study 4: Warm temperate sclerophyllous forests and woodlands of the Sydney Basin

The Sydney basin has a high diversity of vegetation types and species distributed along rainfall and temperature gradients, including high numbers of obligate seeder plants. A warmer and drier climate may favour obligate seeders over resprouters, changing the vegetation composition to types that are more fire-sensitive. Fires are also likely to become more frequent and burn larger areas with consequences for biodiversity.

Simulation modelling indicates potential for the area burnt to increase as a result of climate change. Despite this, the likely shifts in the intervals between fires may not be sufficient to significantly increase the risk of extinction (at the landscape scale) of interval-sensitive plant species (assuming that prescribed burning is held at current levels). However, in contrast, increased intensity of fires is likely to have a marked effect on the survival of intensity-sensitive species. Crown-scorching fires are predicted to increase by up to 20% under a high emissions scenario with impacts on species that are sensitive to canopy-scorching fires (e.g. tree-dwelling mammals).

Elevated risks to soil stability, people and property may result from climate-induced changes to fire regimes. The effects of warming and drying on fire-weather, and hence area burned, may outweigh any decrease in fuel loads caused by declining moisture.

In their review of climate change impacts on the National Reserve System, Dunlop and Brown (2008) indicated that biodiversity in south-eastern Australian sclerophyllous systems (i.e. hard-leaf vegetation) was potentially more susceptible to the effects of climate change than other regions in Australia. Thus, in this region, there is the potential for synergistic negative effects on biodiversity due to interactions between climate change and changing fire regimes.

## Summary of findings from case studies

The regional case studies illustrate the diversity of potential impacts of climate change on fire regimes and biodiversity. They highlight the complex interactions that occur between climate change, fire regimes and biodiversity at the landscape scale. Climate change is likely to alter fire regimes (in terms of frequency and intensity of fires) through different mechanisms in different parts of the country. This will, in turn, affect the composition and structure of species assemblages. Climate change will also have direct impacts on biodiversity by causing shifts in the distributions of species. Climate-induced changes to biodiversity will also drive fire regimes by changing the amount and composition of fuels. Hence climate change, fire regimes and biodiversity have complex feedback interactions (positive and negative) with different potential outcomes for different biomes.

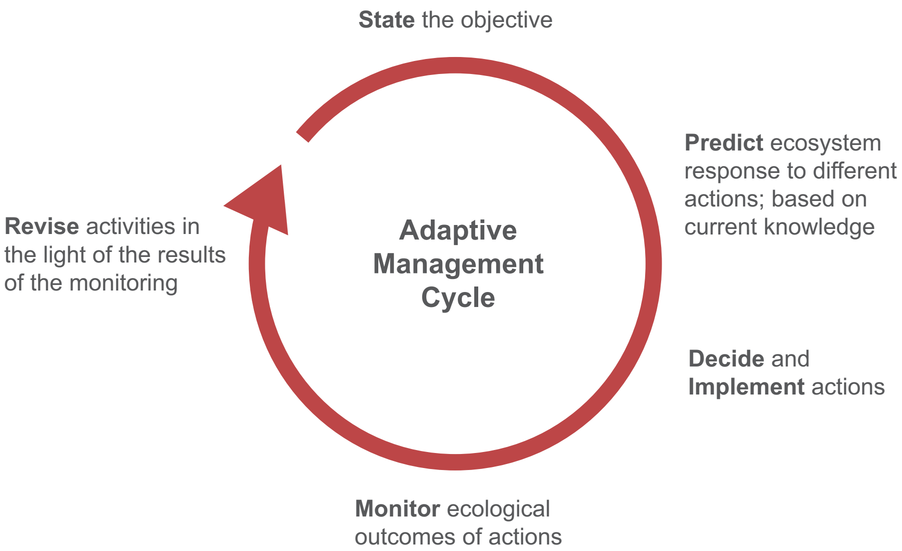
Drought and moderate to severe fire-weather occur annually in savannas, therefore an increase in dryness and severity of fire-weather under climate change is unlikely to have a major impact on fire regimes in this biome. Exotic grasses are more likely to have a major influence on future fire regimes and biodiversity in savannas than is climate change.

Climate change will probably have the most significant impacts on both the fire regimes and biodiversity of sclerophyll-dominated vegetation such as the forests of south-eastern Australia and south-west Western Australia. In these forests, projected increase in very high and extreme forest fire-weather danger as a consequence of climate change is likely to increase area burnt and fire frequency. Stronger or more frequent drought may reduce the rate of post-fire regeneration. Thus, there may be increased risks to both interval and intensity-sensitive species as a consequence of changed climate and changed fire regimes.

## A response framework for fire management for biodiversity conservation in a changing climate

The regional case studies show that management of fire in areas managed for biodiversity conservation is likely to become more complex. There is uncertainty because of the complexity in the interactions between climate change, fire and biodiversity. Fire management will need to embrace this uncertainty.

An adaptive management approach is the ideal for managing fire in a changing landscape because it inherently deals with complexity, uncertainty and competing demands on resources. The key features of an adaptive management framework are as follows.



## Biodiversity management objectives

The traditional management objective for areas protected for biodiversity conservation has been to manage species assemblages as they are. This approach has assumed a relatively static environment where a decline in a population or extinction of a local species is attributed to local changes in the landscape – such as invasion of exotic species, the impact of fire or other ecological factors.

However, in the context of climate change, the environment will be in an increased state of flux. The current conservation goal of minimising the risk of extinction of species from an area will need to consider how persistence of species within specific areas – and migrations of species into and from neighbouring regions – will be affected by climate change and by potential changes to fire regimes. However, there are no prescriptive, generic 'solutions' to the problem of mitigating risk to multiple values and assets posed by changes to fire regimes as a consequence of climate change in areas managed for biodiversity conservation.

Key principles learned over recent decades in conservation biology have ongoing application to the question of climate change, fire and biodiversity. Management may need to focus on minimising species loss by maximising habitat complexity, as suggested by Dunlop and Brown (2008). The habitat aspects of the landscape surrounding protected areas will also need to be considered. Understanding thresholds and domains of concern for impacts of disturbance regimes on biodiversity will be important in this respect.

## Monitoring ecosystem state

Monitoring of the ecosystem state will need to assess the vulnerability of different ecosystem components (i.e. species, functional groups, plant and animal communities) to changes in fire regimes. Principal actions will be to:

- monitor the spatial and temporal components of fire regimes at landscape scales (e.g. long-term trends in fire weather and fuels).
- identify and monitor indicator species – various taxa and communities that are sensitive to different components of the fire regime (i.e. intensity and frequency components).
- understand what thresholds of fire intensity and period of fire interval (domains) will have serious impacts on biodiversity.
- identify the trade-offs that will be necessary to minimise risk to multiple landscape values (i.e. the protection of lives, property and biodiversity).
- develop and refine methods and approaches to evaluate the effectiveness of fire management actions to achieve the stated objectives.

All of Australian states and territories are rapidly developing fire management systems that identify a desired fire interval for biodiversity conservation. However, the development of intensity components of regimes for biodiversity management remains a significant challenge.

## Assessing the drivers of change

Four key predictions on the impacts of climate change on fire regimes can be tested within an adaptive framework: (i) fire weather will become more severe; (ii) fuel mass and type will change due to changes in moisture; (iii) ignitions will change as a function of changing climate and patterns of human settlement; and (iv) the distribution of fire regime components (e.g. interval or severity classes) across the landscapes will change as a consequence of changing fire regimes.

Techniques and data sources are available to monitor all four predictions from meteorological records, field survey, or using remote sensing or geographic information systems (GIS). This information is required now to provide a baseline against which to measure and assess change.

Knowledge of the current distribution of interval and severity classes of fire regime across protected areas can be used as an indicator of habitat suitability for biodiversity conservation management.

## Prescribed burning

Prescribed burning will continue to play an important role in fire and biodiversity management. Prescribed burning can have major effects (positive and negative) on biodiversity, water catchments, and protection of people and property.

There have been calls, especially in south-eastern Australia, for more prescribed burning in the landscape to reduce fuel loads, partly as a strategy to mitigate fire risk in the face of climate change. However, in such temperate Australian forests, relatively large amounts of prescribed burning would be needed to achieve modest levels of risk mitigation for urban and other assets. For example, modelling of the Sydney basin suggests that, to counteract the effects of a high emissions climate change scenario, large increases in prescribed burning (i.e. > fivefold) will be needed. Such an increase may not be feasible based on costs, resource avaliability and other landscape constraints.

The relative benefits and costs of prescribed burning, and its effectiveness in achieving multiple land management goals in different land tenures, requires much more research. Potential changes to fire regimes as a consequence of climate change provide an excellent opportunity to further this research.

## The next steps

A review of the approach to the problem (sections 2 and 3) and technical analyses (sections 4 to 6) constitute the major component of this report. Further research could explore the key management questions: the objectives of biodiversity management, the adequacy of the current management frameworks to deal with the uncertainty of climate change, identification of vulnerable biodiversity assets in the landscape and how altered fire regimes may impact upon them, and the role of fire management – prescribed burning in particular – in achieving conservation aims and mitigating the effects of unwanted fire regime changes.

## Seven priority areas for action

There are seven priority areas for action. One relates to the current state of national fire regimes, three to ecosystem dynamics and three to ecosystem management.

1. **Determine Australia's fire regimes.** Without baseline data, any knowledge of the effects of climate change on fire regimes is necessarily limited.
2. **Determine the potential impact of climate change on fire weather in more regions of Australia.** These can be undertaken using the protocols used for south-eastern Australia summarised in this report.
3. **Evaluate the relative importance of elevated fire danger, elevated atmospheric $CO_2$, and changing moisture availability as determinants of future fire regimes.** This requires much more research and analysis, using regional climate change scenarios, regional fuel change scenarios, and spatially explicit fire and biodiversity models.
4. **Examine the vulnerability of fauna to changes in fire regimes.** Knowledge and assumptions regarding fire regime effects on plants do not necessarily apply to fauna, where further research and field validation are required.
5. **Review and assess current capacity to accommodate change.** The climate change, fire regime and biodiversity scenarios outlined raise the issue of how to manage fire within reserves and other public lands. This report has deliberately avoided providing prescriptive actions because given the uncertainties, the potential options require concerted discussion between researchers and land managers at national, state and territory fora. This is the number one priority action for further research. .
6. **Explore approaches to domain and thresholds of concern.** This could begin immediately, in a range of conservation reserves. In addition to thresholds for interval, thresholds for intensity, fire season and fire type need to be explored.
7. **Undertake benefit cost-analyses of potential management responses.** This issue is particularly acute, because all aspects of fire management are resource-intensive, yet there have been few detailed analyses of returns on management investment.

# 1. INTRODUCTION

## 1.1 Aims, background and rationale

This report investigates the possible future impacts of climate change on fire regimes in Australia, and the consequent potential impacts of such change in fire regimes on Australia's biodiversity. The project is one of several Department of Climate Change projects addressing the issue of climate change and Australia's biodiversity, and is a companion report to two recent publications – 'Australia's Biodiversity and Climate Change: A Strategic Assessment of the Vulnerability of Australia's Biodiversity to Climate Change' (Steffen *et al.* 2009) and 'Implications of Climate Change for the National Reserve System' (Dunlop and Brown 2008; http://www.climatechange.gov.au/impacts/publications/pubs/nrs-report-overview.pdf). The work presented in this report has been undertaken by a research consortium from CSIRO and universities across Australia, in consultation with a number of state and federal agencies through the Climate Change in Agriculture and Natural Resources Working Group (CLAN).

The project arose because of concerns that many Australian species of plants and animals may become extinct as a result of the rapid changes in climate taking place now and that are projected to continue to take place for decades into the future. Because fire is a feature of Australian landscapes, is a component of biodiversity management, and is likely to be affected by climate change, examination of the impact of climate change on Australian biodiversity necessarily involves a consideration of fire. The key questions, then, are how might climate change affect fire regimes, how might these changes affect biodiversity, and what are the implications for the management of biodiversity? Addressing these questions is a considerable national challenge, given the complexity of climate change–ecosystem interactions, and the paucity of research explicitly undertaken on climate change, fire and biodiversity.

As climate changes, the world's biodiversity faces a number of threats, and adapting the management of biodiversity assets, especially within protected areas, to the challenges of global climate change is an internationally significant issue (Hannah *et al.* 2007). Species may go extinct, and there is considerable evidence for the global scale of this threat. On the other hand, species may persist in the landscape through longevity, migration (potentially taking them out of their current protected area boundaries) and/or genetic adaptation. However, climate change will influence disturbance regimes (Dale *et al.* 2001) hence the capacity for species and ecosystems to persist in the face of climate change is likely to be affected by interactions between climate change and disturbance, particularly changed fire regimes.

There is a considerable body of research that has assessed the risk that climate change poses to global biodiversity, and assessment of relative exposure to risk by biome (e.g. Thomas *et al.* 2004). Area burnt is strongly dependent on weather variables (Pausas 2004; Flannigan *et al.* 2005). There is emerging evidence that global warming is causing, and will continue to cause, increases in the severity of fire-weather (Beer and Williams 1995; Flannigan *et al.* 1998; Williams *et al.* 2001; Wotton *et al.* 2003; Hennessy *et al.* 2005; Pitman *et al.* 2007; Nitschke and Innes 2008). Fire regimes in the USA have changed as a consequence of general warming and earlier onset of spring (Westerling 2006). Recent trends towards warmer and drier climates have also been associated with increases in fire frequency and/or area burnt in north American boreal forests (Amiro *et al.* 2001), and the Iberian Peninsula in the Mediterranean Basin (Pausas 2004). In temperate South America, years of widespread fire have been shown to be dependent on drought at monthly, seasonal, annual and supra-annual time scales that can be driven by either decreased precipitation or increased temperatures (Kitzberger and Veblen 2003; Lara *et al.* 2003; González and Veblen 2006).

However, there has been little research on the interactive effects of climate change and disturbance regimes on biodiversity (Nitschke and Innes 2006). There is also a paucity of knowledge about how biodiversity managers may adapt their principles and practices to mitigate the potential threats to biodiversity that are posed by climate change-induced changes to fire regimes. Nevertheless, interactions between climate change and fire can be linked causally, and will impact on biodiversity conservation in the coming decades (Nitschke and Inness 2006).

Recent reports on the impact of climate change on Australia's biodiversity have stressed the need to account for changed disturbance regimes (Dunlop and Brown 2008; Steffen *et al.*2009). Recent research work on global vegetation models has also stressed the need for incorporating fire into vegetation dynamics models (Thornicke *et al.* 2001; Bond *et al.* 2005; Scholze *et al.* 2006; Lenihan *et al.* 2008 ).

To address these knowledge gaps, and to initiate a national approach to examining the potential impacts of climate change on fire regimes and biodiversity, this report aims to:

- review the current state of research on climate change–fire–biodiversity, identify and use relevant data sources and modelling frameworks, and highlight the issues involved in developing this research field
- present a possible national framework at the broad biome level within which to assess the potential impacts that climate change-induced fire regime change will have on biodiversity management
- present preliminary scenario analyses at ecosystem scale – based on published evidence of the fire ecology, datasets and models, and on original technical analyses – for key biomes and ecosystems in different parts of Australia
- discuss the implications of our findings for the management of biodiversity, and propose a response framework for land managers to identify and accommodate the challenges posed to biodiversity assets by changing climate and fire regimes
- identify future research and development needs.

This is a preliminary analysis. Although national in outlook, we cannot address in detail the potential impacts of climate change on the fire regimes and biodiversity of all regions of Australia. We do, however, present results that are pertinent to major, broad bioregions of Australia, such as the sclerophyllous vegetation of the south-east and south-west, the arid and semi-arid zones, and the tropical savannas. Our results have clear implications for the management of biodiversity, both within and outside of protected areas, in the face of changing regimes of climate and fire, and we discuss the importance of devising adaptive strategies to cope with change. However, given the uncertainty associated with our preliminary findings, we do not offer prescriptive pathways for management.

## 1.2 Interactions between climate change, fire regimes and biodiversity: a conceptual framework

This section presents the basic conceptual framework for analysing the various ways in which climate change may affect fire regimes, and how this in turn may affect biodiversity – what we consider to be the major elements of the problem – and (in the next section) how we have approached the problem.

### 1.2.1 Climate change impacts: a complex of cascading interactions

Climate change will drive biological responses in complex ways, by potentially changing physiological and phenological responses in both plants and animals, by changing their geographic distributions, and by affecting disturbance regimes (Hughes 2000, 2003). The manner in which meteorological and atmospheric changes will drive changes to fire regimes and biodiversity is no exception, because climate, weather, atmospheric $CO_2$ and fire (as an agent of disturbance) will all interact, and the outcomes of these interactions are uncertain. Indeed, a generally recognised feature of climate change and its impacts on ecosystems and society is that potential impacts are complex and uncertain. A key component of managing fire and biodiversity in the face of potential climate change impacts is to recognise such complexity and uncertainty.

Dunlop and Brown (2008), in dealing with the impacts of climate change on the National Reserve System, preface their analyses by proposing a 'cascade of impacts' framework for understanding the various ways in which climate change may affect ecosystems and people. The effects include changes in the environment (including climate and disturbance) that affect ecological interactions, which in turn affects societal values. The model framework describes the direct flows and indirect impacts via feedbacks, including adaptation and mitigation actions, and is illustrated in Fig. 1.1.

Dunlop and Brown (2008, pp. 26–27) describe these impacts thus:

*Environmental impacts:* the changes arising from increased greenhouse gas (GHG) concentrations that drive impacts on biodiversity. They include changes in CO2; temperature and rainfall regimes; fire regimes; and sea temperature, chemistry and level. These impacts clearly combine with other (non-climate related) environmental stresses on biodiversity, and are affected by feedbacks from population and ecosystem impacts (e.g. affecting hydrology and flammability – below).

*Biological impacts:* the direct changes to biology of organisms arising from environmental changes. They include changes in physiology and the timing of lifecycle events (phenology).

*Ecological impacts:* result from changed interactions between organisms and their environment, including other species. They include changes in breeding, establishment, growth, behaviour, competition and mortality. These impacts result directly from climate change-related impacts (above), and indirectly via interactions with other species that are affected by climate change leading to changed competition, food, habitat and predation. These indirect impacts can be represented as a feedback from population impacts and ecosystem impacts (below) to ecological impacts. For some species, these indirect impacts may be stronger than direct impacts. Ecological impacts are also affected by how climate change impacts interact with other stresses.

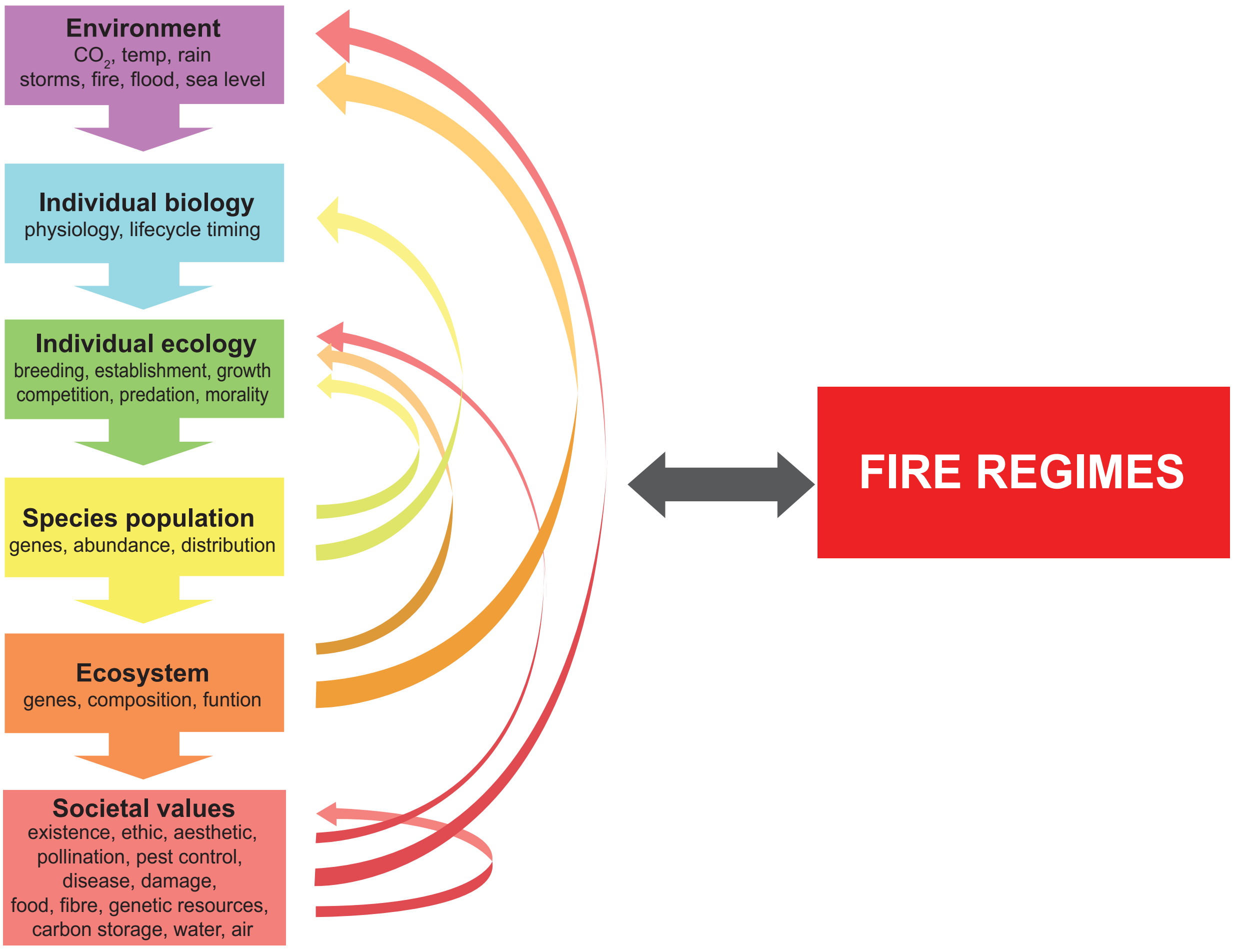



**Fig. 1.1. Schematic representation of cascading impacts resulting from environmental changes caused by climate change.** The direct flow of impacts is represented by large arrows. Important indirect impacts are shown as feedbacks. Climate change–fire–biodiversity effects operate throughout the complexity cascade, as they are affected directly by environmental changes to disturbance regimes (Box 1), interactive effects on species and ecosystems (Boxes 2 to 5), and by the choices society makes about various landscape values (Box 6). Fire regimes interact with all of these processes. Source: Dunlop and Brown (2008, p. 27)

*Population impacts:* the ultimate impact on species in terms of changes in gene frequencies, abundance and distribution.

*Ecosystem impacts:* changes in the identity, composition, structure and function of assemblages and ecosystems.

*Value impacts:* represent the impacts on human well-being, the reason society cares about climate change and biodiversity. These include 'non-use' values, for example: existence of species and ecosystems; land ethic; 'caring for country'; stewardship of the planet for future generations; and aesthetics.

### 1.2.2 Climate change, fire regimes and biodiversity: the elements of the problem

Our ecological understanding has grown quickly over the past half century but there are still wide gaps in our knowledge concerning the interactions between climate, climate change, biodiversity, species' distributions and disturbance regimes. This means that, with respect to climate change, fire and biodiversity, there are no 'off-the-shelf' models or other evaluation tools that can assist researchers and managers explore the consequences of these complex interactions. However, Fig. 1.1 serves as a guide to outlining how the problem of climate change–fire–biodiversity may be conceptualised; identifying the key elements that are needed for any quantitative assessment–analytical approaches, the relevant climate science, key ecological concepts, quantitative models and datasets that can be utilised and scrutinised, where biodiversity management may sit in this climate of change, and ways in which society may respond to the challenges identified.

#### Analytical approaches

To assess the impacts of climate change on ecosystems, various approaches and tools can be used. Our understanding of current meteorological circumstances and developing projections for further change is based on sophisticated computer-based climate models, or Global Circulation/Climate Models (GCMs). In addition, examination of current meteorological records (at daily, yearly, decadal or centennial time scales) enhances our understanding of the nature and direction of climate change. To understand the effects of climate change on our landscapes, various ecological models (including fire spread models) are used to explore scenarios of change (e.g. Pitman *et al.* 2007). In addition, the global change research community uses laboratory and field experiments to examine the impact of changing levels of temperature, moisture and $CO_2$ on species and ecosystems (e.g. Norby *et al.* 2005; Stokes *et al.* 2005). In this report, we have used elements of each of these.

#### Documentation of climate change

The world has warmed in the past century, by about 0.75°C, and is likely to continue to warm in the coming century. These changes are highly likely to be the result of increased emissions of GHGs – such as $CO_2$, methane, nitrous oxide and other gases – as a consequence of human activity. $CO_2$ concentration in the atmosphere has risen from about 280 ppm in 1750 to about 384 ppm in 2007; 70% of this increase has been since 1970 (Solomon *et al.* 2007). These changes affect global meteorological processes, which in turn lead to changes in regional and local climatic conditions (see sections 2.1, 2.2 and 4.2.1 for more detail).

The great majority of observed physical and biological responses to climate change across the world are all in the directions predicted by climate change models – melting glaciers, rising sea levels, and changes in the distribution of plants and animals. With respect to the latter, distributional changes have generally been poleward and uphill, and have also included earlier starts to seasonal migrations and reproduction (Hughes 2000, 2003; Parmesan 2006; Root 2003; Rosenzweig *et al.* 2008).

The documented and projected meteorological changes, and the increases in atmospheric $CO_2$ and other greenhouse gases, all have the potential to affect fire regimes through their effects on fire-weather, via changes in temperature, the amount and seasonality of rainfall, relative humidity and evaporation, wind speed, lightning frequency, and fuels.

#### Fire events and fire regimes

The fire regime concept is central to understanding the ecological impacts of fire. Fires occur as single events – they have a beginning and an end – and as they progress they have measurable properties such as rate of spread, flame height, intensity and ejection of fire brands. However, fires also recur in the landscape, and the fire regime concept explicitly acknowledges this. In terms of ecological impact, then, it is the spatial and temporal patterning of recurrence that is important, not just biological responses to individual fires (Bradstock *et al.* 2002; Gill 1975, 1981). Thus, the fire regime is the history of fires across the landscape, the sequence of fires in the past in relation to their interval, the season of year in which they occurred and their intensity at any one place on the ground.

The components of the fire regime vary widely. In nature, fire intensity may vary a thousandfold or more, intervals between fires may vary from 1 to 300 years or more while 'seasonality' can span many months of the year. Hence, the effects of fire regimes on the environment will vary substantially.

### *Biodiversity and functional groups*

'Biodiversity' refers to the full variety of life from genes to species to ecosystems (Wilson 1988), but in most contexts, including here, it refers more explicitly to the variety of life indigenous to a local area. While often restricted to the variety of higher native plants and animals (as an assumed surrogate of overall biodiversity), it also includes micro-organisms, non-vascular plants like mosses and fungi, invertebrates, and even genes. Biodiversity is arrayed in ecosystems with spatial patterns that extend across landscapes. Ecosystems result from all the interactions that occur between the species that make up biodiversity at any one place in the context of the weather, soils and disturbances that occur there.

The 'functional group' or 'functional type' concept (Noble and Slatyer 1981) is widely used in the study of biodiversity. It describes the life form and the life history characteristics of the diversity organisms, such as mode of regeneration, the age at first reproduction, growth rates and strategies, life forms, size, and longevity.
A commonplace example of 'functional groups' of plants are trees, shrubs, grasses and herbs; grasses and herbs may then be annuals, biennials or perennials. Plants may be obligate seeders, or may be able to regrow vegetatively (resprout) following disturbance, or both. The functional group concept can also be applied to animals by considering traits such as home range, diet and foraging strategy, in addition to reproductive and growth traits. For plant functional types and fires see, for example, Noble and Slatyer (1980) and Pausas *et al.* (2004); for functional type thinking as applied to Australian animals see Corbett *et al.* (2003) and Bradstock *et al.* (2005).

Most species can be allocated to one of a dozen or so functional groups, which simplifies community and ecosystem level analyses considerably. With respect to fire regimes, some species/functional types are sensitive to fire intensity, and others to interval, as discussed in more detail in section 2. However, conservation of biodiversity is in part concerned with the conservation of species, not functional types. So while functional types allow researches and managers to model the effects of fires efficiently, the results eventually need to be applied to particular species in particular areas.

Caption: *Boronia angustiphela* is an obligate seeder that is threatened by increased fire frequency at high altitudes in the Gibraltar Range/Washpool National Park, NSW

Source: Peter Clarke

### Species responses to climate change

We deduce biological and ecological responses to environmental change – whether climate change or change to disturbance regimes – from detailed studies of species' life history, physiology, reproduction, growth and behaviour; from studies of ecological systems; and from palaeoecology. This fundamental biological work tells us that all organisms are able to cope with some degree of environmental change and can maintain basic life processes such as growth, reproduction and dispersal in the face of changes to temperature, moisture, atmospheric $CO_2$ and disturbance.

Beyond particular thresholds, either physiological or disturbance-related, the resilience of individuals or populations is challenged. Individuals may die and, at wider scales, populations may consequently decline or go extinct. Alternatively, species may migrate in response to environmental change, resulting in range shifts, or suffer extensive range reductions, such that they persist only in refugia. A third response is that species may adapt genetically. All three are important in determining the outcomes of climate change and any associated changes to disturbance regimes. With respect to protected areas, such changes may result in the domains of suitability for species no longer remaining within the boundaries of protected areas. Managing fire within a protected area may further affect the distribution and abundance of suitable habitat.

### Data and models

Nationally, it is highly likely that there will be variation between biomes, and ecosystems within biomes, in the sensitivity of fire regimes to climate change. Assessment of this variation in sensitivity at regional scales will require linked climate–fire–biodiversity models, and associated datasets. However, research that links fire regime responses to biodiversity responses is in its infancy. The research field requires both a fire ecology framework within which to frame the climate change impact questions, and a suite of regional fire regime models that are linked to biodiversity models, so that various scenarios can be evaluated. A multiscale, multimodel exercise across biomes has not been attempted for Australia, and is obviously beyond the scope of this report. Some linked models only exist for select regions of Australia. However, there is considerable regional data on projected climate change, and evolving fire–biodiversity models that can be driven by climate data, hence the basic capacity to undertake initial regional scenarios exists. The main body of this report (sections 4, 5 and 6) presents the ecological framework and technical capacity by which an appropriate combination of candidate data sources and models can be developed.

### Societal responses

Society places a number of values on biodiversity: individual species, maintenance of native ecological processes, ecosystem services (soil and water conservation values, carbon sequestration), cultural values and amenity values. Protected areas and off-reserve conservation are both important in biodiversity conservation. The choices and behaviour of people are also important – population continues to grow, especially at the urban interface; and the rural landscape is changing, due to diverse drivers such as 'tree changers' and agricultural intensification.

Biodiversity conservation therefore needs to be viewed in the context of multiple values, multiple land uses and land use objectives, both within and outside of individual protected areas. All land management agencies have a finite pool of resources available at their disposal. Thus, in dealing with climate change–fire–biodiversity interactions, society must recognise: (1) multiple values and objectives; (2) the inherent complexity and uncertainty of biological responses to climate change; (3) that there are risks to some values as a consequence of climate change; and (4) that trade-offs between values will need to be evaluated. Changing markets, e.g. for landscape carbon sequestration, will add to the complexity of biodiversity management.

### Managerial responses

In considering how biodiversity managers respond to the potential challenges that climate change will bring to ecosystems, two fundamental management questions arise:

- Is the focus of biodiversity conservation species or habitats?
- How do we manage ecosystem state to achieve conservation objectives?

Historically, the focus has been primarily the preservation of species. Hence it is species preservation, rather than preservation of vegetation or habitat, which has generally been the primary management objective. Moreover, protected areas such as national parks were never designed as insurance policies against climate change. In the face of this, Dunlop and Brown (2008) argued that managing for habitat heterogeneity will be a critical component of managing for biodiversity conservation in the face of climate change.

With respect to managing ecosystem state, it is axiomatic that the way fire regimes are managed will have impacts on habitat heterogeneity (Bradstock *et al.* 2002), so managing fire regimes is a critical component of managing for biodiversity in the face of climate change. A further question regarding fire management in particular also arises – are the current management frameworks robust and adaptive enough to accommodate the additional uncertainty and complexity that climate change and its impacts will bring to the business of biodiversity management?

Thus, a major managerial response will be to revisit the fundamentals of the objectives of protected area management, and the means by which objectives are achieved. To this end, identification of trade-offs, determination of risk profiles and benefit-cost analyses of management actions will be fundamental components of the development of any management responses to climate change.

## 1.3 How the issue has been approached in this study

We have adopted a fire regime approach to exploring the issue. This approach focuses on the potential impacts that climate change may have on fire-weather, how this might translate to change in fire regimes, and how these interactive effects may impact upon the composition and dynamics of ecosystems. We have considered weather, ignitions and fuels in different biomes of the country. We have examined what the climate–fire regime change scenarios might mean for distribution and abundance of some indigenous elements of the biota and, where possible, compared the relative sensitivity of the biota to shifts in fire regimes as driven by climate change, with the sensitivity to changes in regimes driven by other factors such as changes in the abundance of exotic species, and common fire management practices such as prescribed burning.

To this end, we have used the meteorological record to assess recent change in fire-weather, used various climate datasets and models to asses the likely impact of climate change on fire-weather, applied existing landscape fire models to explore potential climate–fire–biodiversity interactions, and applied the findings from the international scientific literature on $CO_2$ impacts on vegetation to address the question of rising levels of $CO_2$ and fuels. We then discuss the implications of these technical analyses for the management of biodiversity.

The three components – climate change, fire regimes and biodiversity responses – could be considered as corner points in a complex triangle (Fig. 1.2). Each of the two-way interactions – the sides of the triangle – are first examined separately. Section 2 reviews climate change and biodiversity. Section 3 reviews fire regimes and biodiversity. Both sections 2 and 3 draw heavily on recently published sources, e.g. Dunlop and Brown (2008) and Steffen *et al.* (2009) in the case of climate change and biodiversity; Bradstock *et al.* (2002), Andersen *et al.* (2003) and Russell-Smith *et al.* (2003a) in the case of fire and biodiversity. In section 4 we illustrate, using original analyses, how climate change may affect fire regimes, by considering the impact of rising temperatures on fire-weather, how climate change might affect fuels, and a fire modelling framework that will allow the implications of climate change for fire-weather to be expressed spatially in terms of area burnt and intervals between fires. In section 5 we present a national framework within which broad-biome prognostications can be made by considering how climate change may impact upon fire regimes and biodiversity. In section 6 we address the three-way interaction, via four case studies from different parts of Australia: the alpine ash forests of south-eastern Australia, the Mediterranean ecosystems of south-west Western Australia, the tropical savannas of the northern Australia and the sclerophyllous vegetation of the Sydney Basin. In terms of examining biodiversity outcomes of regime change, we present our case studies as heuristic tools, rather than as prescriptive remedies to a potential problem. We have also deliberately concentrated on vegetation, as we believe flora is more proximate to this complex issue than is fauna. In section 7 we present a response framework for biodiversity managers. We stress here that, given the uncertainty of the outcomes of these complex interactions, the framework is designed to stimulate thought about what challenges climate change will bring to fire management in areas managed for biodiversity conservation, and how fire management may be adapted to meet these challenges. It is deliberately not a set of prescriptive pathways or actions.

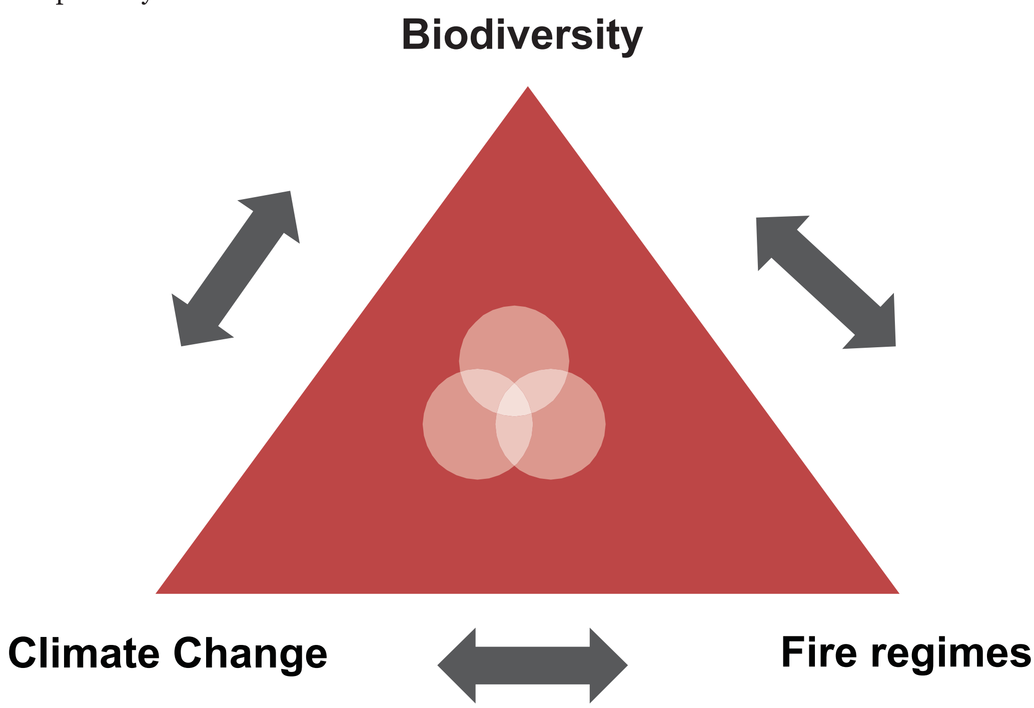



**Fig. 1.2. Conceptual diagram of the climate change–fire regime–biodiversity 'triangle'.** The arrows along the three sides of the triangle represent two-way interactions; the overlapping circles in the centre represent the three-way interaction.

# 2. CLIMATE CHANGE AND BIODIVERSITY: DIRECT IMPACTS

The observed and projected changes to climate, and the potential impacts of climate change on biodiversity, are discussed and summarised in numerous publications, for example Hughes (2000, 2003), Beaumont and Hughes (2002), Walther (2002), Parmesan and Yohe (2003), Root *et al.* (2003), Hickling *et al.* (2005), Lovejoy and Hannah (2005), Parmesan (2006) and Rosenzweig *et al.* (2008). Dunlop and Brown (2008), in their recent report on climate change and the national reserve system, summarised this body of work; this section is based largely on Dunlop and Brown's review.

## 2.1 Documented changes in climate

Climate change over the past century and its likely causes have been been well documented. The globe has warmed by ca. 0.75°C, and there is a high degree of confidence that this has been due to increases in the atmospheric concentrations of $CO_2$ and other greenhouse gases as a consequence of human activity (Collins 2000; Solomon *et al.* 2007). Globally, $CO_2$ has increased from 280 ppm in 1750 to 384 ppm in 2007 with 70% of this increase occurring since 1970. The rate of accumulation of $CO_2$ in the atmosphere is unprecedented in the last 10,000 years (Solomon *et al.* 2007). Globally, temperature has increased by about 0.75°C over the past century. In Australia, average temperatures have increased by approximately 0.9°C since 1910, with greater warming in minimum temperatures (1.2°C) than maximum temperatures (0.7°C) (Dunlop and Brown 2008, quoting http://www.bom.gov.au/cgi-bin/silo/reg/cli_chg/timeseries.cgi).

In addition to changes in mean temperatures, Australia has experienced changes in the frequency of extreme hot and cold weather events. Since 1957 there has been an increase in hot days (35°C or more) of 0.10 days/year, an increase in hot nights (20°C or more) of 0.18 nights/year, a decrease in cold days (15°C or less) of 0.14 days/year and a decrease in cold nights (5°C or less) of 0.15 nights/year (Nicholls and Collins 2006).

Rainfall patterns have changed across Australia between the earlier part of the historical record (1910 to 1950) and the latter part (post-1950). Since 1950, both large and spatially coherent rainfall changes have occurred across the continent. They include rainfall increases of up to 50 mm per decade in the northern third of Western Australia and the Northern Territory, and rainfall declines in excess of 20 mm per decade across much of the eastern seaboard (Smith 2004; Alexander *et al.* 2007; Gallant *et al.* 2007). There was also a notable step-increase in rainfall from the 1950s to the 1970s in south-eastern Australia (Vivès and Jones 2005) and a sustained decrease in rainfall in south-west Western Australia since the 1970s (Pittock 2003). Key contributors to recent changes in Australia's rainfall include variability in El Niño Southern Oscillation (ENSO) activity, enhanced monsoonal activity in the 1970s and changes in other large-scale circulation features such as the Southern Annular Mode (SAM) that affects the passage of fronts across southern Australia.

Other major meteorological changes have also been observed. The frequency of intense cyclones has increased in both the Atlantic and Pacific regions (Easterling *et al.* 2000; Emanuel 2005; Hoyos *et al.* 2006). Along the eastern Australian coast the total number of tropical cyclones has declined since the 1970s, but there has been an increase in the number of very intense systems (i.e. minimum central pressures of 970 hPa or less; Kuleshov 2003). Mean snow cover has declined significantly over the period from 1960–1974 to 1975–1989 in the Australian Alps. Maximum winter snow depth at Spencers Creek in the Snowy Mountains has declined since 1962, with the spring snow depth exhibiting a strong decline trend (i.e. approximately 40% decline in depth since 1962; Osborne *et al.* 1998; Green and Pickering 2002; Nicholls 2005).

These changes are likely to have been caused by both anthropogenic and natural factors. Cullen and Grierson (2008) indicate that rainfall in south-west Western Australia has naturally varied from relatively moist to relatively dry over the past 350 years; modelling studies in the same region have estimated that approximately 40% of the rainfall reductions can be attributed to anthropogenic influences (Cai *et al.* 2003; Cai and Cowan 2006). Anthropogenic changes in SAM may also have contributed to recent declines in rainfall in south-eastern Australia. Anthropogenic warming is increasing the severity of Australian droughts, by raising temperatures (Nicholls 2004). Anomalously warm conditions and associated increases in evaporation made effective rainfall far lower in the 2002, 1994 and 1982 droughts compared to those earlier in the record.

$CO_2$ can also impact on water resources. Gedney *et al.* (2006) presented evidence for increased runoff as a result of suppression of plant transpiration due to $CO_2$-induced stomatal closure in a number of river basins around the world in the 20th century.

## 2.2 Projections on climate change

### 2.2.1 $CO_2$

Atmospheric concentrations of $CO_2$ will increase to between 540 ppm to 970 ppm by 2100 depending on which emission trajectory the world follows (Solomon *et al.* 2007).

### 2.2.2 Temperature

The nature of the Australian climate in decades to come will depend primarily on emissions scenarios – in broad terms, the higher the emissions, the greater the concentration of $CO_2$ and the greater the temperature rise. About half of the uncertainty in the rate of future global warming is due to uncertainty concerning anthropogenic emissions of GHG (Solomon *et al.* 2007; Dunlop and Brown 2008). Even at a low emissions scenario, the Earth is highly likely to experience a 1–2°C warming over the next 50–100 years. Globally, we are tracking at the high end of previous predictions, both in terms of emissions and temperature changes (Rahmstorf *et al.* 2007).

What do these projections mean for Australia's climate? Dunlop and Brown (2008) summarised this as follows. Australia will get hotter; maximum and minimum temperatures in all regions and seasons are expected to increase (Suppiah *et al.* 2007). The annual average temperature is anticipated to increase between 0.2 and 2.2°C by 2030, and between 0.4 and 6.7°C by 2070, depending on the climate model, region and emissions scenario (Suppiah *et al.* 2007). Spring and summer temperatures will increase more than autumn and winter, the warming will be up to 2°C greater inland than on the coast, and night-time temperatures will increase more than day-time temperatures (Suppiah *et al.* 2007). For temperate cities the average number of days above 35°C is expected to increase by 4 to 18 days by 2030, and by 9 to 56 days by 2070.

### 2.2.3 Rainfall

Globally and regionally, the responses of rainfall to rising levels of $CO_2$ are much harder to predict than temperature. Average, seasonality and inter-annual variability in rainfall are all likely to change, but in regionally specific ways. Changes in total precipitation are amplified at the extremes (Easterling *et al.* 2000).

In general, Australia is likely to be drier overall, but there will be regional and seasonal variations. South-west Western Australia, southern South Australia and most of Victoria are very likely to be drier. Tasmania, northern New South Wales, parts of southern Queensland and parts of the Northern Territory may be slightly wetter (mainly in summer and autumn; Suppiah *et al.* 2007).

### 2.2.4 Storms, snow and frost

Large storms and cyclones are expected to be more severe, with higher winds, causing more damage, flooding and coastal inundation (Pittock 2003; Walsh *et al.* 2004). An increasing proportion of rain is expected to fall in more intense events (20 to 30%). There is no clear evidence about regional changes in frequency and movement. Cyclones may move further south (Leslie and Karoly 2007).

Snow cover and extent will continue to decline (Hennessy *et al.* 2003). The total area of snow cover is expected to decrease by 14 to 54% by 2020 and 30 to 93% by 2050. Frost occurrence is expected to decrease overall. However, temperature is not the only driver of frosts; changes in synoptic circulation and decreased cloud cover could lead to increased severity of frosts and changes in their timing in some conditions.

### 2.2.5 Uncertainties

All climate change projections are subject to some degree of uncertainty. Different aspects of climate change impact are subject to different levels of uncertainty, as is implied by the 'cascading complexities' paradigm (section 1.2.2; Fig. 1.1). The further 'down' the cascading chain, the more uncertain are the climate change impacts. For example, the impacts of rising levels of $CO_2$ on global and regional temperatures are better understood, and therefore subject to less uncertainty, than are the impacts of changing climate and $CO_2$ on either plant community level responses, or fire-weather and fuel loads. The interactive effect of projected changes to fire-weather and fuels on fire regimes is more uncertain again, and so on.

## 2.3 Species responses to climate change: extinctions, migrations, adaptations

The impact of climate change on ecosystems will be primarily through the effects on individual species. Moreover, quantitative analyses consistently report that different species and different ecosystems are expected to respond to climate change in very different ways. In response to climate change, species may go extinct, they may persist in the landscape, they may migrate or they may undergo adaptation and evolution.

Current predictions suggest substantial alteration to species' ranges and abundances in response to changed habitat suitability (Beaumont and Hughes 2002, Hickling *et al.* 2005), with the possible extinction of many taxa because of physical (geographical and altitudinal) limits to movement (Thomas *et al.* 2004). Species that make up a community are unlikely to shift together, with substantial time lags and periods of reorganisation (Andrew and Hughes 2005).

Increased atmospheric $CO_2$ is predicted to favour the establishment of woody plants over grasses, and resprouters over obligate seeders, because below ground starch stores may be increased (Hoffmann *et al.* 2000; Bond *et al.* 2003). However, this may depend on disturbance regime. If periods of negative carbon balance are increased through higher fire frequency/intensity, the net effect on resprouters may be negative and under the control of complex interactions. Hoffmann *et al.* (2000) have shown that resprouting is enhanced under elevated $CO_2$ when nutrients are supplied experimentally, but not under nutrient limitation. This pattern could have important ramifications in temperate Australia where low-nutrient soils may counter any enhanced $CO_2$ effect on resprouting ability. Knowledge of these interactive processes in Australian ecosystems is minimal.

Post-fire regeneration during the wet season in *Melaleuca* forests, Kakadu National Park, NT
Source: Barbara McKaige (CSIRO)

Over geological time, species, populations and landscapes have, of course, been changing in concert with changing climates, with extinctions, evolution of new species, and range shifts. Our current patterns of biodiversity are in part the result of such fundamental, long-term evolutionary sifting. However, in contrast to the geological past, we now have fragmented landscapes and high rates of climate change. The observed and predicted global warming is, in geological and evolutionary terms, very rapid. This places serious constraints on the capacity of the biota to adapt and/or migrate in the face of climate change.

Disturbance – in this case fire – will interact with these climatic changes and species' responses to climate change. Other disturbances such as cyclones, floods, droughts and invasion of exotic species may also occur. Ecosystems may be more sensitive to changes wrought by other drivers of change (e.g. where invasive species change fuel conditions) that they are to changes in climate and/or fire regime.

Predicting the outcomes of such synoptic changes to climate and disturbance is therefore extremely difficult. The concepts of the 'fundamental niche' and the 'realised niche' (Hutchinson 1957; Austin 1992) are useful here. The fundamental niche is the distribution that a species could have given only constraints due to climate and soil. Competition and disturbance will affect the realised niche, the envelope of environmental space where a species is present, and which is a subset of the distribution of species if it were unconstrained by factors such as competition and pathogens.

Any climate shifts may not necessarily change the fundamental niche (for example, the temperature range within which it can grow and reproduce does not change). However, the potential distribution – and hence the realised niche in space – may well change. Importantly, the disturbance regime is usually not defined in most models of the realised niche. Under these circumstances, changes to climate and disturbance may have synergistic effects acting together to amplify a result of climate change, or antagonistic effects where disturbance may potentially negate the effects of climate change. Whatever the manner in which climate change, disturbance and species interact, new ecosystems in new locations may arise.

Despite uncertainty regarding climate change impacts, some sort of change is inevitable – the question is what level of change will be detrimental to biodiversity conservation values? Dunlop and Brown (2008; section 4.3) proposed three mental models for conservation managers to assess and deal with change in biodiversity assets as a consequence of climate change. These are:

1. *in situ* changes in relative abundance by most species
2. rapid or long-distance distribution expansion by a few species
3. gradual distribution changes by many species.

All of these potential biological responses will have implications for fire regimes. Disturbance – in this case fire – will interact with these climatic changes and species' responses to climate change. Ecosystems may be more sensitive to changes wrought by other drivers of change (e.g. where invasive species change fuel conditions) than they are to changes in climate and/or fire regime.

## 2.4 Management responses to climate change and biodiversity

With changing patterns of climate, new and evolving domains of suitability for species may therefore lie outside the current boundaries of protected areas. In addition, refugia within protected areas may contract, or species may not be able to migrate across human-constructed barriers. Change will almost certainly alter the proportions of functional types within landscapes, and new functional types (especially exotic species) may enter landscapes – placing further constraints on protected areas to conserve local and regional native biodiversity, and fundamental ecological processes.

The national conservation reserve system was established largely without concern for the sorts of changes in biodiversity assets that might occur as a consequence of climate change. However, the three models of biodiversity change in response to climate change are couched primarily in terms of changes in abundance and distribution, as these are the primary biological drivers that managers are likely to respond to.

Thus, the different mental models can be used to help design monitoring and research programs, and help in scenario-setting processes. Dunlop and Brown (2008) pointed out that this approach is similar to the scenario planning process that underpins the Intergovernmental Panel on Climate Change emissions scenarios (IPCC 2007) and that has become an important part of global strategic planning exercises designed to inform climate change adaptation strategies. We propose that the same principles will apply to assessing and dealing with change in biodiversity assets as a consequence of climate change-induced changes to fire regimes.

Dunlop and Brown (2008) summarised the challenges for biodiversity management in the face of climate change and its associated uncertainties thus: 'Benchmarks for species and ecosystem outcomes may need to be revised, and it will be increasingly less feasible to regard species as static and in equilibrium with other species and the environment, and to assume "all else is constant". This change in the fundamentals of biodiversity will be an issue for research (including the suitability and parameterisation of many biodiversity models) and for conservation planning and management.'

There are numerous existing management responses aimed at coping with the direct effects of climate change on biodiversity. A review is beyond the scope of this report, but at state levels they include various state policy frameworks, e.g. the NSW Biodiversity and Climate Change Adaptation Framework; (http://www.environment.nsw.gov.au/resources/threatenedspecies/0762biodivccadapt.pdf) and the Queensland Government climate change initiative (http://www.climatechange.qld.gov.au/response/about_qccce.html). They also include continental and regional scale corridors (to assist with migration), such as the Great Eastern Ranges Initiative (http:// http://www.environment.nsw.gov.au/ger/background.htm) and the Kosciuszko to Coast (K2C) project (http://www.k2c.org.au); and various species recovery and translocation plans.

There are also biodiversity assessment tools that, although not explicitly tools for incorporating climate change into conservation planning, are nevertheless tools for identifying and mapping areas with high biodiversity values including tracts of habitat that form corridors. One example is the Queensland Environmental Protection Agency (EPA) Biodiversity Assessment and Mapping Methodology (BAMM); http://www.epa.qld.gov.au/nature_conservation/biodiversity/biodiversity_assessment_and_mapping_methodology_bamm/.

There are of course other tools, projects and policies; our particular examples are by no means meant to be exhaustive. However, in assessing this array of climate change adaptation instruments, it is apparent that, within both state and Commonwealth jurisdictions, the instruments invariably deal with fire in only a general way. More importantly, the ways in which current and future fire regimes may interact with biodiversity conservation adaptations to climate change, and the way in which fire regimes may be managed in the future within such programs, are unknown.

# 3. FIRE REGIMES AND BIODIVERSITY

## 3.1 Fire events and fire regimes

Fire has been a factor in the Australian landscape for millions of years. Fire is a product of landscapes (topography and associated vegetation), climate and ignition. Fire – along with climate, relatively poor soils, and geological isolation – has shaped Australia's distinctive flora and fauna (Gill *et al.* 1981; Bowman 2000; Bradstock *et al.* 2002).

The 'fire regime' concept (Gill 1975, 1981) is critical to fire research and management. It recognises that fires recur across landscapes and that the ecological consequences of fire cannot be adequately understood by treating fires as discrete events. Much research has focused on documenting how particular components of fire regimes (e.g. fire intensity, season of occurrence, intervals between fires) vary with environmental factors such as topography, fuels and fire-weather, and land management practices such as prescribed burning.

The fire regime concept is particularly important for investigating how variation in climate may affect fire. That the history of fires affects the outcome of a current fire is a fundamental idea underpinning the management of fire in landscape management. When considering the impact of changing weather and climate on fire, it is important to remember that past fires influence a future fire as much as future climate and weather might. The fire regime concept is also important for investigating the effects of fire on the environment. With respect to biodiversity, the variation in the proportions of landscapes affected by the components of the fire regime is critical to the persistence of species.

## 3.2 Responses of the biota to fire regimes

In this section we review briefly the current understanding of how fire regimes affect biodiversity. The terminology and some concepts are introduced. The field of fire regimes and biodiversity has been influenced strongly by the seminal books by Gill *et al.* (1981), Bond and van Wilgen (1996), Whelan (1995) and Bradstock *et al.* (2002) on fire regimes and biodiversity in Australia. A book by CSIRO's Dr Malcolm Gill (Gill 2008) and a recent special issue of the *International Journal of Wildland Fire* (Williams and Bradstock 2008) on large fires and their ecological consequences are also useful references.

### 3.2.1 Fire and biodiversity: the importance of fire regimes and species life history

Australia is a fire-prone continent (Fig. 3.1). Fires, including large bushfires, have occurred across the Australian landscape for millions of years, and people have been using prescribed fire for various purposes for tens of thousands of years.

Thus, since humans have occupied the Australian continent, there has been an interaction between unplanned fires, prescribed fires and biodiversity. Fire has been a major selective force in the evolution of fire response traits, in both plants and animals (Bradstock *et al.* 2002). However, the manipulation of fire regimes specifically for biodiversity conservation has been a phenomenon of the past 30 or so years (Bradstock *et al.* 2002).

Central to thinking about fire ecology in Australia over the past three decades have been the concepts of the fire regime (Gill 1975) and the functional group (Noble and Slatyer 1981). As stated previously, the fire regime is the history of fires: the sequence of fires in the past in relation to their interval, the season of year in which they occurred and their intensity at any one place on the ground. The functional group concept describes the life form and the life history characteristics of organisms, such as post-fire mode of regeneration, age at first reproduction and longevity.

The fire regime concept is particularly important for investigating the effects of fires on the environment, because it implies that the condition of the environment when the fire arrives is, or can be, affected by previous fires, which, in turn, were, or could have been, affected by the fire before that, and so on. The components of the fire regime vary widely. In nature, fire intensity may vary a thousandfold or more; intervals between fires may vary from 1 to 300 years or more while 'seasonality' can span many months of the year. Hence, the effects of fire regimes on the environment will vary substantially.

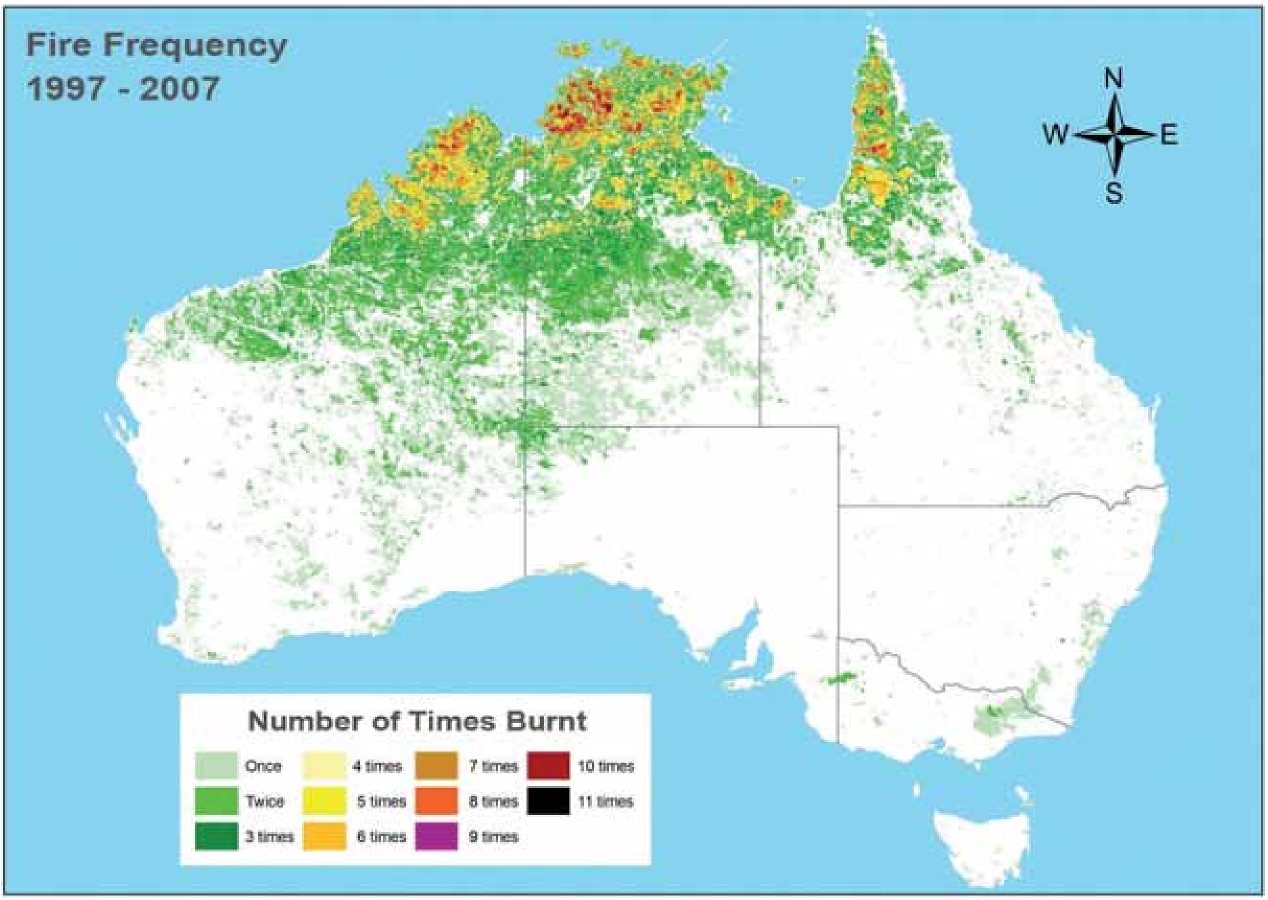



**Fig. 3.1. Fire frequency across Australia, 1997–2007.** Source: Andrew Edwards and Jeremy Russell-Smith, Bushfires NT.

Different plant and animal species can be vulnerable to levels of different components of the regime but not all species behave the same way, even in the same locality. A very common example of the effects of fire regime on biodiversity is that of the impact of interval between fires on plants (typically shrubs or trees) that only regenerate by seed – the so-called 'obligate seeder species' (Whelan *et al.* 2002).

Some obligate seeding species store their seed within the plant canopy and are (typically) readily killed by fire – including low-intensity fire – but regenerate after fire from seed released during and after the fire. If such plants experience a fire soon after the previous one, and the new seedlings have not had a chance to flower and produce seed of their own, then the species may go extinct locally (provided that there is no seed stored in the ground and no dispersal of seed from outside the burnt area).

Alternatively, in the absence of fire for decades (or longer) some plant populations may die of old age. If there is no opportunity to establish seedlings, as may be provided by fire, then the species is lost locally. Species that are sensitive to too short an interval between fires may also be sensitive to too long an interval.

With respect to fauna, some species are highly suited to recently burnt areas, thriving in the relatively simple and open habitats. Others, however, require habitats that are more complex and which may take decades to develop after fire (Quinn 1994; Whelan *et al.* 2002).

Individual fires are heterogeneous – even in large, intense fire events such as those in Victoria and New South Wales in 2001–2002, 2003 and 2006–2007. Even unburnt or lightly burnt patches remain within the fire perimeter. Fire severity maps developed after these fires (e.g. Hammill and Bradstock 2006) or ground-based assessments of variation in severity (e.g. Williams *et al.* 2006) show this, and such heterogeneity is a fundamental feature of large fires elsewhere in the world (M.G.Turner *et al.* 2003; Bradstock 2008).
This heterogeneity of fire can be important for the persistence of some, but not all, plant and animal species, again demonstrating that not all plants and animals respond in the same way to the same fire event.

Fire heterogeneity in rainforest and wet sclerophyll forest, Gibraltar Range National Park/ Washpool National Park, NSW, a part of the Gondwanna Rainforest World Heritage Area.

Source: Peter Clarke

Fire regimes also vary across landscapes. Plant and animal species in fire-prone environments have adaptive fire response traits that reflect the selective force of variation in past fire regimes. Hence there is no 'optimal' fire regime because of the changing nature of the environment. In a management context this means that no single fire regime can be rigorously applied to achieve the maintenance of biodiversity. It also follows that species are not 'fire-adapted' or 'adapted to fire' as such, but are adapted to particular fire regimes and their components. Examples are given in Bradstock *et al.* (2002) of the need for diverse approaches to the implementation of fire regimes in the management of conservation reserves as recognised in many areas of Australia.

This sketch of the fundamental features of fire–biodiversity interactions has highlighted the importance of fire intensity, the interval between fires and the season in which it occurs. These are the components of the fire regime as defined by Gill (1975, 1981) and Bradstock *et al.* (2002). The effects of a single fire are pertinent to populations of plants and animals, and to the physical structure of an ecosystem. But the effects of recurrent fires on species as entities in themselves, or as habitat elements or 'fuel species' (a species that contributes in a major way to the fuel array; Gill 1999), depend on the fire regime.

## 3.3 Current fire management frameworks for biodiversity

### 3.3.1 Fire management for biodiversity conservation

Management of fire regimes is currently recognised as an important part of land management across all national and state jurisdictions, and occurs in a number of contexts, including biodiversity conservation. An understanding of the interactions between fire regimes and functional groups, together with critical aspects of the timing of life events, is critical to the management of fires for biodiversity. Individual plants and animals may die during an individual fire, but the critical issue is whether or not the species is capable of persisting.

In current Australian landscapes the fire regime consists of both prescribed fires and unplanned fires. These may contribute in different ways to the fire regime. A prime consideration is the measurement of outcomes in relation to the management objectives specified for particular locations.

From the point of view of conservation management, the potentially rapid change in fire regimes due to climate change and other constraints imposed by anthropogenic activity may not allow species to respond to the selective force of fire and persist within the borders of protected areas. This constraint on persistence will be compounded by changing fire regimes. There are many contemporary examples in the fire-prone vegetation of south-eastern Australia where local extinction of species has increased due to changed fire regimes, or where the boundaries between communities such as rainforest and sclerophyll forest, or mulga and spinifex, may be changing due to fire regime shifts. There is an ongoing debate about the role of fire regimes and their management in the contemporary Australian setting in determining the range of Australia's major vegetation formations (Bowman 2000).

### 3.3.2 Codes of practice and management systems

Each inquiry into a serious bushfire event draws attention to a perceived conflict in the nature of landscape-level land management that is required to achieve effective protection of human lives and property on the one hand, and to allow maintenance of biodiversity on the other. At the core of the conflict is the rudimentary state of our knowledge of the relationship between fire regime and biodiversity, and our inability to predict, with any precision at a particular location, the consequences of a changing fire regime for any element of the biota.

The most common response to these issues – namely: (i) the uncertainty of knowledge about impacts on particular species; and (ii) the conflict in achieving property protection and biodiversity conservation – has been twofold. First, identification of fire management zonings at a landscape level allows identification of the primary management objectives of land in various parts of the landscape in relation to the protection of various types of assets (e.g. Asset Protection Zone, Flora and Fauna Management Zone, Fire Exclusion Zone, Strategic Fuel-Reduced Corridor Zone; Esplin *et al.* 2003; Ellis *et al.* 2005). Second, estimates can be made on the basis of the best current knowledge of the perceived vulnerability of particular groups of the biota to fire regimes (e.g. Gill and Nicholls 1989). Vulnerability of plant species may be discerned from their functional groups including immediate response to fire (resprout or not) and time of regeneration (interfire, post-fire) (e.g. Noble and Slatyer 1980), and life-history markers (e.g. time to seed production from seedlings – primary juvenile period – or resprouts – secondary juvenile period (Gill 1975); and longevity).

Management systems in a number of Australian states and territories are developing rapidly; they use functional groups of plants to set thresholds – especially for desired minimum and maximum inter-fire intervals – and some form of variation in these intervals can be based on statistical analysis of data or, more often, on expert opinion. Assessing the thresholds for vulnerability of animals is less well developed, but can be related to habitat associations and life-history characteristics (Whelan *et al.* 2002; Tasker *et al.* 2006 ; Cawson and Muir 2008). This approach is already being applied to the development of management systems that accommodate thresholds for animal species. Generic elements of systems used in Australia are summarised in Gill (2008).

## 3.4 Current and future challenges for managing fire for biodiversity

In general, the models of responses of species and ecosystems to climate change – and proposed ways of managing landscapes to adapt to climate change – do not consider the potential impacts of climate change on fire regimes. Interactions between climate change and fire regimes are complex and poorly understood. Whilst there is a real risk of unwanted biodiversity outcomes as a consequence of new and inappropriate fire regimes, the direction and rates by taxa are highly uncertain and difficult to predict. Thus a key, ongoing challenge for biodiversity management will be to evaluate the frameworks and management practices that are currently used to manage for biodiversity outcomes – including those designed to manage for climate change impacts – in the light of changing patterns of risk to biodiversity as a consequence of changes to both climatic and disturbance regimes.

# 4. CLIMATE CHANGE AND FIRE REGIMES

## 4.1 Past climates and fire regimes

### 4.1.1 Climate and fire prior to human presence

Lynch *et al.* (2007) provided a comprehensive review on past climate and fire, and Dunlop and Brown (2008, pp. 22–25) provided an overview of past climate and its role in shaping the nature and distribution of the Australian biota. This section also draws heavily from the recent review of pre-European fire regimes in Australian ecosystems by Enright and Thomas (2008).

That fire was important in pre-human Australia is evident in the range of responses to fire observed in the flora of extant fire-prone plant communities, with many species able to regrow vegetatively after fire from protected buds (either above or below ground), seeds stored in protective woody fruits in a canopy seed bank (serotiny), fire-stimulated flowering, and heat and smoke-stimulated germination of soil-stored seeds (Gill 1981).

Prominent Australian plant families, including the Proteaceae and Myrtaceae, were major components of (primarily tropical) Australian Eocene flora, with sclerophylly most likely a response to low-fertility soils rather than to seasonality of rainfall and aridity (Kershaw *et al.* 2002). A shift towards drier conditions across Australia is associated with the rifting of the continent from Antarctica and establishment of a circumpolar ocean current around 40 million years b.p. The Earth continued to cool and dry through the Oligocene (40–25 million years b.p.), glaciers formed at the South Pole, and fire-prone biomes with scleromorphous elements in the flora increased (Kershaw *et al.* 2002).

Evidence for fire appears unequivocally towards the end of the Miocene (about 10–6 million years b.p.) as fossil charcoal in south-eastern Australian deposits, and since at least 1.4 million years b.p. as abundant charcoal particles in deep-sea core ODP-820 off the coast of north Queensland, which is indicative of the regional presence of recurrent fire (Kershaw *et al.* 1993, 2002). Pollen and charcoal analyses of two small sections of varved sediments from a Pliocene-age deposit (estimated 3.2 million years b.p.) near Yallalie in south-western Australia, revealed the occurrence of fires in vegetation interpreted to represent a mosaic of dry rainforest (including Araucariaceae) and sclerophyll woodland dominated by Casuarinaceae and Myrtaceae (Atahan *et al.* 2004; Dodson *et al.* 2005). Dodson *et al.* (2005) concluded that although fire was common, fire intervals were longer – and perhaps more variable – than today. Singh and Geissler (1985) reported recurrent fire back to 800,000 years b.p. at Lake George in south-eastern Australia, increasing especially after 140,000 years b.p. – probably in association with intensification of ENSO climate variability, which shows peaks around 130,000 years b.p. and 40,000 years b.p. (Kershaw *et al.* 2003).

Fire has increased as a factor driving evolution of the Australian biota since the onset of the glacial–interglacial cycles of the past two million years. Glacials are generally drier, with more open vegetation dominated by Poaceae, Asteraceae and Chenopodiaceae, and charcoal abundance (evidence of fire) markedly reduced in much of temperate Australia where increased aridity limited fuel availability (Hope *et al.* 2004). On the other hand, in parts of tropical Australia, charcoal levels are maintained, or increase, as fire-prone savanna vegetation replaces areas of wet forest and the effectiveness of the summer monsoon decreases. A range of global and regional insolation factors are correlated with charcoal fragment abundance in long ocean core sequences from northern Australia and south-east Asia, suggesting real climate-driven variations in the occurrence of fire at time scales of thousands and tens of thousands of years.

### 4.1.2 Climate, people and fire

Based on a recent re-evaluation of radiocarbon dates and measurement uncertainties, Gillespie *et al.* (2006) contended that the most likely date for first settlement of Australia by people was 50,000–46,000 years b.p. The first Australians were almost certainly already skilled users of fire, and Jones (1969) proposed the term 'fire stick farming' to describe their purposeful burning and manipulation of plant communities.
To what extent subsequent changes to the biota were wrought by people, as opposed to climate, is difficult to disentangle. At the time of Aboriginal arrival, Australia supported a diverse megafauna of marsupial species with body weights >40 kg, but within 1000–6000 years (i.e. by 45,000–40,000 years b.p.) most

megafauna were extinct (Gillespie *et al.* 2006). Miller *et al.* (2005) identified a sudden shift in vegetation for arid woodlands of central Australia, based on 13C analysis of egg shells from the emu (*Dromaeus novaehollandiae*) and the extinct flightless bird *Genyornis newtoni*, with nutrient-rich C4 grasses replaced by fire-adapted chenopod desert scrub and low-nutrient grasses around 50,000–45,000 years b.p. Pollen and charcoal evidence from marine core GC-17 near North West Cape (van der Kaars and De Deckker 2002) supports this rapid vegetation change as a result of increased fire. However, it is followed by a sustained period of lower charcoal inputs after 40,000 years b.p., suggesting that people may not be implicated. Rather, they suggested that increased aridity, and reduction in biomass to fuel fires, may have occurred due to a weakening of the monsoon at this time – so that apparently subtle changes in climate may have had profound consequences for both fauna and flora. In north-eastern Australia, rainforest decreased in extent rapidly only after about 40,000 years b.p., being replaced by fire-prone sclerophyll (*Eucalyptus*-dominated) vegetation. The timing of rainforest decline coincided with a phase of inferred high ENSO activity (Turney *et al.* 2004) and with the presence of people. Charcoal peaks with a periodicity of around 1500 years b.p. coincided with periods of rainforest decline over the following 20,000 years. Microcharcoal levels decreased around 7,000 years b.p. as temperature and rainfall increased, and rainforest re-invaded sclerophyll forest reclaiming lost areas of habitat, regardless of the presence of humans as an additional ignition source (Kershaw 1986; Lynch *et al.* 2007). Charcoal levels increased again after 4,000 years b.p., which also correlated with increased ENSO-induced climatic variability (Lynch *et al.* 2007). In central and southern Australia the decline of *Callitris* (Cupressaceae) woodlands around Lake Frome in the period 13,000–11,000 years b.p. and Lake Eyre in the period 10,000–5,000 years b.p. corresponded with greater variability in rainfall and with increased human presence (Singh and Luly 1991; Luly 2001). Many of the rapid changes in vegetation reported in the fossil record for the period of human occupation of Australia are correlated with both increased charcoal (more fire), and with global and regional insolation factors, as noted also for the pre-human past (Kershaw *et al.* 2003; Hope *et al.* 2004; Lynch *et al.* 2007), and highlight the potentially important role of climate-driven processes operating on a variety of timescales.

### 4.1.3 Human impacts on pre-European fire regimes

Bowman (2003) proposed a simple three-step model to describe the changing role of fire through time in relation to savanna ecosystem dynamics, which may be generalised to much of the continent: (1) a pre-human period of lightning-driven fire resulting in fires of moderate to large size, producing a coarse scale habitat mosaic to which the biota was well adapted; (2) the Indigenous peoples (pre-European) period of frequent fire, leading to small fires and a fine-scale habitat mosaic (which favoured small animals, but perhaps not the Pleistocene megafauna); and (3) the post-European period of very large fires, which may lead to homogenisation of the landscape, and consequent loss of habitat complexity and declines in the abundance of some mammal and bird species.

In southern Australia, early European explorer and settler accounts describe the frequent occurrence of smoke, and burning of vegetation by Aborigines for a variety of purposes – to maintain open-forest understoreys for ease of travel, to produce new growth of plants for human consumption (or for consumption by animals used by people) and to drive animals for hunting purposes (see Abbott 2003 and references therein). Nevertheless, tree-ring studies suggested that large, high-severity fires still occurred from time to time (Luke and McArthur 1978; Burrows *et al.* 1995; Mackey *et al.* 2002; Mooney 2004; Bradstock 2008). A synthesis of historical records for fires in south-western Australia before and during early European settlement reported that most fires were lit in summer, and nearly all occurred in the hottest four months of the year under typically hot and windy conditions (Abbott 2003). In addition, some forests with an open and grassy understorey were most likely natural – as were some with dense, shrubby understoreys – while many shrublands may have been largely unmanaged, reflecting their low resource quality relative to other parts of the landscape. According to Hassell and Dodson (2003), there was a strong correlation between resource richness (availability of fresh water and moderate to high soil fertility), Aboriginal population sizes at the time of European settlement and the frequency of vegetation treatment by fire.

Aboriginal people patch-burning spinifex fuels, Walalakara Indigenous Protected Area, South Australia.
Source: Department of the Environment, Water, Heritage and the Arts

In northern Australia, the muted response by savanna plant communities to fire suggests a high degree of resilience to impacts on biodiversity from customary burning by Aborigines over thousands of years (Andersen *et al.* 2003, 2005). Compared to the pre-human fire regime, pre-European habitation period fires were probably more frequent and smaller, while evidence for shifts in season of fire are somewhat equivocal. Rather than a shift in season, it is possible that the fire season was broadened, with fires lit early, mid and late dry season in different habitats and for different reasons (Bowman 1998). The high frequency of lightning, and natural patterns of build-up of grassy fuel loads, suggests that the addition of a human ignition source may not have greatly altered the structure and composition of savanna ecosystems. However, evidence for possible tree recruitment failure in response to post-European increases in fire frequency in some areas, and monsoon forest expansion in others, shows that the tree–grass balance is sensitive to fire regime change under current climate and may have shifted in response to pre-European fire regime changes.

### Summary

Climate change over millions of years (since at least the mid-Tertiary) towards a drier, more seasonal climate has been accompanied by major changes in the nature of Australian ecosystems and a massive increase in the occurrence of fire as a disturbance factor. By the time of first human settlement around 50,000–45,000 years b.p. the basic range and distribution of vegetation types encountered by European settlers was largely in place. The specific impacts of climate change and people on biota over the last glacial–interglacial cycle are difficult to disentangle. People may be implicated in the loss of megafaunal biodiversity, but there is no evidence for negative impacts on plant biodiversity. Nor does the range of climate changes encountered over the past 50,000 years (or more) guide us greatly in terms of the potential impacts of future climate change on biota, since we are entering uncharted territory: temperature and $CO_2$ are predicted to rise by the end of this century to levels higher than any in the past 5 million years, and faster (by 1–2 orders of magnitude) than recorded in palaeo-environmental reconstructions of the glacial–interglacial cycles – the speed of this change approaching that of an 'event' in context of the palaeo-record. Much of the country is therefore entering a period of novel combinations of hotter and drier conditions, as opposed to the more usual warm–wet (interglacial periods) and cool–dry (glacial) combinations that have occurred in the past.

## 4.2 Climate change and fire-weather

### 4.2.1 Fire-weather and fire danger indices

Fire behaviour depends very strongly on fire-weather which, in turn, will be affected by climate change. Fire-weather is measured on the basis of rainfall, temperature, humidity, wind speed and the curing of grasses. How these variables affect the forward rates of spread of a fire burning with the wind on level ground has been the measurable outcome of fire danger rating (FDR) systems for forest (McArthur 1967) and grassland (McArthur 1966) in Australia. McArthur defined these indices to assist foresters in relating the weather to the expected fire behaviour in the appropriate fuel type. McArthur (1967) argued that FDR is also directly related to the chances of a fire starting, its intensity and the difficulty of its suppression. There are five categories in the FDR system: low (0–5), moderate (5–12), high (12–25), very high (25–50) and extreme (50+).

The variables used in FDR systems in Australia directly or indirectly affect rates of spread of fires. In both grassland and forest situations wind acts directly, whereas moisture of the forest floor litter or grassy fuels is estimated implicitly from a knowledge of the proportions of live and dead fuels (grass), from combinations of temperature and relative humidity (for quick response, dead fuels in forest or grassland), or from measures of longer-term (soil dryness) or medium- to short-term drought (drought factor, capped at 10 in the McArthur system) in forests.

As discussed in section 2.1, Australia has become warmer, with less rainfall in the south and east, more rainfall in the north-west, and changes in the number of extreme weather events. These climatic trends are highly relevant to understanding changes to fire-weather – both in terms of trends in the recent past, and projections for coming decades. The application of these indices to evaluating the impact of climate change on fire-weather is discussed further in the following two sections. Section 4.2.2 discusses recent trends in fire-weather and section 4.2.3 discusses the impacts of future climate projections on fire-weather.

### 4.2.2 Climate change and fire-weather: recent trends

*Recent trends in climate*

Australian average temperatures have increased 0.9°C since 1910, of which 0.8°C has occurred since 1950, with greatest warming in the east and least warming in the north-west (Figures 4.1, 4.2). The Australian average warmest year on record was 2005, while the warmest year for southern Australia was 2007. The number of hot days and nights has increased, and the number of cold days and nights has declined.

Since 1950, most of eastern and south-western Australia has become drier (Fig. 4.2). Across New South Wales and Queensland these rainfall trends partly reflect a very wet period around the 1950s, though recent years have been unusually dry. In contrast, north-western Australia has become wetter over this period, mostly during summer. Since 1950, the frequency of both the occurrence of days of very heavy rainfall (over 30 mm/day) and the number of wet days (at least 1 mm/day) have decreased in the south and east but increased in the north (Fig. 4.3).

Australian rainfall shows considerable variability from year-to-year, partly in association with the ENSO. El Niño events tend to be hot and dry years in Australia, and La Niña events tend to be cool and wet years. There has been an increase in the frequency of El Niño events since the mid-1970s (Power and Smith 2007). Notably dry years such as 1972–1973, 1977–1978, 1982–1983, 1987–1988, 1991–1993, 1994–1995, 1997–1998 and 2002–2003 all coincided with major El Niño events. Wet years such as 1973–1974, 1998–1999, 2000–2001 and 2007–2008 coincided with La Niña events.

Exceptions to this strong ENSO-linked interannual variability are extended periods of above or below average rainfall, such as:

- the Federation Drought, from the mid-1890s through to 1902
- the lowest rainfall decade on record during most of the 1940s (in most of south-eastern Australia)
- the very wet period in the early 1970s
- the last 5–10 years, which was one of the most severe droughts in Australia's history.

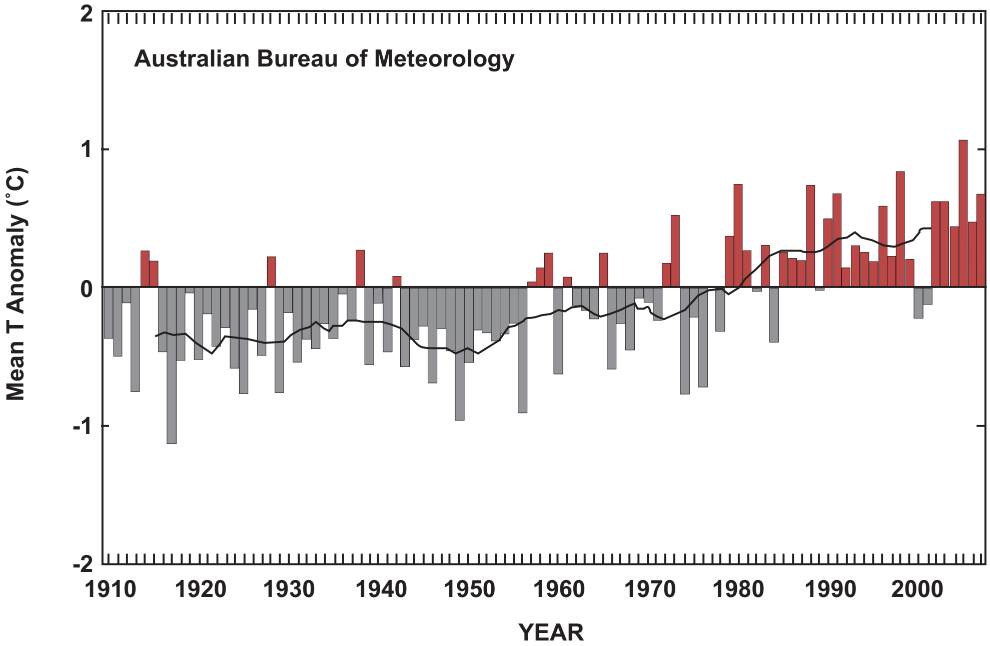



**Fig. 4.1. Australian average annual temperature anomalies relative to 1961–1990.**
Source:  Australian Bureau of Meteorology

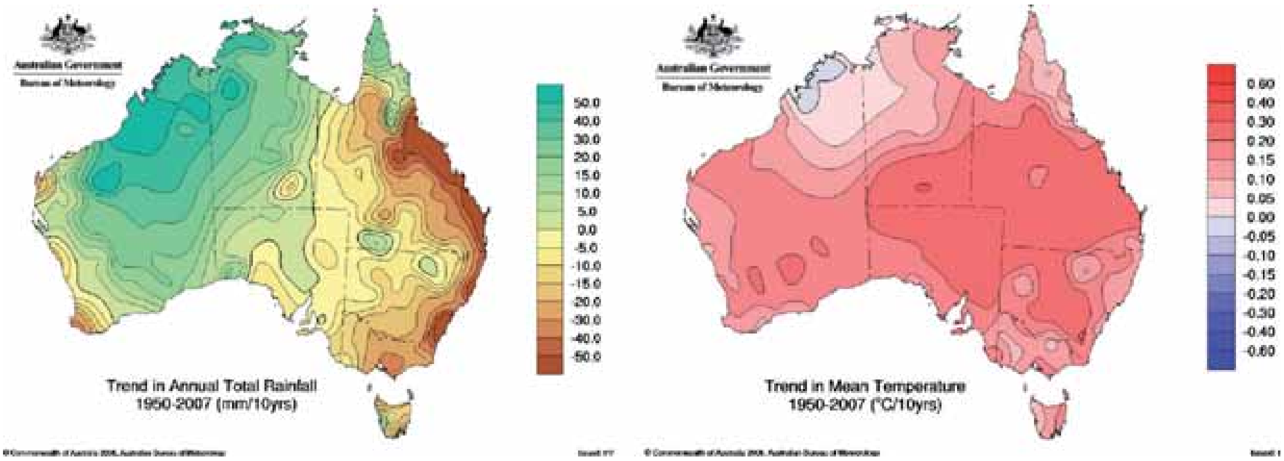



**Fig. 4.2. Trends in annual mean Australian rainfall (left map) and temperature (right map) since 1950.**
Source: Australian Bureau of Meteorology

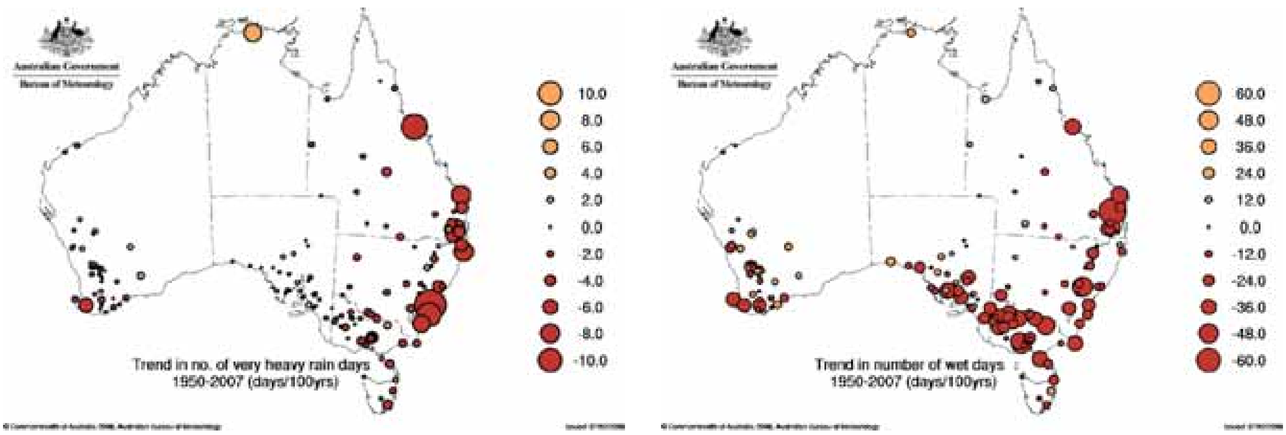



**Fig. 4.3. Trends in very heavy rainfall (over 30 mm/day) and wet days (at least 1 mm/day) since 1950.**
Source: Australian Bureau of Meteorology

**Fig. 4.4. Map showing locations of climate stations used by Lucas *et al.* (2007).** Circles represent stations with data extending to 1973. Stars represent stations where longer time-series are available. Source: Lucas *et al.* (2007)

*Recent trends in variables affecting fire-weather – data sources and data quality*

The tendency for hotter and drier conditions in southern and eastern Australia is likely to have affected fire-weather. This has been investigated by Lucas *et al.* (2007). They identified 26 Bureau of Meteorology observing stations in south-eastern Australia that had daily records of the primary weather variables affecting fire-weather – temperature, relative humidity, wind speed and rainfall – from 1973 to 2007 (Fig. 4.4).

When examining meteorological and climatic data acquired over decades, it is important to consider the homogeneity of the data. Homogeneous data are those that are free from artificial trends and/or discontinuities. These discontinuities can arise from factors such as moving the observing station, changes in instrumentation and/or changes in the observational practices. The data used in this study are generally not homogeneous. Although homogenised high-quality databases of maximum temperature, humidity and rainfall do exist at many of these stations, none of these databases has been updated to include the most recent observations. At the vast majority of the stations, a major change in the record is the introduction of the automatic weather station. This generally occurs sometime in the 1990s, and was accompanied by changes in the instrumentation and, often, a site move. Particularly relevant with this change was the change in anemometer instrumentation for measuring wind. See Lucas *et al.* (2007) for a more detailed discussion.

*Trends in the cumulative Forest Fire Danger Index ΣFFDI*

In most Australian states, fire-weather risk is quantified using one of two indices: the Forest Fire Danger Index (FFDI) or the Grassland Fire Danger Index (GFDI) (Luke and McArthur 1978; see also section 4.2.1). While the details of each calculation are different, the basic ingredients are the same. Observations of temperature, relative humidity and wind speed are combined with an estimate of the fuel state to predict fire behaviour. For forests, the fuel state is determined by the so-called 'drought factor', which depends on the daily rainfall and the period of time elapsed since the last rain. The drought factor is meant to encapsulate the effects of both slowly varying long-term rainfall deficits (or excesses) and short-term wetting of fine fuels from recent rain (Finkele *et al.* 2006). For grassland, the fuel state is determined by the 'curing factor', which is the dryness of grassland from visual estimates expressed as a percentage.

Initially, these quantities were estimated using a mechanical nomogram in the form of a set of cardboard wheels (see Luke and McArthur 1978, pp. 113–118), where the user 'dialled in' the observations to compute the FFDI. Such meters are still used operationally. Noble *et al.* (1980) and Purton (1982) reverse-engineered the meter for FFDI to derive equations suitable for use on computers:

$$FFDI = 1.2753 \times \exp(0.987 \log DF + 0.0338T + 0.0234V - 0.0345RH)$$

where DF is the drought factor, T the air temperature in Celsius, V the wind speed in km/h and RH the relative humidity expressed in percent. The drought factor is calculated using the Griffiths (1999) formulation and uses the Keetch-Byram Drought Index (Keetch and Byram 1968) to estimate the soil moisture deficit.

The variables used to estimate the changes in fire-weather include: (1) the annual cumulative FFDI (ΣFFDI), which is the summation of the daily FFDI values over an entire year (here, a year is defined from July through June to better encompass a continuous fire season in south-eastern Australia than the calendar year); and (2) the number of days within each FDR category; the number of days in either the 'very high' or 'extreme' categories are denoted 'VHE days'.

### *Trends in cumulative annual fire danger in south-eastern Australia*

In Lucas *et al.* (2007), linear trends of ΣFFDI over the 1970–2007 period were computed for stations in south-eastern Australia. Most stations show a significant positive trend, with a few exceptions near the coast and at Canberra. The largest trends are found in the interior of New South Wales and other inland areas.

Individual time-series plots of ΣFFDI reveal some interesting trends. Rather than being a smooth, continuous increase, the ΣFFDI displays a jump in the late 1990s to the early 2000s at many locations (Fig. 4.5). The strongest jumps are seen in the interior portions of New South Wales. Table 4.1 shows the average annual ΣFFDI from 1980 to 2000 and from 2001 to 2007, along with the percentage change between these two periods. This separation point is somewhat arbitrary, and was based on a visual examination of the plots. The start date for the first period (1980) was chosen to eliminate the particularly low values observed in the early to mid-1970s, when a strong La Niña brought an extended period of abnormally high precipitation and low fire danger to much of the region.

Increases of 10–40% in the 2001–2007 period, relative to the 1980–2000 period, are evident at most sites. The changes observed thus far in the 21st century are equal to or exceed the changes predicted to occur by 2050 in the different modelling scenarios. This tendency is particularly pronounced in interior New South Wales. The changes in ΣFFDI at these stations are associated with an increase in the number of VHE days.

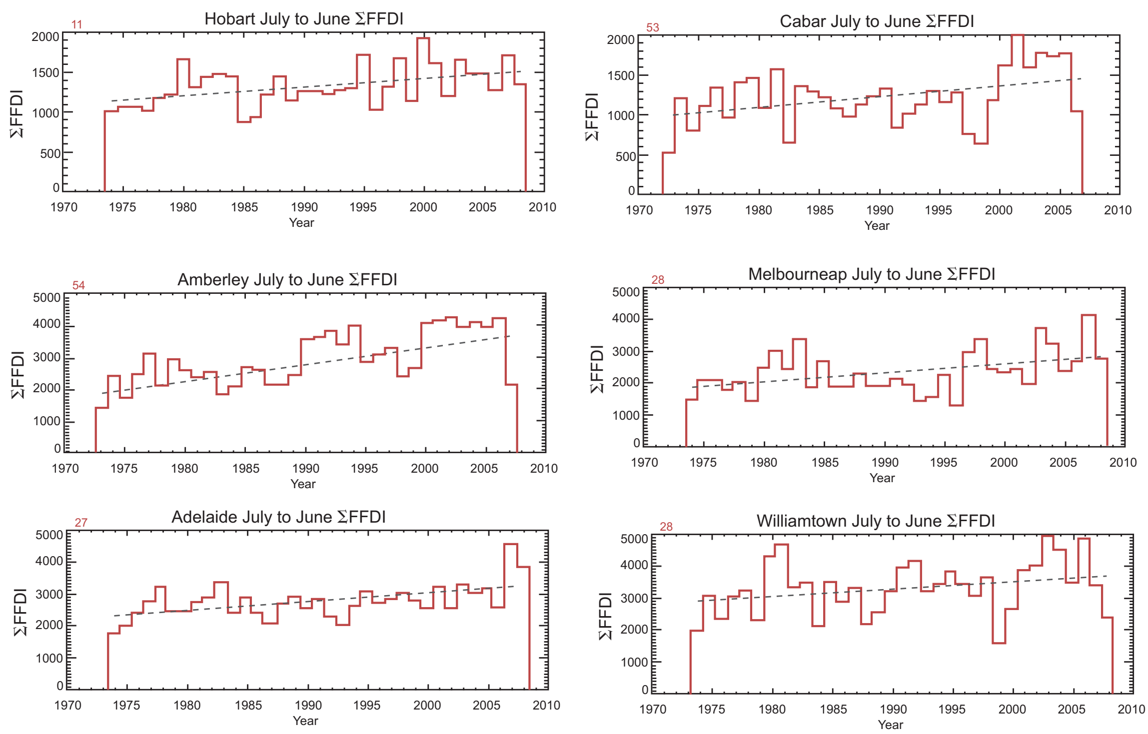



**Fig. 4.5. Time-series plots of ΣFFDI at six locations in south-eastern Australia.** Clockwise from upper left: Hobart, Cobar, Melbourne airport, Williamtown, Adelaide and Amberley. Linear regression line is shown in dashed grey. The trend, in points per year, is given by the red number at the top left of each figure. Source: Lucas *et al.* (2007)

**Table 4.1. Average oFFDI from 1980 to 2000 and from 2001 to 2007.** The percentage change between the two is also shown. Source: Lucas et al. (2007)

| Site | 1980–2000 ΣFFDI | 2001-07 ΣFFDI | % change | Site | 1980–2000 ΣFFDI | 2001–2007 ΣFFDI | % change |
|---|---|---|---|---|---|---|---|
| Adelaide | 2671 | 3051 | 14 | Laverton | 2065 | 2268 | 9 |
| Amberley | 2805 | 3885 | 38 | Melbourne airport | 2274 | 2805 | 23 |
| Bendigo | 2400 | 3439 | 43 | Mildura | 5095 | 5898 | 15 |
| Bourke | 4346 | 7375 | 69 | Mt Gambier | 1932 | 2004 | 3 |
| Brisbane airport | 1970 | 2139 | 8 | Moree | 3753 | 5159 | 37 |
| Canberra | 2484 | 2925 | 17 | Nowra | 1718 | 2242 | 30 |
| Ceduna | 4444 | 5114 | 15 | Richmond | 2452 | 3099 | 26 |
| Charleville | 6215 | 7065 | 13 | Rockhampton | 3125 | 3878 | 24 |
| Cobar | 4519 | 6388 | 41 | Sale | 1639 | 2175 | 32 |
| Coffs Harbour | 1205 | 1490 | 23 | Sydney airport | 1812 | 2475 | 36 |
| Dubbo | 2914 | 4662 | 59 | Wagga Wagga | 3130 | 4451 | 42 |
| Hobart | 1339 | 1424 | 6 | Williamtown | 1950 | 2425 | 24 |
| Launceston airport | 1397 | 1488 | 6 | Woomera | 7478 | 8244 | 10 |

Data heterogenities cannot be absolutely excluded as the source of this apparent jump in fire danger.

However, several factors suggest that this is a real phenomenon. As noted earlier, a statistical analysis of the wind uncertainties suggests that the medians of the distributions are not seriously affected by the errors. Hence, the medians should be relatively homogeneous and computed trends should be realistic. Another factor is the timing and spatial coherence of the jump. In interior New South Wales, the large jump begins in 2001–2002 across a wide area. It seems unlikely that all heterogeneities would occur simultaneously across a broad region of the country.

### *Trends in cumulative annual fire danger in the rest of Australia*

Comprehensive analyses of projected impacts of climate change on fire-weather, and of the recent trends in FFDI, have only been undertaken for south-eastern Australia. Nonetheless, a few plots from elsewhere in Australia are included below to provide at least a brief look at the situation (Fig. 4.6).

Positive trends are apparent at most sites, except for Darwin. Since the mid-1970s, the upward trends are generally of the same magnitude as those noted in the south-east. The late 1990s 'jump' is not apparent at most of these sites, although there are some suggestions of it at Cairns.

### *The longer view*

The use of 34-year data series (1973 to early 2007) to estimate trends may not give a true representation of the variability in the data. At eight stations in the south-east, longer records exist that can be used to test the robustness of these trends. Representative, long time-series plots of ΣFFDI, extending back to the early 1940s at Melbourne airport and Canberra, are shown in Fig. 4.7.

In general, the trends are much weaker than seen in the shorter time-series, and in some cases, not significant. The weaker trends in the longer time-series are a result of interdecadal variability, with extended periods of higher and lower ΣFFDI. The five-year running mean helps highlight these periods. Periods of approximately 20 years are particularly noticeable by inspection. Spectral analysis (not shown) also suggests similar results, with low frequencies dominating. There is a peak at the highest (two-year) frequencies, but not as strong. Thus, fire-weather indices are subject to decadal scale variation, and these broad, interdecadal periods of increasing and decreasing ΣFFDI are likely to continue in the coming century.

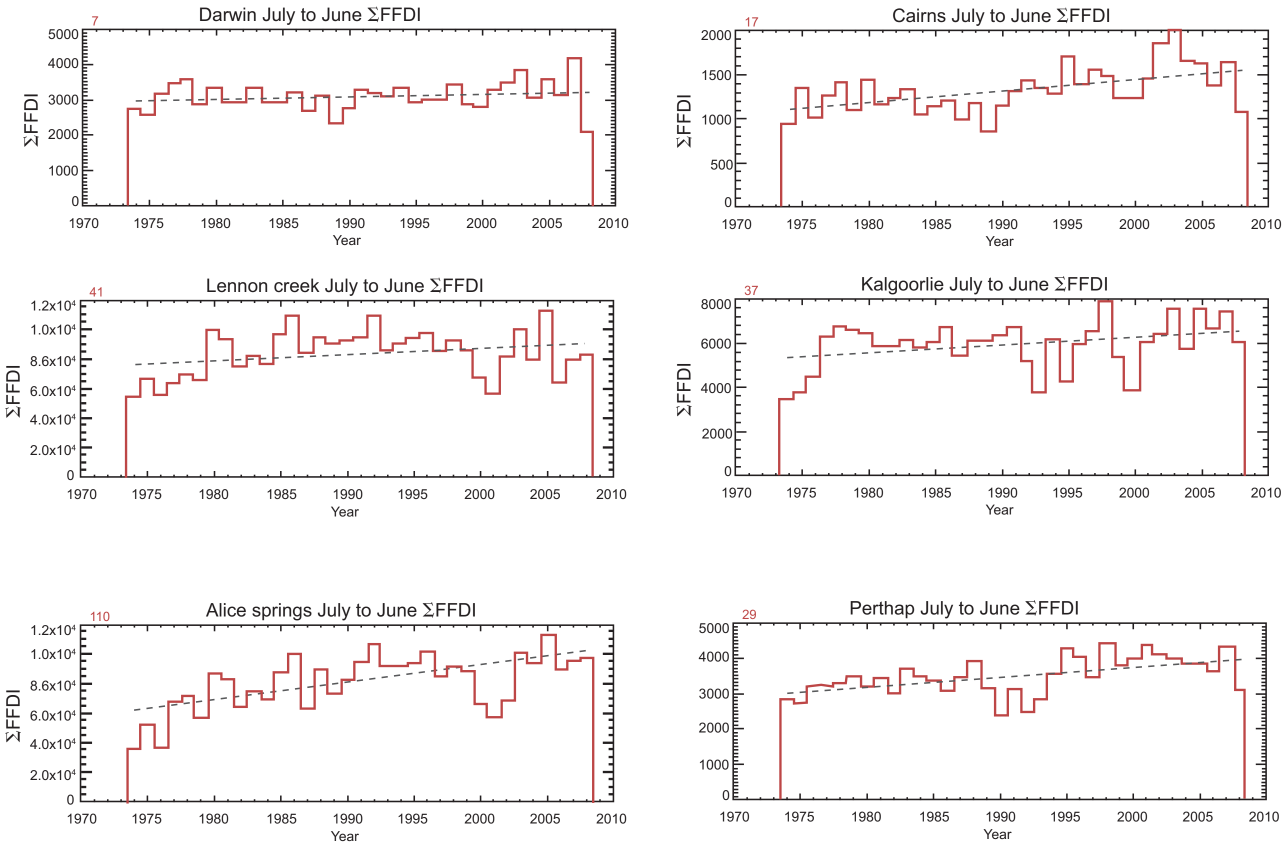



**Fig. 4.6. Time-series plots of ΣFFDI at six locations in regions other than south-eastern Australia.** As in Fig. 4.5, except for (clockwise from upper left) Darwin, Cairns, Kalgoorlie, Perth, Alice Springs and Tennant Creek. Source: Lucas *et al.* (2007)

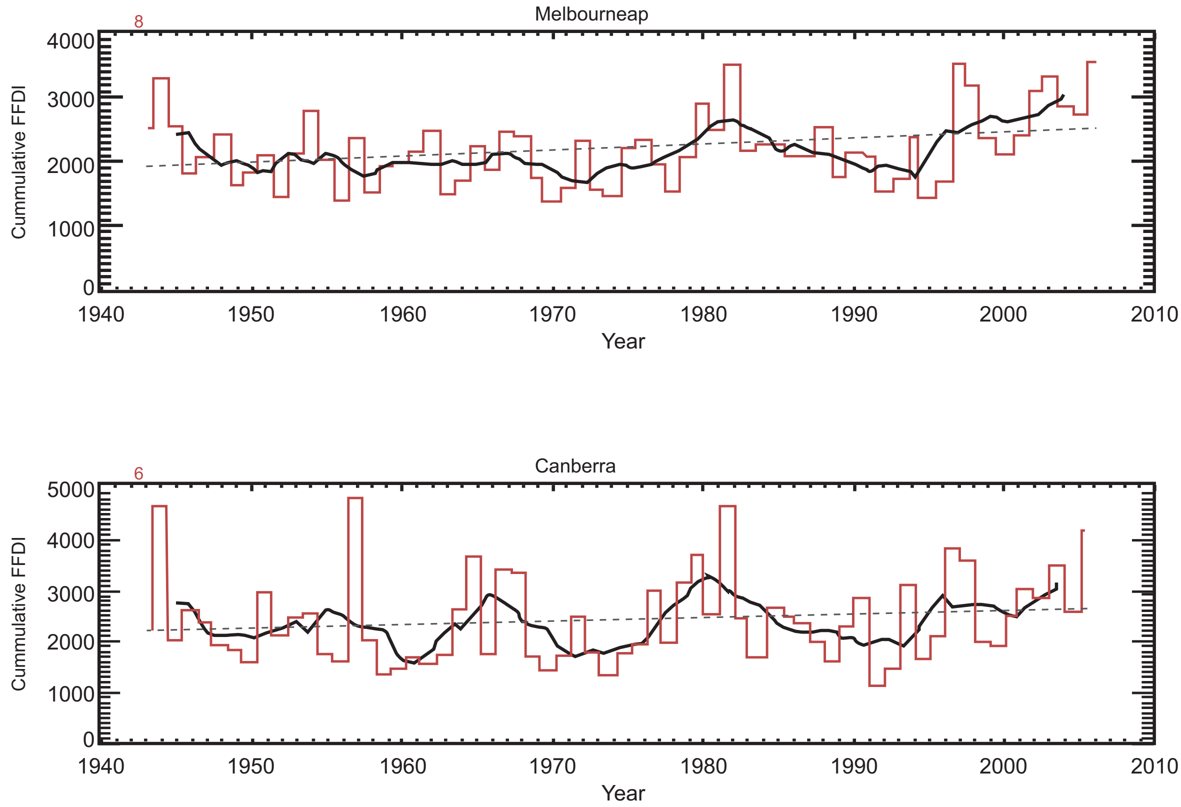



**Fig. 4.7. Long time-series of ΣFFDI at Melbourne airport (ap) and Canberra.** The black line represents a five-year running mean and the grey dashed line is the linear trend. Source: Lucas *et al.* (2007)

### *Other observed changes*

The length and timing of the active fire season has been examined using a simple metric of the time-series extending back to the 1940s. For this purpose, the start (end) of the active fire season is objectively defined as the average date of the first (and last) three occurrences of an FFDI of at least 25 after 1 July of a given year. From these start and end dates, the length of the season can be determined. These dates are computed at Adelaide, Canberra, Wagga Wagga and Melbourne (the methods cannot be applied on the datasets for Hobart and Sydney). Fig. 4.8 shows the season length time-series for the four stations. To highlight the general behaviour, a five-year running mean is also shown. Several features are readily apparent. In most cases, the last few years have been among the longest on record, part of an upward swing since the early 1990s. There is also an apparent decadal variation, with broad peaks in the 1940s, the late 1970s to early 1980s, and in the present. Shorter fire seasons were generally seen in the late 1950s and the 1960s, and in the late 1980s. A general upward trend is suggested, but is not statistically significant. This broad behaviour is similar at each of the other long-period stations.

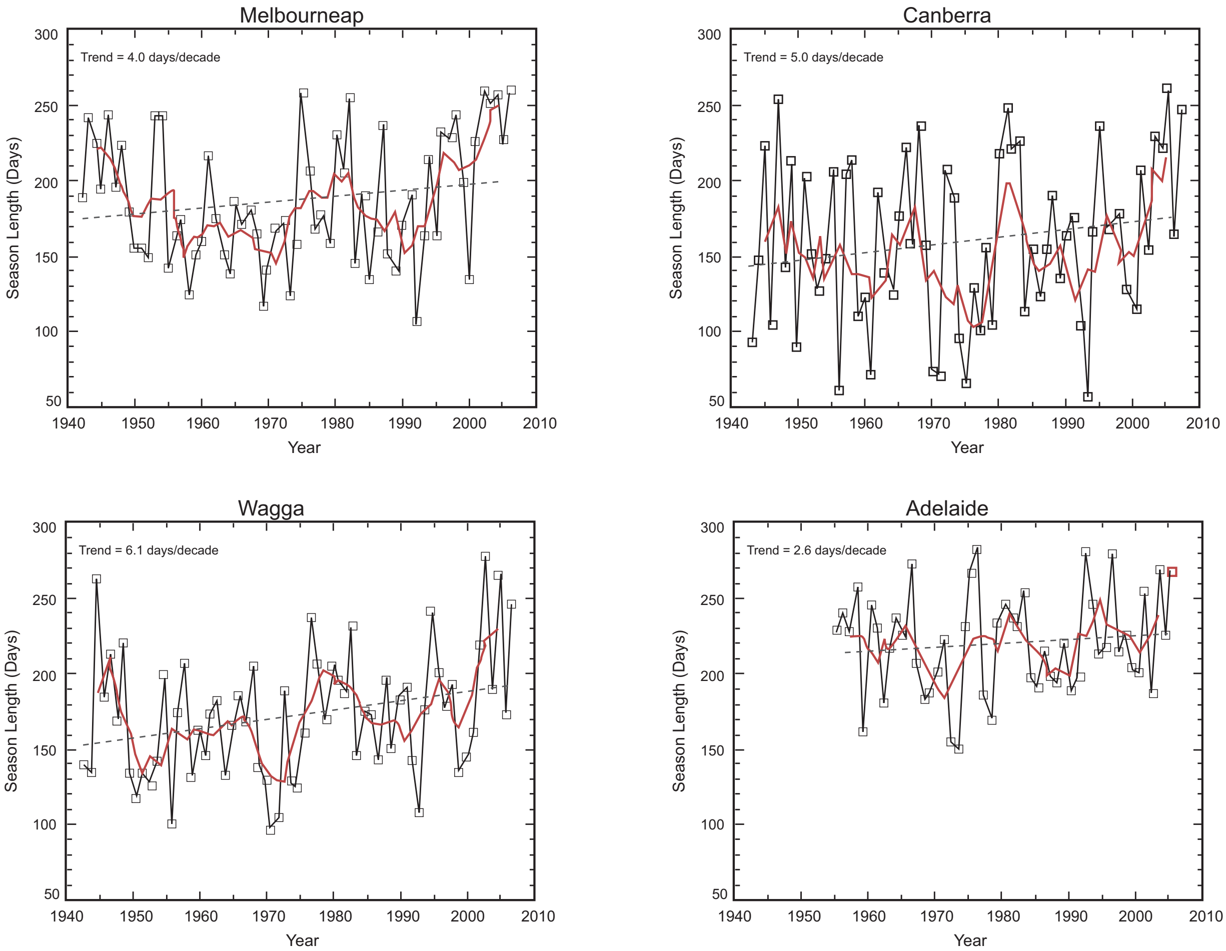



**Fig. 4.8. Estimated season length (days) at Melbourne airport, Canberra, Wagga Wagga and Adelaide.**
The red line represents the five-year running mean and the grey dashed line is the linear trend.
Source: Lucas *et al.* (2007)

Fig. 4.9 shows the start dates of the first and last 'very high' fire danger day (solid and dashed green lines, respectively) and the date of the first 'extreme' days at Melbourne airport. Linear trend lines are fitted to the data. These data clearly suggest that since about 2000, the first incidence of dangerous fire-weather is occurring earlier in the spring. The trends in the start dates, although not significant, are of the order of 3–4 days/decade earlier in the year. Over the last few years, extreme days have been occurring in October. While the earlier ΣFFDI data suggest some strong fire seasons in the 1940s, extreme days in those years did not occur until mid-December. Thus, in south-eastern Australia, the trend towards increasing ΣFFDI is also associated with a trend towards an earlier occurrence in the fire season of days with a fire danger of at least 'very high'.

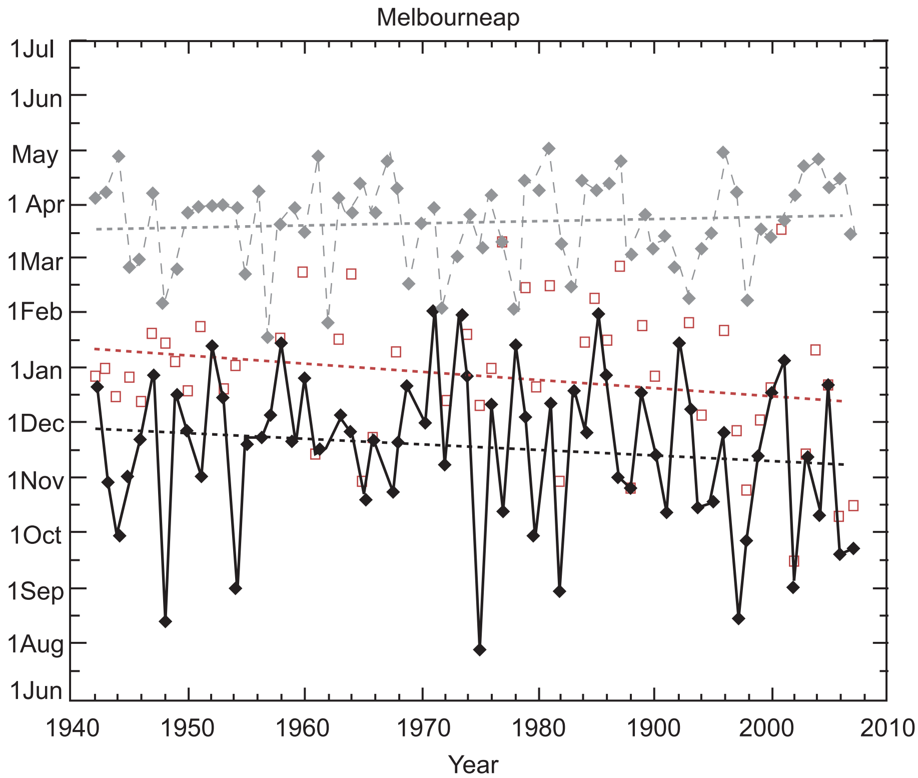



**Fig. 4.9. Time-series of date of first 'very high' day (solid black line), first 'extreme' day (red dots) and last 'very high' day (dashed grey line).** Linear trend lines are also depicted. Source: Lucas *et al*. (2007)

In summary, the longer records clearly show evidence of ongoing interdecadal variation, with recent years showing an apparent jump in fire danger. Careful examination of the records suggests that climate change may be playing a role in this increased FFDI. The available data indicate that the fire seasons we have been experiencing for the last few years have been longer and, in many ways, stronger than any observed dating back to the 1940s. It is not that a given day has a higher FFDI value; rather, there are more very high and extreme fire danger days and fewer low-to-medium fire danger days. A reasonable hypothesis for this behaviour is that we a currently experiencing an upswing in fire danger due to some natural forcing with an interdecadal time scale, and that this is being exacerbated by the subtle, ongoing effects of climate change.

### 4.2.3 Climate change and fire-weather: Projections for 2020 and 2050

Climate change can act in two ways to affect fire-weather. First, it might exacerbate the fire-weather risk of any given day, leading to increased frequency or intensity of 'extreme' and 'very high' fire-weather days. Second, an increase in the accumulated fire risk over a year might represent a longer fire season and a reduction in the number of days suitable for prescribed burning. In this section, we examine the climate change scenarios for the coming decades from both of these standpoints.

*Data sources*

The recent report *Climate change in Australia* (CSIRO and Bureau of Meteorology 2007) described the changes that can be expected to occur in Australia in the coming decades. The results are based on simulations from 23 different climate models and global warming estimates from the Intergovernmental Panel on Climate Change (IPCC 2007). The most important changes for fire-weather occur in temperature and rainfall. The 'median' (most likely) projections for 2050 are shown in Fig. 4.10. The 'matrix' in each plot represents the results over different seasons and across three different emissions scenarios (low, medium and high). In general, significantly higher average temperatures and decreasing rainfall are expected over most of Australia by 2050. These changes are more extreme with higher emissions, and the world is currently following, even exceeding, a high emissions path (Rahmstorf *et al.* 2007). Careful examination of the charts also suggests that the drying will be largest in winter and spring, while the warming will be largest in spring and summer.

To simulate future changes in fire-weather, daily data are needed. This limited the analysis to two CSIRO climate simulations named CCAM (Mark 2) and CCAM (Mark 3). Projected changes in daily temperature, humidity, wind and rainfall were generated for the years 2020 and 2050, relative to 1990 (the reference year used by the IPCC). These projections include changes in daily variability, expressed as a pattern of change per degree of global warming.

The patterns were scaled for the years 2020 and 2050 using IPCC (2007) estimates of global warming for those years, i.e. 0.4–1.0°C by 2020 and 0.7–2.9°C by 2050. This allows for the full range of IPCC scenarios of greenhouse gas and aerosol emissions.

The modelled changes from the various scenarios are then projected onto the observed daily time-series of temperature, rainfall, wind and relative humidity from 1973 to early 2007. This methodology provides an estimate, based on the observed past weather, of what a realistic time-series affected by climate change may look like, assuming no change in year-to-year variability beyond that observed in the past 34 years.

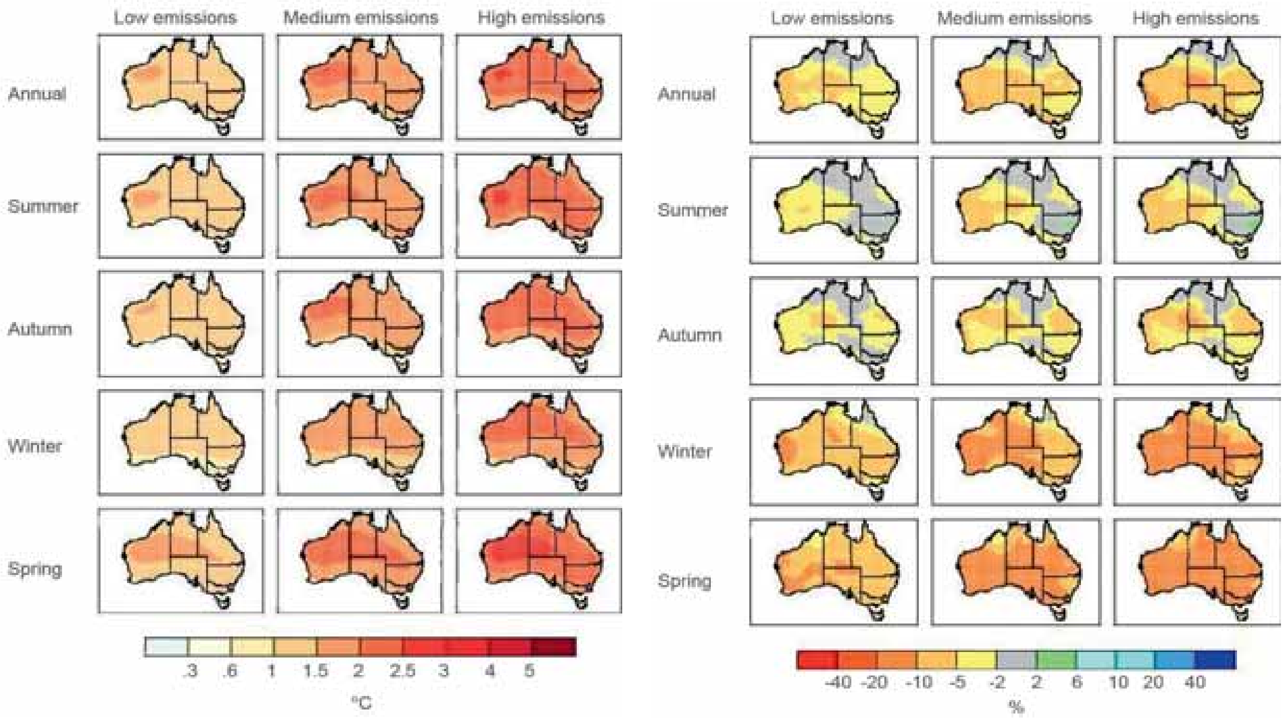



**Fig. 4.10. Median projected changes in temperature (left) and rainfall (right) for Australia in 2050.** Projections are broken down by season (annual, summer, autumn, winter, spring) and emissions scenario (low, medium, high).Source: CSIRO http://climatechangeinaustralia.gov.au

### *Cumulative annual fire danger in 2020 and 2050*

Table 4.2 shows the effects of climate change on the average ΣFFDI from July to June. The CCAM (Mark 3) high scenario produces the largest changes, while the CCAM (Mark 2) low scenario gives the smallest changes. In all simulations, the largest changes are in the interior of New South Wales and northern Victoria. As a general rule, coastal areas have smaller changes. However, the CCAM (Mark 3) simulations indicate changes on the mid-New South Wales coast similar to that for inland New South Wales. At Hobart, a slight decrease is seen in the scenarios using the CCAM (Mark 2) simulation.

The modelled changes are not linear; rather, there is more change between 2020 and 2050 than between 1990 and 2020. By 2020, the increase in ΣFFDI is generally 0–4% in the low scenarios and 0–10% in the high scenarios.

By 2050, the increase is generally 0–8% (low) and 10–30% (high). As a rule, changes expected in the high scenario are roughly twice as large as those in the low scenario. The maximum changes in the 2020 high scenarios are about the same as in the 2050 low scenarios; changes in Hobart are negligible. The annual ΣFFDI values mask much larger changes in the number of days with significant fire risk. Unsurprisingly, many of the characteristics are similar to what is seen with ΣFFDI.

**Table 4.2. Projected changes to FFDI for selected stations in south-eastern Australia.**
Annual (July to June) average cumulative FFDI (σFFDI) for 'present' conditions (based on 1973–2006 data), and for projected changes (% change) under the 2020 and 2050 climate change scenarios, relative to 1990. The outputs are for two climate change models: CCAM (Mark 2) results are denoted 'mk2' and CCAM (Mark 3) results are denoted 'mk3'. Source: Lucas *et al*. 2007

| Site | Present ΣFFDI | % change | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | 2020 low mk2 | 2020 high mk2 | 2020 low mk3 | 2020 high mk3 | 2050 low mk2 | 2050 high mk2 | 2050 low mk3 | 2050 high mk3 |
| Adelaide | 2708 | 2 | 5 | 3 | 8 | 3 | 16 | 5 | 25 |
| Amberley | 2919 | 2 | 7 | 1 | 6 | 4 | 24 | 3 | 19 |
| Bendigo | 2552 | 4 | 9 | 4 | 10 | 6 | 29 | 7 | 31 |
| Bourke | 4758 | 4 | 10 | 3 | 8 | 7 | 33 | 5 | 26 |
| Brisbane airport | 1990 | 1 | 5 | 0 | 4 | 3 | 19 | 2 | 16 |
| Canberra | 2493 | 3 | 9 | 3 | 11 | 6 | 30 | 7 | 37 |
| Ceduna | 4430 | 1 | 5 | 2 | 6 | 3 | 15 | 4 | 20 |
| Charleville | 6127 | 4 | 11 | 2 | 8 | 7 | 37 | 5 | 25 |
| Cobar | 4800 | 4 | 10 | 3 | 9 | 7 | 33 | 6 | 28 |
| Coffs Harbour | 1255 | 1 | 3 | 1 | 6 | 2 | 11 | 3 | 18 |
| Dubbo | 3153 | 4 | 11 | 4 | 10 | 7 | 34 | 6 | 32 |
| Hobart | 1314 | -1 | -1 | 0 | 0 | -1 | -1 | 0 | 3 |
| Launceston airport | 1349 | 0 | 1 | 1 | 6 | 0 | 8 | 3 | 22 |
| Laverton | 2056 | 1 | 6 | 2 | 8 | 4 | 23 | 5 | 30 |
| Melbourne airport | 2306 | 2 | 7 | 3 | 9 | 4 | 22 | 6 | 30 |
| Mildura | 5017 | 2 | 7 | 3 | 8 | 4 | 21 | 5 | 26 |
| Mt Gambier | 1910 | 1 | 3 | 2 | 5 | 2 | 11 | 3 | 18 |
| Moree | 3937 | 4 | 12 | 4 | 10 | 8 | 37 | 6 | 29 |
| Nowra | 1768 | 0 | 2 | 1 | 7 | 0 | 12 | 4 | 29 |
| Richmond | 2152 | 1 | 6 | 2 | 8 | 3 | 20 | 5 | 26 |
| Rockhampton | 3166 | 2 | 6 | 2 | 7 | 4 | 21 | 4 | 22 |
| Sale | 1713 | 0 | 5 | 1 | 8 | 2 | 19 | 4 | 31 |
| Sydney airport | 1897 | 1 | 4 | 3 | 10 | 2 | 11 | 6 | 31 |
| Wagga Wagga | 3319 | 3 | 9 | 3 | 10 | 6 | 29 | 6 | 33 |
| Williamtown | 1984 | 1 | 4 | 3 | 9 | 3 | 14 | 6 | 27 |
| Woomera | 7249 | 1 | 5 | 2 | 6 | 3 | 15 | 4 | 20 |

**Table 4.3. Projected changes to number of days with FFDI of at least 25.** Average number of days per year with FDR of 'very high' or 'extreme' (FFDI ≥ 25) and the percentage change (%) from the current value. Values for 'present' are for 1973–2007. CCAM (Mark 2): 'mk2' and CCAM (Mark 3) 'mk3', as per Table 4.2. Source: Lucas *et al.* 2007

| Site | Now | 2020 | | | | 2050 | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | Low mk2 | Low mk3 | High mk2 | High mk3 | Low mk2 | Low mk3 | High mk2 | High mk3 |
| Adelaide | 18.3 | 19.2 | 19.8 | 20.8 | 22.3 | 19.9 | 20.8 | 26.1 | 30.2 |
| % | – | 5 | 8 | 13 | 22 | 9 | 14 | 43 | 65 |
| Amberley | 13.3 | 14.5 | 14.2 | 16.4 | 15.7 | 15.3 | 14.8 | 22.7 | 20.9 |
| % | – | 8 | 6 | 23 | 18 | 15 | 11 | 70 | 57 |
| Bendigo | 13.9 | 15.6 | 16.1 | 17.5 | 18.4 | 16.6 | 17.1 | 25.2 | 28.6 |
| % | – | 12 | 16 | 26 | 32 | 20 | 23 | 81 | 106 |
| Bourke | 57.2 | 62.3 | 61.4 | 71.3 | 68.6 | 66.4 | 64.5 | 103.7 | 91.5 |
| % | – | 9 | 7 | 25 | 20 | 16 | 13 | 81 | 60 |
| Brisbane airport | 5.2 | 5.4 | 5.3 | 5.9 | 5.8 | 5.7 | 5.6 | 8.5 | 7.5 |
| % | – | 4 | 2 | 14 | 12 | 9 | 7 | 63 | 45 |
| Canberra | 16.8 | 18.3 | 18.9 | 21.5 | 22.8 | 20.0 | 20.6 | 29.9 | 33.4 |
| % | – | 9 | 13 | 28 | 36 | 19 | 23 | 78 | 98 |
| Ceduna | 46.4 | 47.7 | 48.0 | 49.4 | 50.5 | 48.5 | 49.0 | 56.5 | 58.6 |
| % | – | 3 | 3 | 6 | 9 | 5 | 6 | 22 | 26 |
| Charleville | 89.0 | 95.6 | 93.6 | 108.3 | 102.0 | 101.5 | 97.2 | 147.5 | 126.7 |
| % | – | 7 | 5 | 22 | 15 | 14 | 9 | 66 | 42 |
| Cobar | 56.0 | 61.4 | 60.8 | 69.9 | 67.9 | 65.2 | 64.0 | 99.5 | 91.8 |
| % | – | 10 | 8 | 25 | 21 | 16 | 14 | 78 | 64 |
| Coffs Harbour | 1.5 | 1.6 | 1.6 | 1.8 | 1.8 | 1.8 | 1.8 | 2.3 | 2.5 |
| % | – | 6 | 6 | 22 | 20 | 20 | 18 | 57 | 71 |
| Dubbo | 23.0 | 25.6 | 25.3 | 30.0 | 29.2 | 27.4 | 27.1 | 45.9 | 43.8 |
| % | – | 11 | 10 | 30 | 27 | 19 | 18 | 100 | 90 |
| Hobart | 2.0 | 2.0 | 2.0 | 2.0 | 2.1 | 2.0 | 2.1 | 2.0 | 2.2 |
| % | – | -3 | -3 | -2 | 5 | -2 | 2 | 0 | 8 |
| Launceston airport | 1.0 | 1.0 | 1.2 | 1.1 | 1.2 | 1.0 | 1.2 | 1.2 | 2.2 |
| % | – | -3 | 12 | 3 | 18 | 0 | 15 | 18 | 112 |
| Laverton | 11.8 | 12.0 | 12.3 | 12.8 | 13.6 | 12.4 | 12.8 | 16.7 | 19.2 |
| % | – | 2 | 4 | 9 | 15 | 5 | 9 | 42 | 63 |
| Melbourne airport | 14.8 | 15.7 | 15.9 | 17.0 | 17.6 | 16.2 | 16.5 | 21.2 | 23.6 |
| % | – | 6 | 7 | 15 | 19 | 9 | 12 | 43 | 59 |
| Mildura | 56.6 | 59.5 | 60.3 | 65.5 | 66.9 | 62.3 | 63.7 | 84.7 | 90.5 |
| % | – | 5 | 7 | 16 | 18 | 10 | 13 | 50 | 60 |
| Mt Gambier | 11.5 | 11.6 | 11.8 | 12.3 | 12.8 | 12.0 | 12.3 | 14.0 | 15.4 |
| % | – | 1 | 3 | 7 | 12 | 5 | 7 | 22 | 34 |
| Moree | 30.5 | 34.5 | 33.7 | 41.1 | 38.9 | 37.6 | 36.4 | 62.8 | 55.8 |
| % | – | 13 | 10 | 35 | 28 | 23 | 19 | 106 | 83 |
| Nowra | 8.8 | 8.7 | 9.1 | 9.2 | 10.3 | 8.9 | 9.6 | 10.8 | 14.7 |
| % | – | -1 | 3 | 5 | 17 | 2 | 10 | 23 | 68 |
| Richmond | 13.3 | 13.8 | 14.2 | 15.2 | 16.3 | 14.5 | 15.1 | 20.3 | 23.6 |
| % | – | 4 | 6 | 14 | 23 | 9 | 13 | 53 | 77 |
| Rockhampton | 11.2 | 12.0 | 11.9 | 13.2 | 13.5 | 12.4 | 12.8 | 18.6 | 19.4 |
| % | – | 7 | 6 | 17 | 20 | 10 | 14 | 66 | 73 |
| Sale | 5.4 | 5.4 | 5.7 | 5.9 | 7.1 | 5.7 | 6.3 | 8.1 | 11.1 |
| % | – | 1 | 7 | 10 | 32 | 6 | 18 | 50 | 107 |
| Sydney airport | 7.6 | 7.8 | 8.1 | 8.3 | 9.4 | 8.0 | 8.7 | 9.8 | 14.2 |
| % | – | 2 | 6 | 9 | 23 | 4 | 14 | 28 | 87 |
| Wagga Wagga | 32.6 | 34.8 | 35.0 | 39.7 | 40.3 | 37.1 | 37.2 | 56.3 | 57.6 |
| % | – | 7 | 7 | 22 | 24 | 14 | 14 | 73 | 77 |
| Williamtown | 10.3 | 10.8 | 11.2 | 11.5 | 12.8 | 11.3 | 11.9 | 13.9 | 17.8 |
| % | – | 6 | 9 | 12 | 25 | 10 | 16 | 36 | 73 |
| Woomera | 109.1 | 112.3 | 112.8 | 118.1 | 119.4 | 115.2 | 115.9 | 135.4 | 139.1 |
| % | – | 2 | 3 | 8 | 10 | 6 | 6 | 24 | 28 |

Note: – Not applicable.

### *Future days with very high and extreme (VHE) fire danger*

Table 4.3 shows the changes in all VHE days for various climate change scenarios. This category contains all days with an FFDI in excess of 25; it includes very high, severe and extreme days. By 2020, increases in VHE average 2–13% for the low emission scenarios and 10–30% for the high emission scenarios. By 2050, the range of percentage increase of occurrence of VHE days is much broader, averaging 5–23% for the low scenarios and from 20–100% for the high scenarios. The degree of change depends on the location, warming scenario and the particular global climate model used. For example, at Cobar between 64 and 100 VHE days are expected by 2050, compared to 56 at present, whereas at Melbourne Airport there would be 16 to 24 VHE days by 2050, compared with 15 at present. With respect to days of 'extreme' fire danger only, (FFDI > 50) modelling suggests that the number of such days increases 5-25% by 2020 for the low scenarios and 15-65% for the high scenarios. By 2050, the increases are generally 10-50% for the low scenarios and 100-300% for the high scenarios (See Table E1, p3, in Lucas et al. 2007).

### *Considerations and caveats*

By performing a seasonal analysis (data not shown), an estimate of changes in the timing of fire seasons can be made. The greatest changes in the median FFDI are seen in the season of highest fire danger, generally summer (December to February). A large change is also seen in the season prior to the peak season as well. Generally, this change is larger than that for the season immediately following the peak. The 'off season', usually winter (June to August), tends to have the smallest increase.

Taken together, these results suggest that fire seasons will start earlier and end slightly later, while being generally more intense throughout their length. This effect is most pronounced by 2050, although it should be apparent by 2020.

The changes to the fire season do not represent a uniform shift across all years. Rather, the fire seasons shift by different amounts in different years. Some change in the number of VHE days is seen in most years, suggesting that in most years some increase in fire danger will be observed. In many cases, the largest increases are seen in years that are already more extreme in the current climate (e.g. 1982–1983, 1997–1998, 2006–2007). By 2050, the scenarios suggest that what were more 'marginal' years, such as the late 1970s and early 1980s, become equivalent to (or exceed) what are the more extreme years in the current climate. However, many of the less extreme years, which show few VHE days, remain so in the projections, with little increase expected.

Climate change is likely to cause an increase in the number of very high fire danger days in some areas. This may increase the severity of fires. Severly burnt subalpine snowgum (*Eucalyptus pauciflora*) woodland with healthy understorey, on Spion Kopje ridge, Bogong High Plains region, Alpine National Park, Victoria, 2003.

Source: Henrick Wahren, La Trobe University

### *Further research*

Our current understanding about the future impacts of climate change on bushfire and biodiversity remains incomplete. There are many gaps in our knowledge. First and foremost, detailed climate change/fire-weather projections are needed on a national scale. To date, studies have focused on the south-eastern portion of the country, a region where substantial economic assets are found. However, northern regions of Australia are where most fire activity occurs every year. Issues ranging from conservation outcomes to carbon sequestration are likely to be influenced by the way fire regimes change, and how they are managed, in northern Australia.

The fire-weather projections presented in this report are created using the simulated changes in daily variability projected onto the observed weather record for the previous 30+ years. While this is an advantage in that it uses the real behaviour of the atmosphere to quantify future changes, it makes the assumption that future climate variability will not change from year to year. Since some climate models simulate increases in El Niño events, which have a two- to seven-year cycle, we should try to include this effect in the fire-weather projections. Further research is needed to determine the most reliable models for simulating future changes in ENSO – initial work by Min *et al.* (2005), van Oldenborough *et al.* (2005) and Smith and Chandler (in press) have identified five of the best models based on ENSO and rainfall criteria.

Fire projections in this report are based on two CSIRO climate models. This small sample size means we cannot say much about uncertainty due to differences between models. One method of reducing this uncertainty would be to use a wider variety of climate models to make the fire-weather projections. Up to 23 models are available, but only a few would have suitable daily data for assessing fire-weather. Studies using higher-resolution models may also be effective in targeting specific regions, rather than the broad-brush approach used to date.

Other, more difficult, sources of uncertainty not directly tied to the models remain. The impact of climate change on ignitions, either natural or anthropogenic, remains difficult to quantify and requires further research.

### *Conclusions*

A detailed exploration of the potential impact of climate change on fire-weather in south-eastern Australia is presented. Simulations from two CSIRO climate models using two greenhouse gas and aerosol emissions scenarios have been combined with historical weather observations to assess the changes to fire-weather expected by 2020 and 2050.

In general, fire-weather conditions are expected to worsen. By 2020, the increase in ΣFFDI is generally 0–4% in the low emissions scenarios and 0–10% in the high emissions scenarios. By 2050, the increase is generally 0–8% (low) and 10–30% (high). The largest changes are expected in northern New South Wales. Little change is expected in Tasmania.

By 2020, increases in VHE days are generally 2–13% for the low scenarios and 10–30% for the high scenarios. By 2050, the range is much broader, generally 5–23% for the low scenarios and from 20–100% for the high scenarios. The fire seasons are likely to become longer, starting earlier in the year.

These results are placed in the context of the current climate and its trends. During the last several years in south-eastern Australia, including the 2006–2007 season, particularly severe fire-weather conditions have been observed. In many cases, the conditions far exceed the projections in the high scenarios of 2050. Are the models (or our methodology) too conservative or is some other factor at work?

Examining longer-term observations at eight stations, back to the early 1940s in many cases, reveals considerable interdecadal variability. Periods of increasing and decreasing fire-weather danger are apparent in the record. The peaks of these 'cycles' occur roughly every 20 years and the time-series might suggest that we are at (or near) a peak, although there is no physical basis on which to estimate when or to what extent a decrease might occur.

There is also evidence for anthropogenic climate change being a driver of this upswing. The current peaks in ΣFFDI are much higher than observed in past instances. At many sites, there is also a greater number of VHE days and the fire season is starting earlier. Finally, the severity and length of the recent drought (e.g. Nicholls 2006) and the associated fire danger has not been seen in the available records.

The hypothesis posited by Lucas *et al.* (2007) is that the naturally occurring peak in fire danger due to interdecadal variability may have been exacerbated by climate change. This hypothesis can be tested by observation over the next few years to decades. If correct, then it might be expected that fire-weather conditions will return to levels something more along the lines of those suggested in the 2020 scenarios. If fire danger conditions stay this high, then the conclusion must be that the models used to make these projections are too conservative. Whatever the case, continued observation, as well as improved modelling are required to resolve this question.

What of the human impacts of these projected changes? The last few years, particularly the 2006–2007 fire season, may provide an indication for the future. Early starts to the fire season in south-eastern Australia suggest a smaller window for pre-season prescribed burning for fuel reduction. Potentially more frequent and more intense fires imply that more resources will be required to maintain current capacity in bushfire suppression. Shorter intervals between fires, such as those that resulted form fires that burned in eastern Victoria during 2002–2003 and 2006–2007, pose increased risks to multiple landscape values, such as people and property, and biodiversity.

## 4.3 Climate change and fuels

Climate change will have potential impacts on fuel dynamics. Fuel amount – the net result of production and decomposition – is influenced by broad patterns of temperature and precipitation. The vegetation ‘fertilisation’ effect of increasing atmospheric $CO_2$ will also impact on fuels, as will interactions between climate drivers, and biotic processes (e.g. herbivory). The latest global meta-analyses suggest a minimum increase of about 20% in above-ground biomass in woody communities across a wide range of open-topped $CO_2$ experiments (Wang 2007). In forested systems, this will probably translate into increased litterfall and standing fine fuels in locations where rainfall and temperature are non-limiting. Nemani *et al.* (2003) concluded that global net primary production (NPP) increased by 6% between 1980 and 1999; some of this was due to $CO_2$ fertilisation, and some by changed patterns of temperature, moisture and solar radiation. Increased growth in response to CO2 fertilisation has also been demonstrated for grassy vegetation (Stokes *et al.* 2003, 2008; Ziska *et al.* 2005; Dr Mark Hovenden, University of Tasmania, pers. comm.). More precise predictions for Australia are limited by the range of empirical data available because we have no experimental data on the $CO_2$ effects on biomass (fuel) accumulation in post-fire environments in Australia. We also require information on the effects of elevated $CO_2$ on rate of decomposition of litter. We explore some of these potential impacts of climate change on fuels in the following section.

### 4.3.1 Nature and importance of fuel

Fuel is any living or dead combustible material. The fuel complex can be described in terms of quantity (usually expressed as oven-dried weight and referred to as fuel load), fineness, vertical position in the fuel strata, and whether dead or alive. For management purposes, fuel quantity is most commonly expressed in tonnes per hectare (t ha-1) or kilograms per square metre (kg m-$^2$). Fuel can be classified by its location in a fuel array including: surface fuel; near-surface fuel located near the soil surface but suspended in grasses and low shrubs; elevated fuel generally comprised of shrubs; bark fuel; and canopy fuel. Litter is dead plant material that has been shed from living plants, at or near the soil surface, or elevated in shrubs and trees. Some authors (e.g. Birk 1979a, 1979b) include standing dead grass as litter, while others (e.g. Medwecka-Kornas 1971 in Birk 1979a) do not. Different authors refer to different fineness classes when defining litter.

McArthur’s (1967) forest fire behaviour table uses fine (< 6 mm) surface litter as the sole fuel quantity parameter for predicting fire behaviour in open forests. Therefore, most Australian research into the effect of fuel load on fire behaviour has focused on this component of the fuel array (Catchpole 2002). McArthur (1967) assumed a linear relationship between fuel load and the forward rate of spread for a given set of meteorological conditions and slope in in open eucalypt forest (Fig 4.11; Noble *et al.* 1980). Burrows (1999 a,b) found a linear relationship between rate of spread and slope under wind conditions but could not find a relationship between the two variables for wind driven fires in laboratory and field experiments using jarrah litter (Catchpole 2002). Gould *et al.* (2007) similarly found a weak relationship between surface fuel attributes and rate of spread in jarrah forest for wind speeds less than 2 m/sec, but there was no correlation between surface fuel and rate of spread at higher wind speeds.

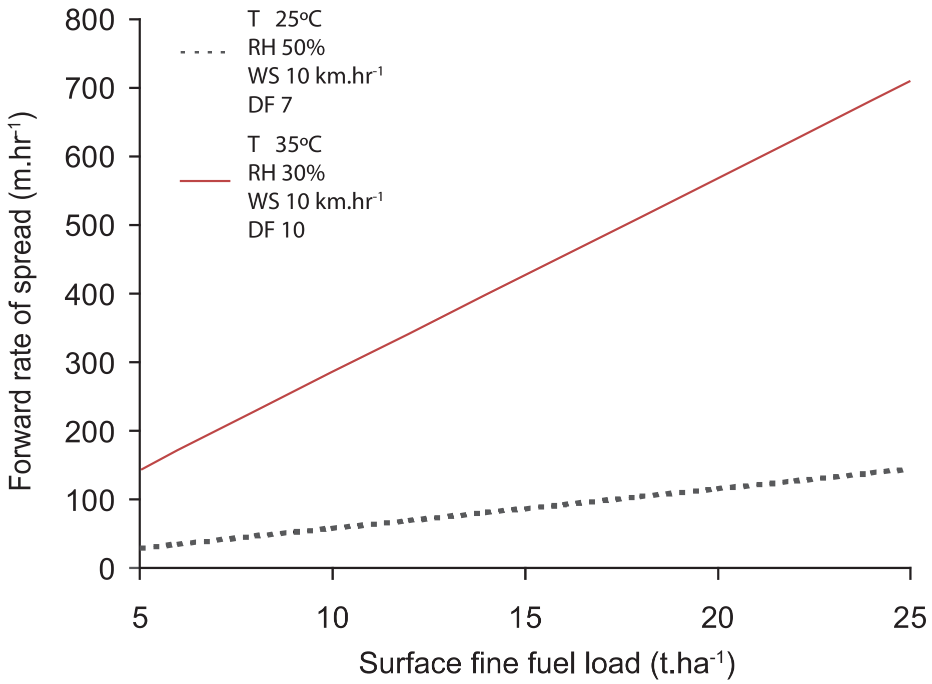



**Fig. 4.11. Predicted relationship between forward rate of spread and fuel load of fires.** Fires are burning under two sets of meteorological conditions with different temperature (T), relative humidity (RH), wind speed (WS) and drought factor (DF). Derived from McArthur (1967).

### 4.3.2 Fuel dynamics

The rate of accumulation of fine surface litter is a function of the rate of litter accession (or production) and decomposition. It varies seasonally and annually, and is broadly under the control of climate. Patterns of annual litter fall vary, depending on climatic influences and forest type. Walker (1981) reviewed literature on seasonality of litter production and found a marked seasonal pattern for all parts of Australia. In north-eastern New South Wales, litter fall peaks in summer and declines in winter. For example, a two-year record of litter fall from tall open forest dominated by *Eucalyptus pilularis* near Taree, New South Wales, demonstrated that litter fall was greatest for summer (37% of annual total), followed by spring (32%), autumn (20%), and winter (11%). In northern Australia, where the fine fuels are predominantly grassy, litter fuels may nevertheless constitute a significant part of the fuel bed in mesic savannas, and peak in the late dry season (Williams *et al.* 1998).

Litter decomposition is a result of biological processes involving decomposing organisms such as fungi, microinvertebrates and macroinvertebrates. The rate of decomposition is controlled by a range of factors including moisture content of litter, temperature and nutrient status (Murphy *et al.* 1998). Site-specific moisture and temperature regimes partly determine decomposition rates (Walker 1981). Wood (1970) reported the percentage weight loss of eucalypt leaves in the first year for a range of species and site conditions. In the case of snow gum (*E. pauciflora*) and alpine ash (*E. delegatensis*), the decomposition rates were higher for wet sites, although the effect of increased moisture was much more apparent for *E. delegatensis*. Further, Raison *et al.* (1986) demonstrated that litter decomposition rates were lower on recently burnt sub-alpine forest sites compared with long-unburnt sites, apparently because of greater aridity resulting from the removal the understorey, which produced less shading and reduction in the depth and mulching effect of a deeper litter layer.

#### *Fuel accumulation – the negative exponential model of fuel dynamics*

The most commonly reported model of litter dynamics is the negative exponential model, initially proposed by Kerridge (1948) and Jenny *et al.* (1949), for modelling energy flow in ecosystems. Olson (1963) expanded on this by incorporating continuous steady litter fall, which is better suited to the eucalypt forests of Australia (Walker 1981), and which has been used for modelling litter accumulation in a wide range of vegetation.

Olson (1963) demonstrated that:

$$X = \frac{A}{k} \times \left(1 - e^{(-kt)}\right)$$

The rate at which the litter mass (X) accumulates with time (t in years) depends on the value of litter accession (A) and the decomposition constant (k), which is defined as the average fraction of the steady-state litter load that decomposes each year. Therefore, the curve (e.g. Fig. 4.12) defining the relationship between accumulated fine surface litter and time since fire (also known as the fuel or litter accumulation curve) may be one of a number of variants of the same basic shape (O'Connell 1988). Climate change has major implications for fuel dynamics, as it may affect litter accession and decomposition rates, and therefore alter the nature of fuel accumulation at a site. This, in turn, will impact on rates of fire spread and intensity.

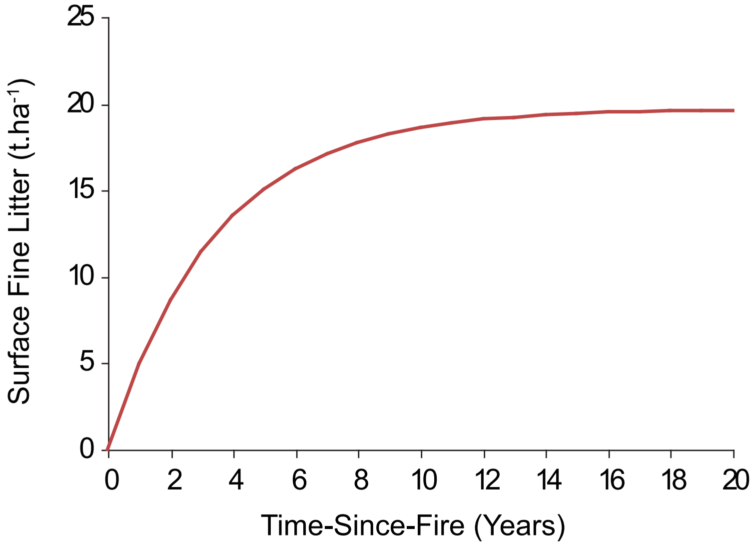



**Fig. 4.12. Generalised shape of fine surface litter accumulation in a *Eucalyptus pilularis* forest in north-eastern New South Wales.** Source: After Birk and Bridges (1989).

As can be seen in Fig. 4.12, fuel load is predicted to reach a stable, maximum (quasi-equilibrium) value over time as rates of accession are balanced by rates of decomposition – in this case, of around 18 t/ha. Some quasi-equilibrium fine-fuel loads in other forest types from south-eastern Australia, listed for drier to wetter forests, are:

- red ironbark (*Eucalyptus sideroxylon*) and grey box (*E. microcarpa*), 7 t/ha
- mixed messmate (*E. obliqua*) and narrow-leaved peppermint (*E. radiata*) forests, 15 t/ha
- mountain ash (*E. regnans*), 25 t/ha. (Source: Esplin *et al.* 2003)

### 4.3.3 Potential impacts of climate change on fuels

*Climate change and forest fuel loads – an example from the karri and jarrah forests of south-west Western Australia*

The productivity of terrestrial ecosystems (measured either as Net Primary Productivity (NPP) or more crudely as accumulated above-ground biomass) is strongly correlated with gross measures of climate including rainfall and temperature, such that rates of biomass accumulation (and litterfall) increase as available moisture and temperature increase (Begon *et al.* 2006; Whittaker 1975). While this generalisation holds at a global scale, within regions, local factors of site quality affect biomass accumulation rates, with vegetation on steeper slopes and coarser soils generally less productive than those on flatter land and finer-textured soils (e.g. Johnson *et al.* 2000). Nevertheless, at a regional scale, climate change may be expected to influence fuel dynamics, and we illustrate this by examining the influence of declining rainfall on fuel dynamics in the temperate eucalypt forests and woodlands of south-west Western Australia.

In south-western Australia, the karri (*Eucalyptus diversicolor*) and jarrah (*E. marginata*) forests of the Warren and Jarrah Forest Biogeographic Regions (Fig. 4.13) are characterised by moderate to extreme fuel loads, which are capable of carrying surface fires again within three to 10 years of previous fire (Table 4.4; McCaw *et al.* 1993).

Karri (*Eucalytptus diversicolor*) forest of the Pemberton region, south-west of Western Australia.
Source: Angas Hopkins

The dominant environmental gradient in the region is one of increasing aridity as distance north and east from the southern corner of Western Australia increases; rainfall decreases from >1400 mm/year to <700 mm/year, and potential evapotranspiration increases ( Dell *et al.* 1989; Gentilli 1989; Burrows and Wardell-Johnson 2003; CSIRO and Bureau of Meteorology 2007). Karri forest becomes more open and gives way to jarrah forest, which in turn becomes more open and gives way to wandoo (*E. wandoo*) woodlands. Rates of fuel load accumulation with time since fire decrease along this aridity gradient – reaching from 35–58 t/ha after 25 years in karri forests, 15–23 t/ha in jarrah forests and 5–15 t/ha in wandoo (Sneeuwjagt and Peet 1985) – as do rates of decomposition (Brown *et al.* 1996). Local variation in site quality can also influence both forest type and fuel accumulation rates, with jarrah rather than karri occupying some high-rainfall areas on low-nutrient soils, so that patterns of fuel accumulation are more complex than implied by a simple correlation with rainfall. Nevertheless, it is instructive to consider this simple example of how declining moisture availability might affect fuel loads in these forests at the broadest scale.

While the possible water-use efficiency benefits of increasing $CO_2$ are unclear, and may or may not offset the impact of lower water availability on biomass production, here we must (for now) assume that fuel accumulation rates (as determined by the balance between rates of litter fall and decomposition) will relate to future rainfall in the same way that they do today. In that case, under a scenario of a 20% decline in rainfall (Fig. 4.10) we estimate future maximum fuel loads in the karri forest of 35 t/ha after 25 years (equivalent to present-day open karri with 30% canopy cover; Sneeuwjagt and Peet 1985) in the wettest parts of the species range; fuel loads typical of present-day jarrah forests through much of the present karri forest region; and very low fuel load communities in areas currently occupied by dry, open jarrah forests and wandoo woodlands (Table 4.4).

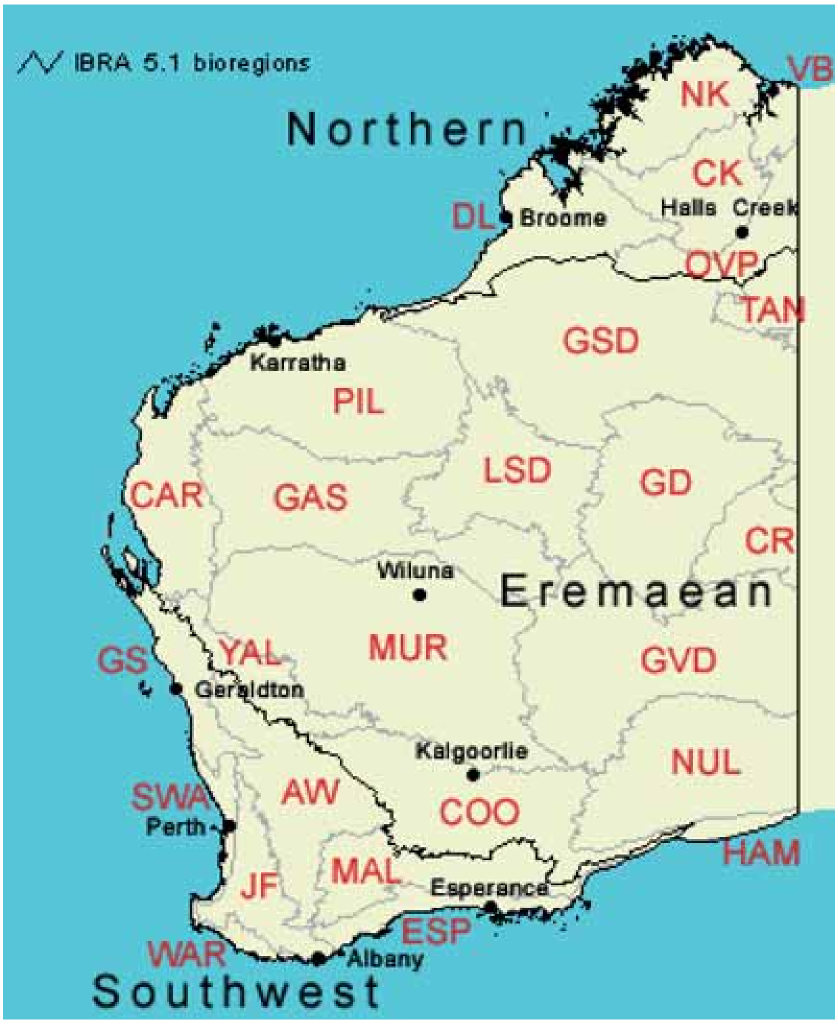



**Fig. 4.13. Interim Biogeographic Regionalisation of Australia (IBRA) for Western Australia.** Map shows boundaries of the major botanical regions (black lines) and IBRA regions (grey lines). Within the south-west there are seven biogeographic regions: GS – Geraldton Sandplain, SWA – Swan Coastal plains, AW – Avon Wheatbelt, JF – Jarrah Forest, WAR – Warren, MAL – Mallee, ESP – Esperance Plains (Environment Australia 1996). Karri forest is restricted largely to the Warren region, with small outliers to the west near Margaret River and to the east in the Porongorup Ranges. Jarrah forest is restricted to the Jarrah Forest and Swan Coastal Plains regions.Source: FloraBasehttp://florabase.calm.wa.gov.au/help/ibra/ After: Australian State of the Environment Committee (2001), and Western Australian Herbarium (1998)

It is possible to identify fuel loads below which fire suppression is feasible under given means (fire-fighting resources) and FFDI conditions. For example, Gill *et al.* (1987) set 8 t/ha as that which gave an intensity of 4000 kW/m on level ground in forests when FFDI=100. For south-eastern Australian sclerophyll forests, fuel load levels of 10 and 15 t/ha are identified as critical threshold levels in relation to fire suppression effectiveness, representing likely fire intensities of around 1000 kW/m under moderate to high and low to moderate fire danger conditions, respectively (e.g. see Luke and McArthur 1978; Tolhurst and Cheney 1999). In south-western Australia, the threshold fuel load in jarrah forest from a fire suppression perspective is 8–9 t/ha and for karri is 19–20 t/ha. This difference takes account of the fact that karri fuels are typically wetter, so despite a higher fuel load, fire intensity is reduced on this parameter (McCaw and Burrows 1989; Burrows 2008; N Burrows pers. comm.). In karri forest, the 19–20 t/ha level is reached within three to 10 years of last fire (depending on forest type and site productivity). In jarrah, the 9–10 t/ha level is reached within five to 15 years. Under a climate change scenario of a 20% reduction in rainfall, these levels take from 10 to >25 years to reach in karri (although based on annual rainfall we project that only the drier karri forest fuel load levels will occur), eight to >25 years in jarrah, and probably never in wandoo. Thus, the reduction in fuel load accumulation rates is much greater than might be expected in relation to the magnitude of the hypothesised reduction in rainfall. Such an outcome would markedly change the fire dynamics of this region. High fire danger weather would increase the risk of conditions conducive to rapid fire spread and high-intensity fires on the one hand – perhaps most acute in historically wetter forests such as karri and tingle which, under climate change, will be drier for longer each year. On the other hand, slower fuel accumulation rates limiting the availability of fuel for fires so that fires may not propagate or may be more readily suppressed. At this time there is no clear evidence as to whether forest productivity and fuel accumulation rates have changed in response to decreasing rainfall in south-western Australia over the past 30 years, but a range of studies here – spanning the period from the 1960s to 1990s (e.g. McCaw *et al.* 1996) – provides a useful platform for investigation of this issue.

However, despite a marked decline in rainfall in south-western Australia over the past 30 years, there is so far no published evidence of change in either ecosystem productivity or in the distribution of vegetation types. This may indicate system inertia in terms of if, and how quickly, wetter forest types might be converted to drier forest types as conditions become limiting for recruitment and growth – with fire the most likely driver of rapid change in state. It may also reflect a lack of suitable data, and field-based studies are urgently required in these vegetation types (and at the ecotones between types) to identify the extent and rate of change.

### *$CO_2$ 'fertilisation' effect and fuels*

Increases in the concentration of $CO_2$ in the atmosphere can increase the growth of plants in the absence of change in any other variable such as rainfall or temperature. As the concentration of $CO_2$ in the atmosphere has changed globally, an increase in growth can be expected globally if other factors do not intervene to curtail its effect. Thus an increase in fuel loads might be expected.

The latest meta-analyses suggest a minimum increase of about 20% in above-ground biomass in woody communities across a wide range of open-topped $CO_2$ experiments (Wang 2007). Resource limitation mediates this increase with higher nitrogen levels increasing above-ground biomass accumulation, as does system maturity (Wang 2007). However, we have no empirical data on the effects of elevated $CO_2$ on biomass (fuel) accumulation in post-fire woody environments in Australia from open-top experiments. Nevertheless, we could anticipate that: (i) post-fire woody growth is enhanced; and (ii) the degree of enhancement will differ between resource-poor and resource-rich environments. In sclerophyll forests, woodlands and shrublands, 'maturity' is regularly curtailed by fire (e.g. Keith *et al.* 2002a), hence vegetation may be maintained in a state that is highly responsive to elevated $CO_2$. If fuel production is enhanced under elevated $CO_2$ and fire-weather becomes generally more severe, then all else being equal, fire 'severity' in sclerophyllous shrub and tree-dominated systems is likely to increase.

The impacts of elevated $CO_2$ on systems dominated by grassy fuels have been studied in several 'Free Air Carbon dioxide Enrichment' (FACE) experiments. In grass steppe in the USA, Morgan *et al.* (2004, 2007) found increases in grass yields of 41% but also that C3 grasses may gain an advantage over C4 grasses, and shrubs over grasses. Thus the situation is by no means simple; both species composition, and fuel structure and composition, can change. A similar result for C4 grasses (*Themeda triandra*) and associated seedlings of common trees (acacias and eucalypts) was found in a FACE experiment in the tropical savannas of the Townsville region (Stokes *et al.* 2003; Stokes and Ash 2006) The Townsville FACE experiment also showed that crude protein levels in the *Themeda* declined (leading to a higher C:N ratio), which could potentially cause decomposition rates to decline. This, in turn, could cause grassy fuels to accumulate to higher levels. Although the growth of trees may be enhanced by increases in atmospheric $CO_2$ (Norby *et al.* 2005), the way this increase is partitioned (McCarthy *et al.* 2006) is important in the fuel context. In hardwoods in North America, litterfall did not increase; but for conifer needles and bark it did (Finzi *et al.* 2001). In addition, stage of forest growth and soil properties may affect the result (McCarthy *et al.* 2006).

**Table 4.4. Fuel load (t/ha) in relation to time since last fire in major forest types of south-western Australia, along a gradient of declining rainfall/increasing potential evapotranspiration.** The gradient is from closed (100% canopy cover) karri (Eucalyptus diversicolor) and wet jarrah (E. marginata) forests, to wandoo (E. wandoo) (open 80% canopy cover) woodlands (data interpreted from Sneeuwjagt and Peet 1985). Columns with values in red are estimated fuel loads assuming a 20% reduction in annual rainfall in the vegetation type in the adjacent column, and a linear relationship between rainfall and forest type productivity. Cell values coloured yellow show the time (age in years) to accumulation of critical fuel loads in relation to fire suppression for karri (19–20 t/ha), and cell values coloured blue are for jarrah (9–10 t/ha) under present and projected -20% rainfall conditions. Note that once dry, open jarrah forest (20% canopy cover) is reached, fuel loads under the climate change assumptions used here are not projected to exceed 10 t/ha within 30 years of previous fire.

| Age | Karri 100% | -20% | Karri 60% | -20% | Karri 30% | -20% | Jarrah 80% | -20% | Jarrah 50% | -20% | Jarrah 20% | -20% | Wandoo 60% | -20% | Wandoo 40% |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 12 | 4 | 7 | 3 | 4 | 2 | 3.5 | 2 | 2.4 | 1 | 1 | 1 | 1.6 | 1 | 1 |
| 2 | 16 | 6 | 10 | 5 | 6 | 3.5 | 5 | 3.5 | 4 | 1.6 | 1.6 | 1.6 | 2.4 | 1.6 | 1.6 |
| 3 | 19 | 8 | 12 | 7 | 8 | 5 | 7.2 | 5 | 5.2 | 2.5 | 2.5 | 2.3 | 3.2 | 2.3 | 2.3 |
| 4 | 22 | 10 | 15 | 8 | 10 | 6 | 8.5 | 6 | 6.3 | 3.4 | 3.4 | 2.8 | 4 | 2.8 | 2.8 |
| 5 | 24 | 12 | 17 | 9 | 12 | 7 | 9.8 | 7 | 7.5 | 4.2 | 4.2 | 3.3 | 4.6 | 3.3 | 3.3 |
| 6 | 27 | 14 | 19 | 10 | 14 | 8 | 10.8 | 8 | 8.5 | 5 | 5 | 3.7 | 5.1 | 3.7 | 3.7 |
| 7 | 29 | 15 | 21 | 11 | 15 | 8 | 11.8 | 8 | 9.5 | 5.8 | 5.8 | 4 | 5.6 | 4 | 4 |
| 8 | 32 | 17 | 23 | 12 | 17 | 9 | 12.8 | 9 | 10.3 | 6.5 | 6.5 | 4.3 | 6 | 4.3 | 4.3 |
| 10 | 36 | 20 | 26 | 13 | 20 | 10 | 14.4 | 10 | 11.5 | 7.7 | 7.7 | 4.8 | 6.7 | 4.8 | 4.8 |
| 12 | 41 | 23 | 30 | . | 23 | 11 | 15.5 | 11 | 12.7 | 8.8 | 8.8 | 5.4 | 7.3 | 5.4 | 5.4 |
| 15 | 46 | 20 | 34 | 14 | 25 | 12 | 17.5 | 12 | 14.2 | 10.5 | 10.5 | 6 | 8.2 | 6 | 6 |
| 20 | 52 | 25 | 40 | 16 | 30 | 13 | 20.2 | 13 | 16.5 | 12.7 | 12.7 | 7 | 9.5 | 7 | 7 |
| 25 | 58 | 30 | 44 | 18 | 35 | 15 | 22.5 | 15 | 18.5 | 15 | 14.8 | 7.7 | 10.4 | 7.7 | 7.7 |
| 30 | | | | | | | | | | | | | 11.2 | 8.3 | 8.3 |

Thus, there is the potential for changes in fuel type and for increases in fuel load as a consequence of increased atmospheric $CO_2$, but there appears to be no published basis for predicting an increase in litter in *Eucalyptus* stands at present. There is no suggestion for a decrease in fuel load within any one type, as a consequence of increased atmospheric $CO_2$. We do not know whether some other factor may prove limiting (e.g. water and/or nutrients) in different vegetation types (many Australian soils are low in nutrients and much of southern Australia is projected to become drier), and so offset any benefit of higher $CO_2$ in terms of vegetation productivity (and thus fuel loads).

## BOX 4.1. Climate change, fuels and interactions with decomposers

Importance of invertebrates for biodiversity conservation and ecosystem function

Although insects and other invertebrates are often seen as small and insignificant, their sheer weight of numbers, diversity of forms and presence in most habitats means their importance should not be underrated. Insects and other invertebrates are the most diverse group of organisms on Earth (May 1990), with many beneficial and detrimental effects on humans and natural ecosystems. Invertebrates play essential roles in the following ecosystem processes (Schowalter 1996; Wolters *et al.* 2000):

- they facilitate the propagation of plants through pollination and seed dispersal
- they influence the composition of plant communities through herbivory and seed predation
- they play a pivotal role in nutrient cycling through the breakdown of leaf litter and wood, dispersal of fungal spores, disposal of carrion and dung, and through soil turnover
- they influence vertebrate animal communities through their role as a food source, and by acting as parasites and as vectors for diseases.

To understand the full consequences of global climate change it is critical that we determine how this biodiversity and ecosystem functioning might be altered.

### *Consequences of climate change for invertebrates and the role they play*

Climate change has the potential to profoundly influence invertebrate biodiversity at a range of scales. Perhaps of even greater concern are the unknown consequences for ecosystem functioning through disruption of plant–invertebrate interactions, and processes of decomposition and nutrient cycling.

### *Biodiversity and plant–invertebrate interactions*

As indicated in section 2.3, current predictions suggest substantial alteration to species' ranges and abundances in response to changed habitat suitability (Beaumont and Hughes 2002; Hickling *et al.* 2005), with the possible extinction of many taxa because of physical (geographical and altitudinal) limits to movement (Thomas *et al.* 2004). Species that comprise a community are unlikely to shift together, with substantial time lags and periods of reorganisation (Andrew and Hughes 2005), leading to unknown consequences for invertebrate associations and mutualistic interactions.

Because of the strong coupling between plant and invertebrate life history strategies, recent research has suggested that climate change may have substantial implications for both biologically and commercially important invertebrate–plant interactions (Cannon 1998; Bale *et al.* 2002; Harrington *et al.* 2001; Throop *et al.* 2004; Botes *et al.* 2006). Some of the issues are:

- advanced flowering date – changes in pollination and seed set if times of invertebrate development, flight and foraging do not match plant life histories
- increase in insect pests – increased growing season may mean more pests for longer periods, with more generations of pests per year. Lack of winter mortality, extended periods of activity and increases in distribution may lead to more pests over a wider area. Direct fertilisation effect of $CO_2$ may lead to increased plant productivity, leading to lower nitrogen concentration as C:N ratios rise.
  This may lead to reduced nutritive value to herbivores, which in turn may lead to increases in herbivory to compensate for lower nutritional levels

- biological control – increased environmental variability might disrupt natural and implemented biological control processes (i.e. by disrupting the synchrony between growth, development and reproduction of host and herbivore/predator, etc.).

### *Nutrient cycling, climate change and ecosystem function*

Interactions among vegetation, soil flora and litter, and soil-dwelling invertebrates are responsible for regulation of carbon mineralisation and immobilisation in litter and soil, and hence the availability of nutrients for plant growth (Schowalter 1996). Climate change may influence nutrient cycling through:

- increased concentrations of leaf phenolics and tannins, which changes litter quality (palatability) for decomposers (Kasurinen *et al.* 2007), leading to altered rates of leaf litter decomposition (and hence fuel build-up). These rates depend on soil temperature and moisture – factors that might alter with changed climate
- changes to the carbon balance. Plant litter is a critical issue, because the decomposition of plant litter (and incorporation into soil organic matter) is a key component of the global carbon budget, contributing approximately 70% to the total annual carbon flux. Changes in the rate of decomposition and associated $CO_2$ release from dead plant material as a consequence of global warming could have profound repercussions for terrestrial carbon sequestration, feeding back to atmospheric $CO_2$ concentrations and the global climate (Cornelissen *et al.* 2007). These issues are serious considerations for both fuel management and the maintenance of carbon budgets.

### *Interactions between climate change and fire regime: consequences for invertebrates and the role they play*

Fire influences the composition and structure of invertebrate communities both directly and indirectly through changes to habitat structure (Mouillot *et al.* 2002; York 1999). There are substantial, but poorly understood, interactions between invertebrates and fire regimes, primarily because of their effects on litter (fuel) decomposition and nutrient cycling. A diverse group of bacteria, protozoa, fungi and invertebrate animals mediates the process of decomposition and nutrient release from organic inputs. Invertebrates modify microbial activity by processes such as litter fragmentation, mixing, dispersion of microflora and chemical changes that occur during the digestion of leaf litter. The close relationship between vegetation change and soil carbon dynamics suggests that any disruption of the coupling between plants and soil organisms as a result of global change may have deleterious consequences for functioning of terrestrial ecosystems (Wolters *et al.* 2000).

Soil and litter biota are a sensitive link between plant detritus and the availability of nutrients to plant uptake, with any factors affecting the quality or quantity of plant detritus likely to change this link (Swift *et al.* 1998). Changes in rates of litter (fuel) accumulation will also lead to subsequent changes in the fire regime (intensity and frequency of fires). Subsequent deleterious effects on terrestrial invertebrates, with the potential for negative feedback cycles, could further alter trophic interactions and rates of litter decomposition. It has been suggested, however, that trophic interactions between fungi and invertebrates (Frouz *et al.* 2002), and ectomycorrhizal fungi and host plants (Malcolm *et al.* 2008), could lead to feedback cycles mediating rates of decomposition (leading to increased decomposition in enriched litter).

The nature of these interactions and the overall resilience of invertebrates to altered fire regimes are poorly understood. Rising $CO_2$ may lead to changing decomposition rates, and many invertebrate decomposers are known to be sensitive to interval (York 1999). However, the balance between production and decomposition of litter (and hence ultimate fuel effect) will depend in complex ways on how the biota controlling decomposition respond to changes in temperature, rainfall $CO_2$ and fire regimes themselves. All are areas of high priority for ongoing research.

*Climate change and fuels – management implications*

The implications for landscape management of these scenarios of changed fuel dynamics under climate change are complex. Rising temperatures and drier landscapes will almost certainly bring a higher risk of fire-weather conducive to fire spread. However, the implications of climate change scenarios for fuels are less certain. Lower rainfall may lower fuel accumulation rates, and thus may limit the availability of fuel for fires and the propagation potential of fire, or may make fire more readily suppressed. On the other hand, lower fuel moisture levels may offset this effect to some extent. The effect of $CO_2$ fertilisation on vegetation growth, which is unknown at regional scales, adds another dimension of complexity to the potential outcomes of climate change–fuel–fire regime interactions. Regardless of whether slower fuel load accumulation rates may help fire managers to cope with a worsening fire-weather scenario, there may be adverse biodiversity consequences as a consequence of declining moisture availability, adding to the complexity of landscape management.

Despite the uncertainty of climate change outcomes for fuel illustrated here, the example illustrates how space-for-time studies along environmental gradients (in this case, moisture) can be used to evaluate potential impacts of climate change on fuels. An approach such as this provides explicit fuel levels and rates of change that can be monitored using conventional fuel monitoring techniques. More analyses such as this for south-west Western Australia, and for different vegetation and fuel types elsewhere in Australia, would be instructive.

## 4.4 Climate change and ignitions

Prior to the arrival of people in Australia, almost all ignitions would have been the result of lightning. The importance of lightning as a cause of fires in new-world landscapes was greatly underestimated during the early phases of European settlement (Baker 2002). Only with the growth of modern meteorological science in the 20$^{th}$ century, and the more recent advent of ground- and satellite-based lightning detection instrumentation, has more accurate data become available on the numbers of lightning strikes per unit area per year, and on the numbers (and proportion) of fires started by lightning.

The modern frequency of ground-flash lightning strikes in Australia shows a strong north to south gradient, with highest values (up to 8 km/year) in the tropical north-west and lowest values (~0.5 km/year) in southern Australia and Tasmania (Kuleshov *et al.* 2006). Values are generally low through arid central Australia (0.5 to 2 km/year) and moderate along the eastern coastline (2–3 km/year). Lightning strikes tend to show a positive correlation with elevation (e.g. Baker 2002; Kilinc and Beringer 2007), although this is not important in many parts of Australia due to the subdued topography. The conversion of ground-flash strikes to fires is dependent upon the duration of the strike and the condition of the vegetation (fuel) at the time.

In northern Australia, Kilinc and Beringer (2007) reported that lightning strikes were most frequent in the dry to wet monsoonal transition period, so that lightning-initiated fires were most likely in the late dry season to early wet season. In south-western Australia, McCaw and Hanstrum (2003) found that 75% of all lightning-initiated fires recorded on public lands in the eight-month fire season from September to April during the period 1995 to 2001 occurred in the two summer months of December and January, coinciding with dry fuel conditions that facilitated ignition and fire spread. Luke and McArthur (1978) stated that 80% of fires in the semi-arid grassland and woodland ecosystems of western Queensland were caused by lightning (although the evidence for this assertion is unclear), with large fires most likely following years of high rainfall. Recent, major fire events in Australia have been characterised by multiple ignitions within a single storm front: >80 fires started on the night of 7–8 January 2003 in north-eastern Victoria; four of these eventually burnt >1 million hectares of forest and farmland (Esplin *et al.* 2003). In the summer of 1960–1961, lightning started 110 fires in south-western Australia – many during the passage of a single storm near Dwellingup – which burned 146,000 ha of forest and destroyed the town (McCaw *et al.* 2003). These data indicate the significance of lightning as a source of ignitions, particularly in savanna, semi-arid rangeland and dry sclerophyll forest areas. On the other hand, high moisture levels have probably limited the role of lightning started fires in wet forests.

Globally, lightning ignition of fires may increase because of increased convection activity in a warmer world (Goldammer and Price 1998). Price and Rind (1994) found that global lightning frequencies would increase by 30–40% under a 2 x $CO_2$ climate, with increases in Australia greatest in the south-east and central-west of the continent. Human ignitions may increase as a consequence of changing patterns of settlement, especially population growth (Keeley and Fotheringham 2002).

There have been few systematic, regional or long-term treatments of the problem anywhere in the world. In the Argentine Lake District of temperate South America, Kitzberger and Veblen (2003) demonstrated a link between the number of ignitions recorded since 1938 and higher summer temperatures in Argentina. Veblen *et al.* (2008) also argued that increased temperatures in the tropical Pacific after 1976 led to both warmer summers and increased numbers of lightning-ignited fires in the Argentine Lake District.

Hence, the impact of climate change on ignitions is likely to be complex, but some increase in the frequency of occurrence of lightning could be expected.

## 4.5 Modelling the impact of climate change on fire regimes at landscape scales using the FIRESCAPE simulation model

In the preceding sections of this chapter, we explored the impacts of climate change on fire-weather and fuels in selected regions of Australia for which the appropriate data were available. A primary conclusion was that the most likely climate change scenarios – warming and drying over much of the continent – would lead to an increased risk of increased fire danger, as measured by cumulative FFDI or days on which FFDI are above critical high-end values. The scenario for fuels was less certain, because of potentially differential effects of rising temperatures, declining moisture, and $CO_2$ fertilisation on rates of accumulation and decomposition.

Here, we take these projections of change in fire danger, and assess how they might affect fire regimes by modelling the impact of climate change-induced changes to fire danger on fire regimes at a landscape scale for a defined region – the Australian Capital Territory in south-eastern Australia – where topography, plant communities and fuels vary markedly across the landscape. This study, based on the work of Dr Geoff Cary and co-authors, uses the spatially explicit fire model FIRESCAPE to analyse the effect of climate change on a critical component of fire regime – the interval between fires.

### 4.5.1 Fire-weather and fire models

Weather significantly impacts on fire behaviour (McArthur 1966; McArthur 1967; Catchpole 2002). The moisture content of fine dead fuels are governed in part by temperature, relative humidity and precipitation, while drought levels affect the moisture content of coarser fuels, deep litter beds and live vegetation. The speed and direction of wind are also critical in determining area burned directly and via their impact on the behaviour of burning fire brands that can ignite fires well ahead of a main fire front (McArthur 1967). Therefore, shifts in climate resulting from global warming are likely to result in greater area burned in a range of landscapes around the world (Cary *et al.* 2006).

In Australia, there has been significant progress in understanding the possible impacts of climate change on fire-weather (see section 4.2). This section presents an overview of some of the important advances made in modelling the impacts of climate and climate change on fire regimes. Central to this work in Australia has been the FIRESCAPE landscape fire simulation model (Cary and Banks 1999; McCarthy and Cary 2002; Cary *et al.* 2006). In this section, we examine the FIRESCAPE model and its application in a range of climate change studies. There are a number of similar models that are used internationally, and this section also includes an international model comparison of the effects of climate change on fire regimes.

### 4.5.2 The FIRESCAPE model

FIRESCAPE (Cary and Banks 1999; McCarthy and Cary 2002; Cary *et al.* 2006) simulates spatial patterns of fire regimes across vegetated landscapes. The model has been implemented and evaluated in the Australian Capital Territory region (Cary and Banks 1999; Cary 2002; Cary *et al.* 2006), south-western Tasmania (King *et al.* 2006, 2008) and Glacier National Park, Montana. Published studies of climate change and fire regimes are limited to the version of FIRESCAPE that has been implemented in the Australian Capital Territory (FIRESCAPE-ACT).

In the model, simulation of processes affecting fire regimes occurs on three temporal scales. Weather variables and ignition locations are simulated daily. When ignition occurs, spread of fire to neighbouring pixels is updated hourly using spread rates calculated from fuel load, slope and hourly weather interpolated from daily weather variables. Fuel loads are affected by fire severity and time since the most recent fire, and thus fuel dynamics effectively operate on a yearly time-step.

The probability of lightning-caused ignitions increases with: (i) increasing pixel elevation, reflecting the orographic effect of mountains on weather; and (ii) the relationship between the elevation of a pixel and the average elevation of the surrounding landscape (McCarthy and Cary 2002). These findings, developed from the data of McRae (1992), are consistent with an understanding of atmospheric electricity and are consistent with similar studies in mountainous terrain (Vankat 1983). In the simulations presented in this chapter, sequences of daily weather variables (precipitation, temperature, relative humidity and wind speed) were generated using the principles developed by Richardson (1981) such that day-to-day correlations within and between variables are maintained, and the simulated weather has the same statistical properties of the observed weather sequence. In the study of the interaction between climate, fire regimes and management in the Sydney region (section 6.4), observed sequences of weather were used directly for simulation because of specific limitations associated with the generation of stochastic weather.

Rate of spread of fire from pixel to pixel was derived from the principles of elliptical fire spread (Van Wagner 1969), Huygens' Principle (Anderson *et al.* 1982) and the fire behaviour table associate with McArthur's Forest Fire Danger Index (McArthur 1967; Noble *et al.* 1980). Variation in terrain, fuel amount and wind direction results in irregularly shaped fires. Fuel loads are simulated using Olson's (1963) model of biomass accumulation, which has been parameterised for a range of Australian systems. Fire line intensity (kW/m) (Byram 1959) is calculated for the spread of fire from one cell to the next for characterising this aspect of the fire regime and for determining whether each pixel-to-pixel spread attempt is successful or whether that part of the fire line extinguishes.

The model has been validated by comparing each aspect of the simulated fire regime against general observations from the literature (Cary 2002) and by comparing the average interval of simulated fires against the frequency of fire scars from dendrochronological data (Cary and Banks 1999).

### 4.5.3 Early analysis of climate change and fire regimes

Cary and Banks (1999) used FIRESCAPE in the Australian Capital Territory region to perform one of the earliest Australian analyses of the possible impacts of climate change on future fire regimes. Spatial pattern in fire regimes was simulated using three climate scenarios based on the then contemporary scenarios for climate change by 2030 (DEST 1994). The simulation scenarios were: current climate (Current climate); current climate with daily temperatures increased by 2°C (+ Temperature); and current climate with daily temperatures increased by 2°C and summer daily rainfall increased by 20% (+ Temperature + Rainfall). A map of cell-by cell tests of significance of differences (student's t-test) in average interval between fires was generated using the current and changed (+ Temperature + Rainfall) climates (Fig. 4.14).

Overall, the effect of the 2030 climate scenarios used in this analysis was a decrease in the average intervals between fires across the landscape. The simulated average interval between fires was predicted to be significantly lower in 2030 compared with normal climate for around half of the cells in the study region, while changes in fire interval were not significant for most of the remaining pixels.

### 4.5.4 Analysis of climate change and fire regimes for the Australian Capital Territory

Following from Cary and Banks (1999), Cary (2002) analysed the effect of a more comprehensive climate change scenario, specifically produced for the study by the CSIRO Division of Atmospheric Research, on simulated fire regimes in the Australian Capital Territory region. Daily weather in FIRESCAPE was adjusted according to the differences between 20 years of daily data simulated under 1 x $CO_2$ and 2 x $CO_2$ climates using a regional climate model (DARLAM) nested in the CSIRO9 GCM (Fig. 4.15). Sensitivity of fire regimes to three climate change scenarios (small temperature increase of 0.6°C, moderate temperature increase of 2.0°C and large temperature increase of 3.4°C) was tested. In all scenarios, other meteorological variables were scaled to represent the same fraction of the full effect of change demonstrated in Fig. 4.15 that the corresponding temperature increase represented for each scenario.

Caption: Fires in the Australian Capital Territory, 2003
Source: Steven Wilkes

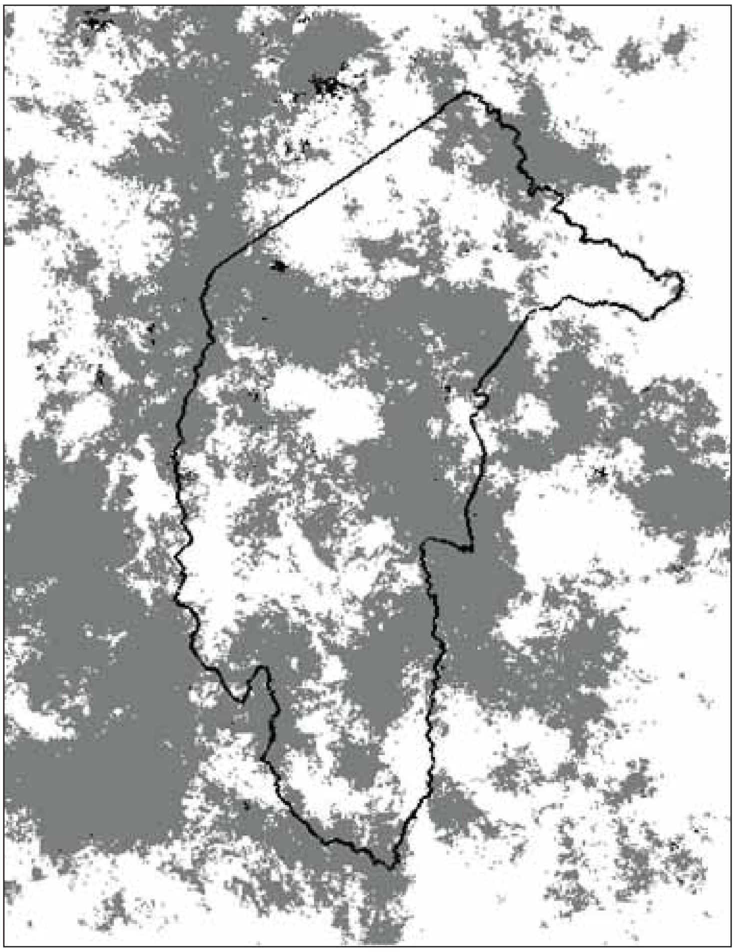


**Fig. 4.14. Spatial pattern in significance of difference between average inter-fire interval simulated for (then) current and changed (+ Temperature/+ Rainfall) climates in the Australian Capital Territory region for 2030.** Lightly shaded areas represent a statistically significant reduction in inter-fire interval by 2030. Unshaded areas represent average inter-fire intervals that are not significantly different between the scenarios. The small areas of dark shading (e.g. due north of the Australian Capital Territory; north-western corner of the Australian Capital Territory) represent statistically significant increases in average inter-fire interval (modified from Cary and Banks 1999).

A moderate change in climate (increase in daily maximum temperature of 2°C with other variables equivalently scaled) halved the simulated interval between fires across the landscape (Fig. 4.16, Fig. 4.17a). These results do not incorporate the effects of climate change (i.e. the interactive effects of rainfall, temperature and $CO_2$) on fuel dynamics and altered patterns of lightning incidence (Goldammer and Price 1998).

Cary (2002) found that the landscape-wide average of fire intensity increased by around 25% from the no-change simulation compared with the large change in climate, although the variability in intensity within each simulation was very high. The intensity effect most likely responsible for large decreases in inter-fire interval with climate change in the model system was that fires tended to extinguish less frequently under the warmer climates (Fig. 4.17b), translating into in a greater tendency for perpetuation of fire in the landscape and thus increasing the chance that fire-weather conducive to burning large areas will be encountered Cary 2002). This suggests a management action of increased fire suppression effectiveness at the point of fire ignition (see Cary 2002 for more detail).

A critical finding of Cary (2002) was a proposed relationship between changes in summary measures of fire-weather (e.g. the average annually summed Forest Fire Danger Index) and average inter-fire interval across a landscape (Fig. 4.17c). This relationship provides a critical link between analyses of the effect of climate change on fire-weather (section 4.2) and components of the fire regime that are fundamentally important for the dynamics of biodiversity (section 3).

### 4.5.5 Degree of consensus among different models: the importance of multiple models and model comparison

Single-model studies lack the generality that can only be developed from studies involving multiple models and standardised designs. Cary *et al.* (2006) undertook a multi-model evaluation of the effect of changed climate on area burned under the auspices of the Global Change in Terrestrial Ecosystems (GCTE) in Task 2.2.2 (Relationships between Global Change and Fire effects at the Landscape Scale). GCTE was a Core Project of the International Geosphere-Biosphere Programme (IGBP), an international scientific research program established in 1986 by the International Council of Scientific Unions (ICSU).

The sensitivity of simulated area burned to variation in terrain (flat, rolling and mountainous), fuel pattern (finely and coarsely clumped), climate (observed, warmer/wetter and warmer/drier) and year-to-year variation in weather was determined for four existing landscape-fire-succession models (EMBYR, FIRESCAPE, LANDSUM, SEM-LAND) and a new model implemented in LAMOS using climate from Corsica (LAMOS-DS). Temperatures in the warmer climates were 3.6°C higher than the observed climate. The warmer/wetter climate was characterised by a 20% increase in precipitation and the warmer/drier climate was characterised by a 20% decrease compared to observed.

Sensitivity was measured as the variance in area burned explained by each of the four variables, and all of the possible interactions amongst them, in a standard generalised linear modelling analysis. Modelled area burned was most sensitive to climate and variation in weather, with four models sensitive to each of these factors and three models sensitive to their interaction. Models sensitive to variation in climate were FIRESCAPE, LANDSUM, SEM-LAND and LAMOS-DS. These models generally exhibited a trend of increasing area burned from observed, through warmer and wetter, to warmer and drier climates (Fig. 4.18). Terrain and fuel pattern (average fuel load held constant), on the other hand, were found to be important for one model each.

These results are important for several reasons. First, they demonstrate that the types of model used in this study are generally more sensitive to variation in climate and inter-annual weather variability compared with terrain complexity and fuel pattern. Second, the relative effect of a warmer climate (+3.6°C) on area burned was broadly consistent across four out of the five models involved in the study (Fig. 4.18). The absolute differences in area burned between models are a result of differences in climate, vegetation and ignition rates among the locations where the models have been implemented (northern Rocky Mountains, south-eastern Australia, Central Canada and Corsica). However, in the models that responded to changes in climate, area burned increased by a considerable amount for either the warmer/wetter or warmer/drier climates, or both. Thus, there is a clear need for more international, collaborative research into model development, evaluation and comparison.

The effect of global change on a number of important processes affecting fire regimes has not been included in the simulations presented in this section because of uncertain climate change impacts on them. Price and Rind (1994) analysed the possible implications of climate change on lightning and found that global lightning frequencies would increase by 30–40% under a 2 x $CO_2$ climate, with increases in Australia greatest in the south-east and central-west of the continent. Fire regimes will be sensitive to changes in ignition frequencies of this magnitude and there is scope to update the earlier analyses on lightning distributions using more recent warming scenarios. Regarding human ignitions, Keeley and Fotheringham (2002) have demonstrated a strong link between changes in population size and numbers of fires in California, and the trend of increasing numbers of fires in the Australian Capital Territory region may be similarly linked to population growth (section 6.2.1; Case study on alpine ash forests), which would need to be included in modelling to fully understand fire regimes of the future.

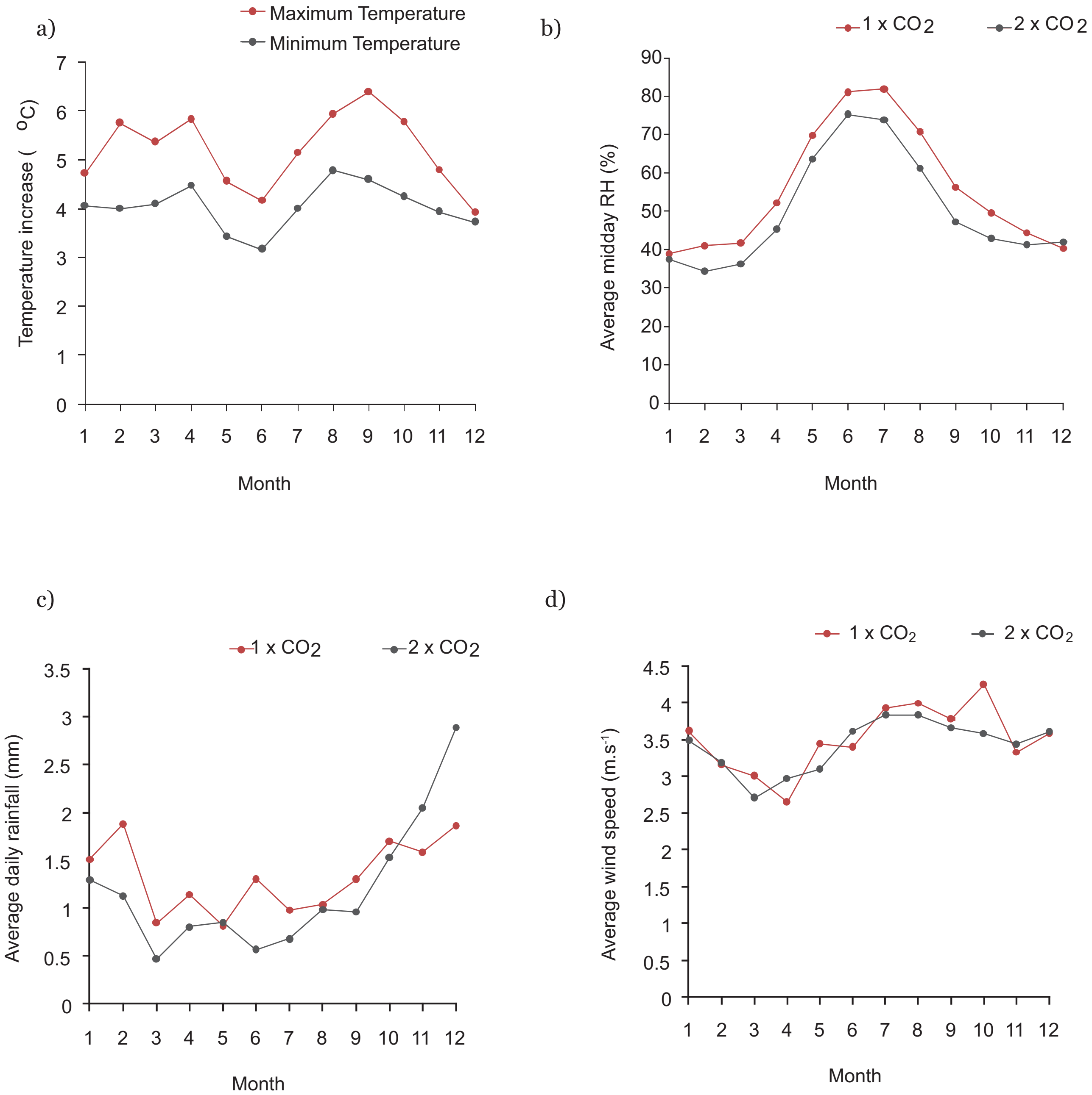



**Fig. 4.15. Climatic averages for the Australian Capital Territory climate change case study region from 1 × $CO_2$ and 2 × $CO_2$ equilibrium experiments.** (CSIRO DARLAM regional climate model nested in the CSIRO9 GCM). (a) average daily maximum and minimum temperature increase; (b) daily averages mid-day relative humidity (RH) (%); (c) average daily rainfall (mm); and (d) average daily wind speed (m/sec). Climate sensitivity for the 2 × $CO_2$ simulation was 4.338°C. Source: Cary (2002).

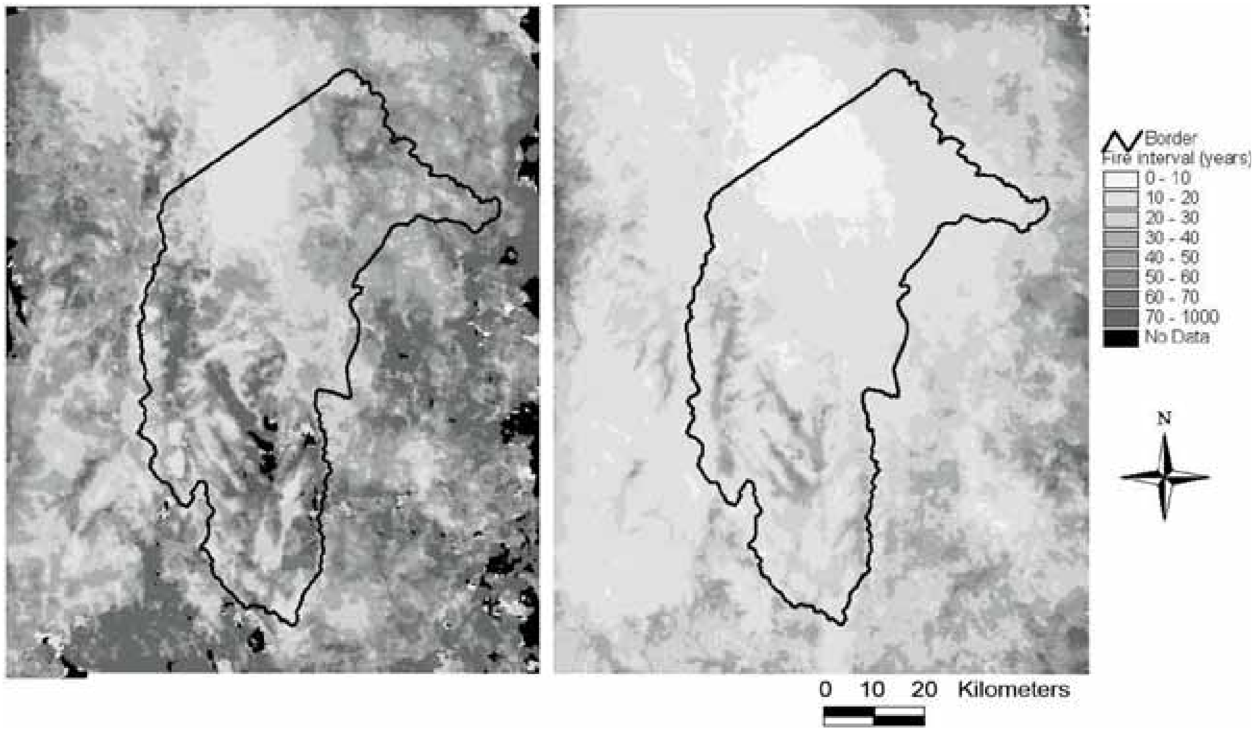



**Fig. 4.16. Effect of climate change on the spatial patterns of average inter-fire interval (IFI in years) predicted from the FIRESCAPE model for the Australian Capital Territory region.** Scenarios presented are for present climate (left) and a moderate change in climate (right). Modified from Cary (2002).

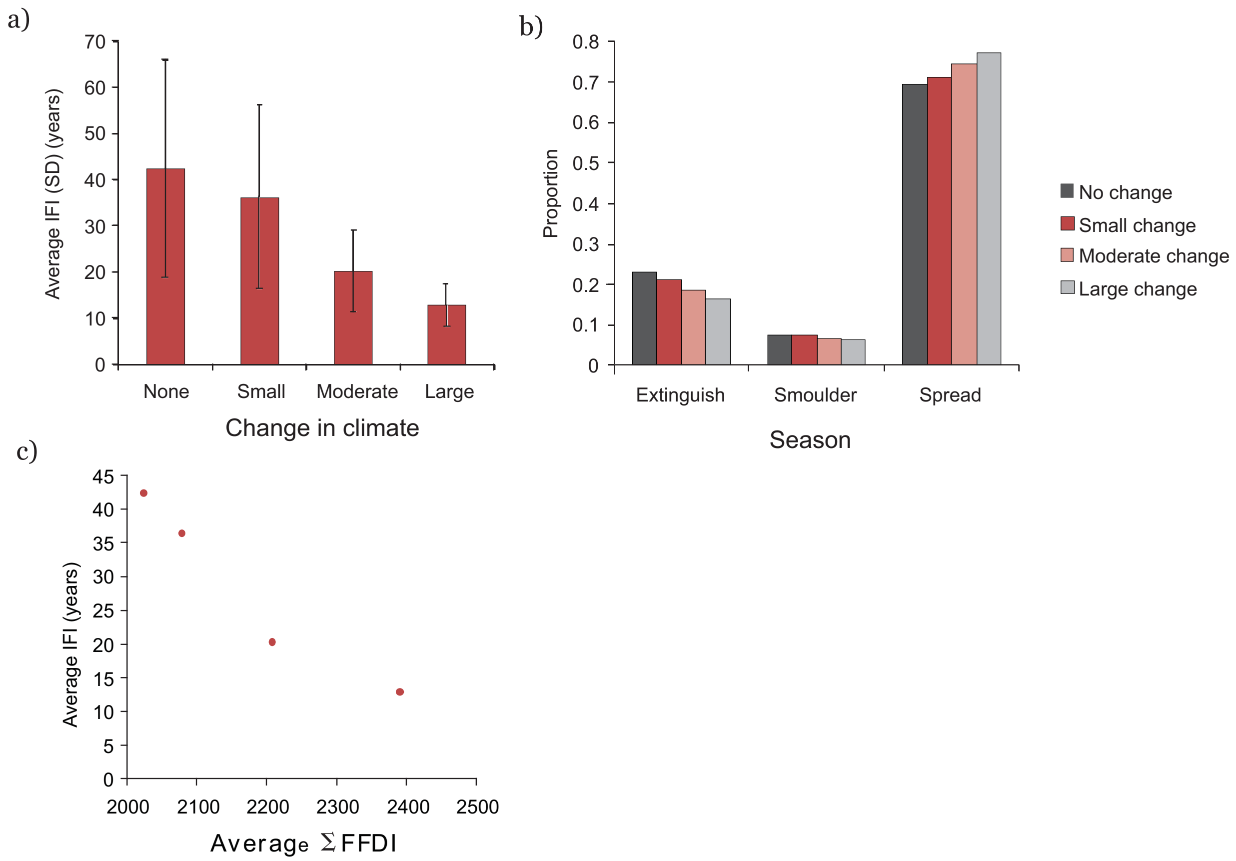



**Fig. 4.17. Fire intervals and climate change in the Australian Capital Territory.** (a) landscape average inter-fire interval versus climate change scenario; (b) landscape average proportion of pixel-to-pixel spread events that extinguish, smoulder and spread for each climate change scenario; and (c) relationship between average inter-fire interval and ΣFFDI for the full range of climate change scenarios for the Australian Capital Territory region. Source: Cary (2002).

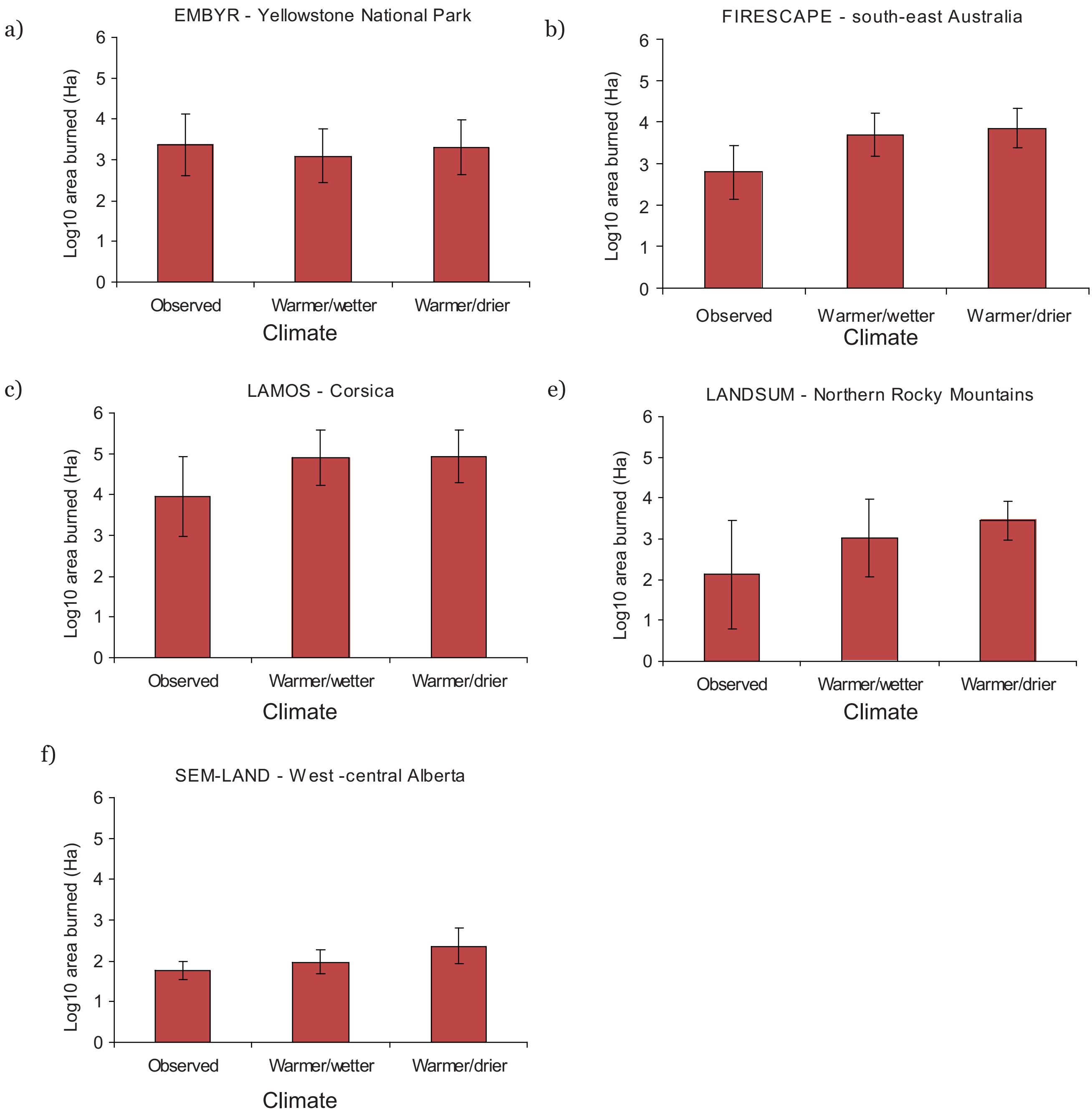



**Fig. 4.18. Modelled effect of climate change on area burned.** Area burned under three climate scenarios in the GCTE comparison of landscape-fire models (Cary *et al.* 2006): (a) EMBYR, Yellowstone National Park; (b) FIRESCAPE – south-eastern Australia; (c) LAMOS(HS) – Corsica; (d) LANDSUM – northern Rocky Mountains; and (e) SEM-LAND – west-central Alberta.

### 4.5.6 Other important processes and uncertainties

The extent to which fuel dynamics are affected by changes in vegetation productivity under future climates is far from being resolved. Increased $CO_2$ may result in increased productivity of litter and lower ratios of nitrogen to carbon resulting in slower litter decomposition. Shifts in climate might also affect production and decomposition of litter directly. For example, Wood (1970) demonstrated that decomposition rates of litter in moister montane and alpine sites were higher than corresponding drier locations. The topic is also explored in Box 4.1.

Finally, there is considerable uncertainty associated with modelling fire spread over a larger range of vegetation types in Australia under both current and future climates. The FIRESCAPE models presented in this section invoke McArthur's (1967) fire spread algorithms, which are known to under-predict rates of spread under high wind speeds and in shrubby fuels along with some other circumstances (Jim Gould, CSIRO, pers. comm.). A new model of fire spread has recently been produced by Gould *et al.* (2007) and its effect on modelled fire regimes is unknown. A foreseeable difficulty in implementing the new model is that FIRESCAPE does not simulate dynamics of the various fuel characteristics required to run the new spread model. Further, an ever-increasing suite of fire behaviour models designed for specific vegetation types is being produced (e.g. Catchpole *et al.* 1998 for shrublands). However, models for moist forest, particularly in south-eastern Australia, are largely lacking.

## 4.6 Uncertainty in projections: implications and importance

A consistent signal from our analyses of potential impacts of climate change on fire regimes is that climate change will lead to an increased likelihood of severe fire-weather. Pitman *et al.* (2007) also concluded that that the likelihood of a significant increase in fire risk over Australia resulting from climate change is very high.

Model predictions presented in section 4.5 indicated that this is likely to lead to increased fire intensity, a greater area burnt and reduced interval between fires. Models generally behaved in a similar manner to a given climate change scenario.

However, the impacts of climate change on fuels are subject to larger uncertainty than the impact of climate change on fire-weather. There is the potential for more fuel as a consequence of $CO_2$ fertilisation, or the prospect of less fuel due to decreasing moisture availability. Thus there is the potential for the weather, moisture and $CO_2$ elements of climate change to act on fire regimes in a congruent manner, pushing fire regimes in same direction. Alternatively, there is the potential for these elements to operate in an antagonistic manner with declining moisture, through its effect on fuels, thus mitigating the effects of more severe fire-weather. If the elements act in tandem, then the primary question is about the magnitude of change; if they act antagonistically, then both magnitude and direction of change are uncertain.

# 5. OVERVIEW OF THE POTENTIAL EFFECTS OF CLIMATE CHANGE ON AUSTRALIAN FIRE REGIMES: A 'FOUR-SWITCH' MODEL OF THE BIOGEOGRAPHY OF VARIATION IN AUSTRALIAN FIRE REGIMES

In the previous section we examined the impact of climate change on two primary drivers of fire – fire-weather and fuel. We also presented an application of a fire model that assessed sensitivity of a component of regime (in that case, interval) to climate change-induced changes in fire-weather. There is, to our knowledge, no landscape fire model that can assess climate change impacts on weather, fuel and $CO_2$, and produce outputs in terms of other components of regime (e.g. intensity) and biodiversity. The brief of the project is, however, to examine this very problem from a national perspective and report on prospects for achieving this.

To do this, two knowledge platforms are needed: (A) a national overview of fire regimes and the causes of their variation, which explains the current spatial pattern of regimes and is robust enough to accommodate climate change impacts; and (B) regional, spatially explicit fire models (driven by daily weather variables and appropriate fuel accumulation models) that are linked to spatially explicit vegetation dynamic models (driven by species abundance and life history, and disturbance-response attributes).

To this end, we present a conceptual framework to identify and evaluate sensitivities of different fire regimes to different drivers of change. It is national in approach, but naturally, not comprehensive – this sort of exercise cannot be done for every sub-region of Australia (e.g. the simplified IBRA regions used by Dunlop and Brown 2008 in their analyses). However, this is a necessary first step to allow climate change, fire regime and biodiversity interactions to be scrutinised. This will provide the basis for the subsequent research to assist the biodiversity research and management community.

We do this by presenting a continental-scale view of why fire regimes vary, by identifying what are the principal drivers and constraints of regimes, and how climate change is likely to affect four key determinants of regime: (i) fuel mass and amount; (ii) fuel moisture and its availability to burn; (iii) fire-weather and the potential for fire spread; and (iv) ignition. Each can be thought of as a switch, with the rates of turning 'on' and 'off' determining fire regime. In this section, we explore some of these interactions for selected areas of Australia, which have vastly different fire regimes, as a means of addressing (A) above. Ways of approaching (B) above will be outlined in section 6.

## 5.1 Introduction

Fire regimes in Australia currently vary widely across differing ecosystems and climatic conditions (Bradstock *et al.* 2002). This variation reflects the interplay of key drivers of fire, namely: weather, fuel and ignitions. Globally, there is general uncertainty about the relative importance of these drivers and underlying influences of land management in shaping fire regimes (Westerling *et al.* 2006; Marlon *et al.* 2008).

Studies on these drivers and resultant variations in fire regimes in Australia vary in context and scope, though recent continental-scale analysis (Russell-Smith *et al.* 2007) indicated that fire activity is strongly related to rainfall variations and land use. Climate change can potentially alter each of the primary drivers of fire, and an integrated understanding of causes of fire regime variation within Australia is therefore required to predict changes in the future. Potential methods for predicting quantitative changes to fire in the future vary (e.g. statistical and simulation models), and the scope of application of these methods is constrained by data and ease of application of relevant models (see above and Cary 2002).

As an initial step towards developing such predictive capacity, we present in this section a conceptual model of the drivers of fire regimes, using case studies that represent a national spectrum. The aim is to:

- illustrate that variations in fire regimes across all ecosystems reflect systematic variation in rates of a common set of key processes, rather than idiosyncratic local effects
- predict whether climate change will have equal effects on fire regimes throughout Australia, and if not, where highest levels of change may be anticipated
- predict consequences of other global change effects.

## 5.2 A biophysical basis for predicting variation in fire regimes: the 'four-switch' model

Extremes of moisture availability govern the growth/accumulation of biomass/fuel (wetness) on the one hand and its ability to burn (dryness) on the other (Bond and Keeley 2005; Pausas and Bradstock 2007). Fire is therefore related to moisture in a non-linear way (Fig. 5.1). Effects of dryness are unimportant without effects of periods of high moisture and resultant plant growth. Temporal fluctuations in moisture availability will therefore influence fire frequency. Characteristic patterns of mean and variance in moisture and fire regimes in Australia may explain fire regime variation in Australia (Gill *et al.* 2002). Effects of moisture variation on selection of key plant functional types ('fuel species' *sensu* Gill *et al.* 2002) may play a key role in this regard.

Large fires are known to account for the bulk of area burned over time in many ecosystems (Boer *et al.* 2008; Cui *et al.* 2008). Significant area burned, via large fires, can only occur when available fuel – fuel that is sufficiently dry that it is available to burn – is spread across large areas. Connectivity of available fuel will be important due to the contagious nature of fire spread (Peters *et al.* 2004; Pueyo 2007). Landscape- and regional-scale patterns of moisture fluctuation will therefore have strong effects on area burned.

Temperature, wind speed and humidity directly influence the rate of spread of fires and the degree to which fires can bridge fuel discontinuities (Catchpole 2002; Peters *et al.* 2004; Boer *et al.* 2008). Area burned will be a function of these 'fire-weather' variables, as recognised in various empirical and physical models of fire behaviour and fire danger rating systems (Catchpole 2002). Climate affects the frequency of lightning and ensuing probability of ignition (e.g. Price and Rind 1994).

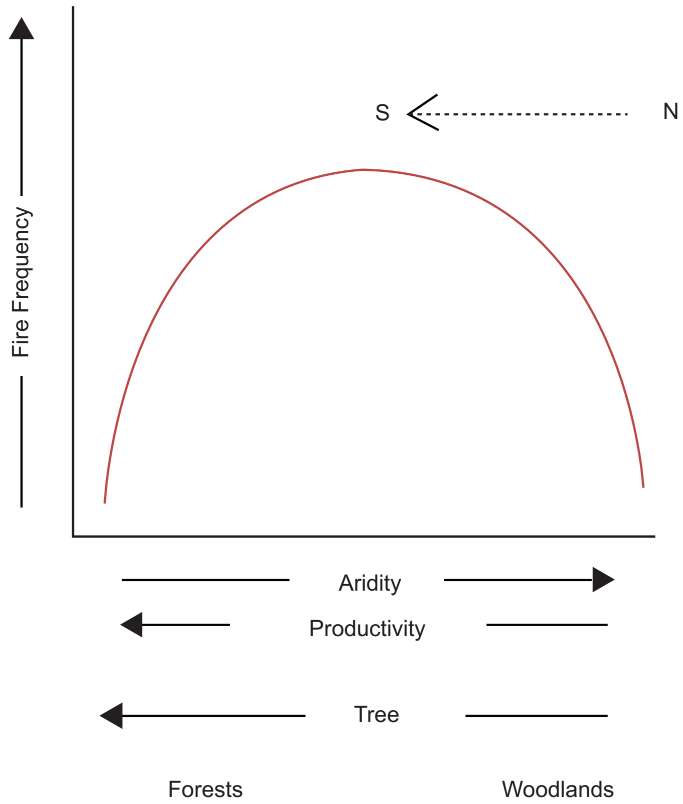



**Fig. 5.1. Potential relationships between moisture, productivity and fire frequency.** (After Bond and Keeley 2005; Pausas and Bradstock 2007). Trends in tree cover are indicated.

Biomass production, its availability to burn (as determined by moisture content), fire-weather and ignition represent a hierarchy of conditional processes governing fire. They can be regarded as four hypothetical 'switches' that must be activated simultaneously for fire to occur. Area burned will be a direct function of the spatial scale (i.e. the contiguous area) over which they jointly prevail. Should any switch be 'off' in a particular locality, fire will not occur. Fire can therefore be constrained in differing ways, i.e. through the effect of turning different switches off. Some switches are 'off' more often than others in different ecosystems, resulting in fundamentally different fire regimes. Identification of the 'limiting' switch (i.e. the switch that is activated least often) and its relationship to differing ecosystem characteristics is the key to understanding these variations.

Hypothetically, ecosystems can be arranged in multidimensional space (Fig. 5.2), with each switch represented as a temporal axis (i.e. the mean time interval between triggering). The rate of switching in each case will be a function of interactions between climate (mean and variability), soils and plant functional types characteristic of differing ecosystems. Diverse bioregions in Australia are the outcome varied combinations of these factors (Hutchinson *et al.* 2005).

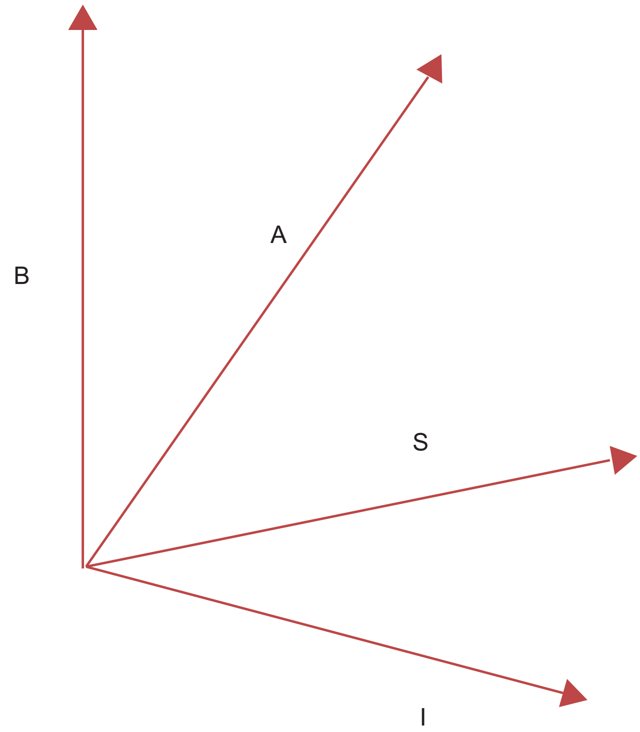



**Fig. 5.2. Temporal variations in biomass growth (B), availability for burning (A), ambient weather conducive to fire spread (S) and ignitions (I) via hypothetical axes.** Each axis defines variations in length of recurrence interval for conditions conducive to large fires. Ecosystems will occupy differing positions in the multidimensional space defined by these axes, according to inherent biophysical attributes.

### 5.2.1 Biomass (B)

Fire behaviour models in Australia (Catchpole 2002) indicate that surface or near-surface fuels (primarily dead leaf material) are critical for the spread of fire in most vegetation types. Fuel quantity will therefore be affected by foliar cover (Walker 1981), life form and leaf attributes. A primary distinction can be made between herbaceous and woody plants because the characteristics, rate of supply and arrangement of material derived from these life forms is different.

Tree cover will be a determinant of the type of surface fuel. Areas of low tree cover in Australia (e.g. woodlands with <30 % overstorey foliar cover) are vast (Beadle 1981; Groves 1994). Tree cover is positively correlated with patterns of available moisture that arise from climate and soil interactions (Beadle 1981; Williams *et al.* 1996; Specht and Specht 1999; Fensham *et al.* 2005), interactions with soil depth/texture (e.g. Beadle 1981; Specht and Moll 1983; Bowman and Prior 2005; ). Fire regimes may also reduce tree cover below edaphic/climatic potential (Leidloff and Cook 2007).

Eucalypt open forests (i.e. canopy cover >30%) (Gill 1994; Specht and Specht 1999) occupy areas with relatively high moisture availability (Fig. 5.1). Litter from trees, particularly *Eucalyptus* and other closely related genera, forms the principal surface fuel (Walker 1981; Raison *et al.* 1983). Variable contributions from shrubs and herbs augment this tree litter. Fuel quantity and extent is therefore a function of the turnover of perennial foliage – with major variations in cover and load of surface fuel driven primarily by the degree of tree and shrub cover (a function of moisture availability), and time since last fire (Walker 1981; Specht and Specht 1999; Berry and Roderick 2002).

In woodlands, connectivity of surface fuels is provided by herbaceous material (e.g. grasses and herbs) (Hodgkinson 2002; Prober *et al.* 2007, 2008). Woodlands (10–30% tree cover; Specht and Specht 1999) are common in drier environments (Fig. 5.1). There is interplay between woody litter (trees) and herbaceous fuels in woodlands. Litter fuel from trees is typically discontinuous because of relatively low cover, so that overall fuel availability is strongly influenced by fluctuations in herbaceous biomass.

### 5.2.2 Availability to burn (A)

In forested systems dominated by woody fuels, drought alters the moisture status of compacted litter beds derived from woody plants and thus availability to burn in horizontal (e.g. differing aspects) and vertical (e.g. within litter bed) planes. This changes the propensity for large fires to develop by altering connectivity of fuels across landscapes. Droughts may also temporarily increase litter fall from trees and shrubs (e.g. Pook *et al.* 1997), and are major factors governing fire activity in southern, forested regions of Australia (Gill 1984). By contrast, in ecosystems dominated by grass and herb fuels, dry periods initiate rapid leaf death (curing) of perennial and ephemeral species (e.g. Spessa *et al.* 2005).

Ecosystems in the northern, monsoonal tropics experience prolonged annual wet and dry seasons; whereas those in the southern, temperate region may experience severe drought on a multi-decadal cycle, influenced by El Niño oscillations (Cullen and Grierson 2008). Pronounced water deficits are a seasonal feature of arid and semi-arid environments (Hutchinson *et al.* 2005).

### 5.2.3 Fire spread (S)

Severe ambient weather conditions (high temperatures, high wind speeds and low humidity) result in the rapid spread of fires, irrespective of fuel type (Catchpole 2002). Ease of ignition and flame transfer is increased by high temperatures and low humidity. Wind directly affects flame length and depth, and propagation of fire via embers. The area burned will therefore be directly influenced by these variables (Gill *et al.* 2002; Boer *et al.* 2008). Fire-weather is a function of latitude and rainfall. Annual incidence of severe fire danger conditions tends to decline with increasing rainfall and latitude (Gill and Moore 1990; Williams *et al.* 2001 ; McCaw and Hanstrum 2003; Lucas *et al.* 2007).

### 5.2.4 Ignition (I)

Definitive data on ignitions from lightning versus anthropogenic sources is elusive, due to remoteness of many regions and difficulty of attribution to sources (e.g. McCaw and Hanstrum 2003). Many official records indicate large proportions of ignitions of unknown origin. Lightning ignition rates (I) generally vary from being common annually in the tropics to rare in the cool temperate (Kuleshov *et al.* 2005). Anthropogenic ignitions are a function of population density and land use, with high rates near urban development (Gill and Williams 1996; Bradstock and Gill 2001) and low rates in dry pastoral/agricultural landscapes (Noble and Grice 2002; Russell-Smith *et al.* 2007).

## 5.3 Contemporary fire regimes in Australia: patterns of limitation

Potential influences of fuel, fire-weather and ignitions on fire regimes are coupled. Grass/herbaceous fuels will be important where moisture deficits (low to moderate tree cover) and high levels of fire danger are severe and regular (Fig. 5.1). Woody, litter fuels (in forests/woodlands with moderate to high tree cover) occur where such effects are less severe and regular (Fig. 5.1) or where trees can access other sources of water (e.g. tropical savannas; Bowman and Prior 2005). Growth and curing of grasses and herbs (perennial and ephemeral) respond rapidly to fluctuations in available moisture. Species of $C_4$ grasses may be highly influential in this regard, due to their productivity under pronounced seasonal conditions (Osborne 2008). As a result, fire regimes in woodlands dominated by grass/herbaceous fuels will be governed by fluctuations in biomass growth (B). Moisture variation will therefore strongly influence fire regimes.

By contrast, in forested systems where woody, litter fuels are prominent, regular patterns of fuel dynamics occur, due mainly to re-accumulation after fire. Fuel amount (B) is less tightly coupled to moisture variation. In the absence of fire, sufficient litter is usually present for propagation of fire (i.e. fuel amount or mass is non-limiting). Fire will therefore be limited primarily by fluctuations in fuel availability (A) and propagation potential (S), governed respectively by drought and ambient weather at the time of a fire.

Ecosystems can be arranged using a simplified model (Fig. 5.3), with biomass growth (switch B) on one axis, and a combination of biomass availability (switch A) and fire spread potential as a function of ambient weather (S) on the other. This combines these separate influences in a way that is consistent with fire danger rating systems (e.g. McArthur Forest Fire Danger Index; Luke and McArthur 1978) which contain both drought and ambient weather components. Thus high values of fire danger indices, conducive to rapid rate of spread and large area burned, are improbable without both severe drought and ambient weather.

This approach contrasts the sensitivity of fire regimes to relative effects of biomass growth (B) on the one hand and fire danger (A + S) on the other. Predictions of potential fire regimes (i.e. the limits of average fire frequency) can be made, with realisation of potential being dependent on the rate of ignition (i.e. the fourth switch: Fig. 5.1). Comparisons with measurements of area burned, fire return interval and other fire regimes characteristics can be used to evaluate these predictions.

Fire regime potential was explored for five biomes (Fig. 5.3) indicative of a gradient of tree cover spanning high to low moisture (Fig 5.1, Fig. 5.3) and thus a balance between woody litter and herbaceous fuels. The biomes (Table 5.1, Fig. 5.3) are: tropical open forest (TF); arid woodlands (AW); temperate grassy woodlands (TGW); dry sclerophyll shrubby forests (DSF); and wet sclerophyll forests (WSF). These biomes represent gradients of latitude, climate (monsoonal tropics to cool temperate) and available moisture. Potential fire regimes in forested communities (tree cover >30%) tend to be limited primarily by occurrence of severe fire-weather. The degree of limitation increases as a function of increasing moisture (Fig. 5.3). By contrast, potential fire regimes in woodland communities tend to be limited primarily by biomass growth. Limitation increases with decreasing moisture (Fig. 5.3). Communities with 'intermediate' tree cover (about 30%) may be particularly susceptible to factors that alter the balance between woody overstorey and grass/herb understorey (e.g. a change from limitation by B to A or S, or vice versa).

The biome studies also represent the influence of particular plant species/functional types and land use context (e.g. pastoral to urban hinterlands). Examples are discussed using relevant data from typical regions in each instance (Table 5.1).

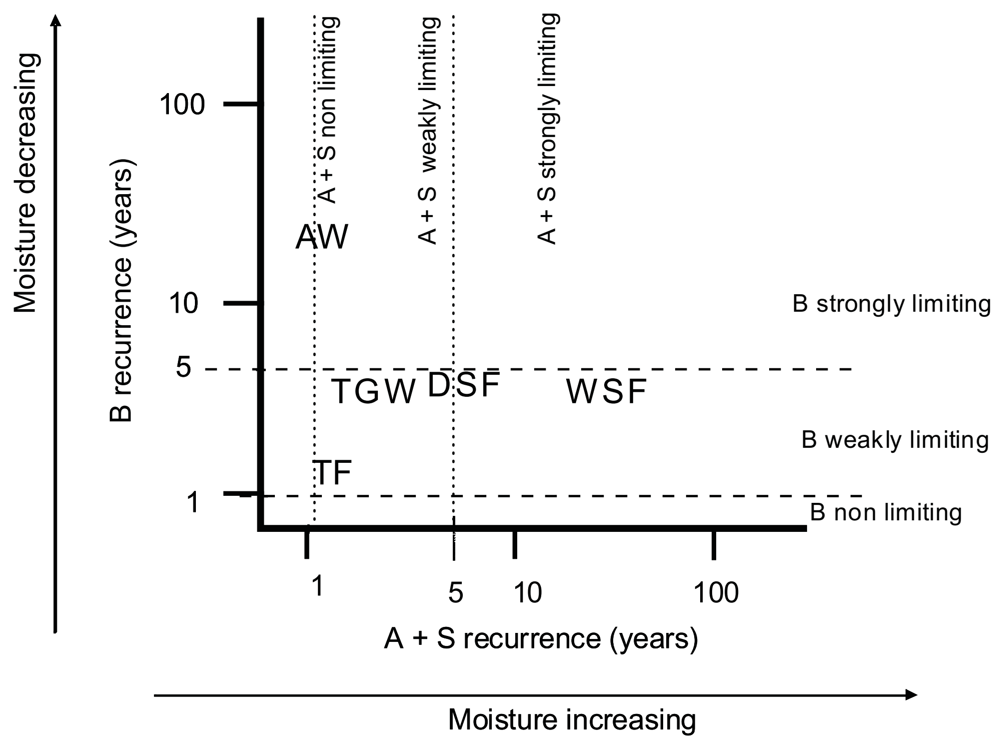



**Fig. 5.3. Temporal recurrence of factors (biomass growth B; availability A; ambient fire-weather S) governing potential for large fires in different forest and woodland biomes.** Tropical open forest (TF), temperate dry sclerophyll forest (DSF), cool temperate wet sclerophyll forest (WSF), temperate grassy woodland (TGW) and arid woodland (AW). Domains of limitation by B and A + S are represented by dashed and dotted lines, respectively.

### 5.3.1 Potential fire regimes

TF represents grassy, open forests (i.e. tree foliar cover >30%; Table 5.1) that occur in high-rainfall, near-coastal regions of Northern Australia. These are the mesic variant of tropical savanna forests and woodlands (Williams *et al.* 2002) where moisture availability is sufficient to support relatively high tree cover, consisting of a mix of evergreen, deciduous and semi-deciduous species. TF represents an extreme where the influences of both available biomass (B) and fire danger (A+S) converge (Fig. 5.3) so that neither influence is strongly limiting (Gill *et al.* 2002). That is, frequency of occurrence of conditions conducive to major fires is close to annual, due to the effects of both regular annual prolonged wet season and dry seasons on grass/herb fuels under a monsoonal climate. The wet season provides a concentrated moist period for rapid growth of grasses, while the dry season provides prolonged annual drought (A) and lengthy sequences of days with fire-weather (S) suitable for the rapid spread of large, high-intensity fires (Table 5.1; Williams *et al.* 2002). Annual grasses such as *Saga* spp. (= sorghum; Williams *et al.* 2002) are suited to these conditions, and therefore have potential to control fire regimes and effectively usurp the influence of litter accumulation from trees.

In contrast to TF, both AW and TGW represent differing degrees of temporal limitation in herbaceous biomass growth (Table 5.1, Fig. 5.3). As in TF, severe fire-weather in AW and TGW is non-limiting (Table 5.1, Fig. 5.3).

Variants of AW cover a vast expanse of the continent. The ground layer consists of perennial hummock grasses (e.g. *Triodia* and *Plectrachne* spp.) or tussock grasses (e.g. *Aristida* and *Eragrostis* spp.), and ephemeral herbs fuels are common (Allan and Southgate 2002; Hodgkinson 2002; Southgate and Carthew 2007). The tree layer is commonly dominated by *Acacia* spp. (Hodgkinson 2002). Herbaceous cover (perennial and ephemeral) can be sparse in dry times, but also grows vigorously in response to rain (Griffin *et al.* 1983; Allan and Southgate 2002). Time since fire, in interaction with rainfall, also affects herbage cover, although these effects can be negated after high rainfall (Gill 2000; Allan and Southgate 2002). High-rainfall events therefore irregularly provide fuel connectivity by stimulating herbage that is otherwise absent or slow to develop (Gill 2000; Allan and Southgate 2002; Hodgkinson 2002). Such events occur at decadal to multi-decadal recurrence intervals with consequent effects on fire regime potential.

Tussock Spinifex grasses in the Gawler Ranges National Park, South Australia.
Source: Department of the Environment, Water, Heritage and the Arts

TGW is intermediate between TS and AW, reflecting moisture status in deep, well-drained soils on the margins of inland, semi-arid environments (Beadle 1981; Table 5.1). Perennial and ephemeral grasses/herbs are prominent, and *Eucalyptus*, *Callitris* and *Allocasuarina* spp. are common as trees. Availability of herbage may fluctuate with level of stocking of domestic grazing animals and occasional prolonged drought (Noble and Grice 2002; Prober *et al.* 2007). Recovery after fire is relatively rapid (e.g. 1–5 years), depending on fluctuations in rainfall (Prober *et al.* 2007, 2008). Potential for fire is therefore high (e.g. 1–5 year frequency; Fig. 5.3), because temporal biomass limitation is relatively weak (i.e. biomass fluctuations are rapid and relatively regular). The balance between C3 (e.g. *Poa* sp.) and C4 (e.g. *Themeda* sp.) grasses in these systems may be important (Prober *et al.* 2007). Biomass and spatial cover may decline if C4 abundance declines.

In comparison to TF (biomass non-limiting), biomass in the temperate forest examples (DSF, WSF) is weakly limiting due to post-fire recovery (DSF, WSF). There is sufficient surface litter for propagation of fire under a broad range of weather conditions at about five years since last fire (Raison *et al.* 1983; Morrison *et al.* 1996). Quasi-equilibrium surface fuel loads are reached after about 10 years since fire (Raison *et al.* 1983; Walker 1981). Fire potential is therefore tends to be limited by fire-weather (Fig. 5.3).

In DSF, days of extreme fire danger (i.e. FFDI>49) are relatively rare (e.g. average of one day per annum) in mainland south-eastern and south-western regions (Gill and Moore 1990; McCaw and Hanstrum 2003; Lucas *et al.* 2007). In south-eastern DSF, characterised by non-seasonal rainfall, regular ENSO-related droughts (e.g. five-year frequency) may provide severe fire danger conditions suitable for major fires (Verdon *et al.* 2004; Nicholls and Lucas 2007; Table 5.1). South-western DSF occurs in a Mediterranean climate, with a lengthy annual summer dry period. Severe fire danger episodes are generated regularly (e.g. five-year frequency) by strong cyclonic activity to the north (McCaw and Hanstrum 2003; Table 5.1). Extreme fire danger is rarer in southern Tasmania (Lucas *et al.* 2007; Table 5.1), though a positive association between summer dryness and area burned prevails in western WSF regions of the island (Nicholls and Lucas 2007). Fire potential is therefore relatively regular in mainland DSF (5–10 year intervals; Fig. 5.3) and less regular (>10 years; Fig. 5.3) in western Tasmania (WSF).

### 5.3.2 Realisation of fire regime potential

Contemporary ignition rates in TF are presently high, due mainly to anthropogenic sources, resulting in high frequency fire regimes (Table 5.1). Fire regimes are therefore close to their potential limit (Fig. 5.3). Variations in summer rain may affect the quantity of grass fuels and length of the dry season, with high rainfall leading to greater area burned (Harris *et al.* 2008). In AW, ignition rates (anthropogenic and lightning) also appear to be sufficient to saturate opportunities where herbaceous fuel after curing is non-limiting, leading to episodes of major fire over vast areas (Heydon *et al.* 2000; Allan and Southgate 2002; Russell-Smith *et al.* 2007).

By contrast in TGW, fire is often constrained through low anthropogenic ignitions and a higher level of availability of fire suppression due to higher population density (e.g. agricultural holdings and small towns), leading to long-term absence of burning in natural fragments (Table 5.1). In larger, non-fragmented tracts of temperate woodlands, multi-decadal fire regimes predominate under the influence of lightning (Table 5.1). Current fire recurrence intervals are therefore considerably longer than their potential minimum (cf. Fig. 5.3). High domestic stocking in many regions has degraded perennial grass populations (e.g. Noble 1997; Hobbs 2002). Replacement by ephemeral grasses, herbage and woody plants has resulted in diminished fuel levels, enhancing the sensitivity of fire potential to rainfall variations, and contributing to a decrease in ignitions and area burned (Noble and Grice 2002).

In DSF, ignitions are highly varied (Table 5.1). In some areas of eastern New South Wales, for example, anthropogenic ignition rates (sometimes complemented by lightning) are sufficient to exploit most opportunities for major fires, resulting in 5–10 year fire recurrence intervals at landscape scales (e.g. Bradstock and Kenny 2003). In more remote regions with low rates of anthropogenic ignition, average inter-fire intervals may be >20 years under the predominant influence of lightning (Bradstock 2008). Similar trends apply in other regions such as south-western Australia (Table 5.1). Fulfilment of fire regime potential is therefore variable (Table 5.1, Fig. 5.3).

Table 5.1. Attributes of biomes that affect fire regimes in differing Australian ecosystems.

| | Tropical open forest (TF) | Temperate grassy woodlands (TGW) | Arid woodlands (AW) | Temperate dry sclerophyll forests (DSF) | Cool temperate wet sclerophyll forests (WSF) |
|---|---|---|---|---|---|
| Indicative bioregion and agro-climatic zone (Hutchinson *et al.* 2005) | Arnhem Coast; Central Arnhem; Hot, seasonally wet/dry (I1) | New South Wales south-western slopes and Brigalow Belt South. Warm, seasonally wet/dry. Long, hot summers and mild winters with significant moisture limits on growth including Mediterranean climates (summer dry season, e.g. Adelaide E1) and mid-latitude eastern continental climates with wetter summers and drier winters (e.g. central New South Wales E3). | MacDonnell Ranges Desert – high water limitation (G) | Sydney Basin – Warm, wet (F3). Long, hot summers and mild winters.<br>Jarrah forest, Mediterranean climate (summer dry season, e.g. Perth E1) | Tasmanian west, Cold (B2) Very cold winters with short warm summers. |
| Contemporary land uses | Pastoralism, conservation reserves, Indigenous | Winter cereals and summer crops, grazing | Pastoralism, conservation, Indigenous | Urban, conservation reserves, water catchments, forestry (south-west), mining (south-west) | Forestry, hydroelectricity, conservation reserves |
| Mean annual rainfall (mm – station, Australian Bureau of Meteorology) | 1539 (Darwin post office) | 585 (Dubbo, Darling St)<br>529 (Adelaide, West Terrace) | 279 (Alice Springs post office) | 1214 (Sydney, Observatory Hill)<br>868 (Perth, regional) | 1212 (Maydena) |
| Rainfall seasonality (Months of highest and lowest mean rainfall – mm, Australian Bureau of Meteorology) | Predominantly summer (393 January, 1 July – Darwin). | Even seasonal distribution (63 January, 43 September – Dubbo)<br>Predominantly winter (72 June, 20 January – Adelaide) | Summer peak (43 January,<br>9 September – Alice Springs) | Late summer – early winter peak (131 June, 69 September – Sydney)<br>Predominantly winter (182 June,<br>9 January – Perth) | Winter – early summer peak (128 August, 60 February -Maydena) |
| Canopy tree cover/ composition | >30%, *Eucalyptus, Terminalia* | 10–30% *Eucalyptus, Casuarina, Callitris* | <10–30% *Acacia* | 30–70%, *Eucalyptus, Angophora, Allocasuarina* | *Eucalyptus, Nothofagus, Atherosperma* |

Table 5.1. Attributes of biomes that affect fire regimes in differing Australian ecosystems. (continued)

| | Tropical open forest (TF) | Temperate grassy woodlands (TGW) | Arid woodlands (AW) | Temperate dry sclerophyll forests (DSF) | Cool temperate wet sclerophyll forests (WSF) |
|---|---|---|---|---|---|
| Understorey structure/ composition | Annual (e.g. *Saga*) and perennial grasses | Perennial (e.g. *Themeda*) and annual grasses, herbs and shrubs | Hummock grasses (e.g. *Triodia*), tussock grasses (*Astrebla*), herbs | Diverse hrubs (e.g. Proteaceae, Fabaceae, Myrtaceae), sedges (e.g. Cyperaceae), graminoids | Shrubs (sclerophyll and mesic) |
| Average number of very high to extreme fire-weather days per annum | >30 (Jabiru) Williams *et al.* (2002) | 23 (Dubbo), 18.3 (Adelaide) Lucas *et al.* (2007) | >60 (Alice Springs) Williams *et al.* (2001) | 7.6 (Sydney), Lucas *et al.* (2007)<br>>10 (Perth), Gill and Moore (1990) | 2 (Hobart) Lucas *et al.* (2007) |
| Fire season | Winter – spring | Summer – autumn | Spring – summer | Spring – early summer (south-east<br>Summer – autumn (south-west)) | Late summer - autumn |
| Contemporary ignition sources | Predominantly human | Predominantly lightning | Human and lightning | Predominantly human, variable lightning | Predominantly lightning |
| Contemporary range of inter-fire intervals | 2–5 years, Russell-Smith *et al.* (1997, 2007), Williams *et al.* (2002) | >10 year, often multi-decadal in length Hobbs (2002), Lunt *et al.* (2006) | 10 – 80 years (Allan and Southgate 2002) | 5–15 years near urban areas, 10–25 years in remote areas (south-east) Bradstock and Gill 2001, Gill and Moore (1996), Lamont *et al.* (2003). | > 20 years ,Gill and Catling (2002) |

In WSF, lightning ignition rates and anthropogenic ignition rates are relatively low (e.g. King *et al.* 2006; Table 5.1). These may be insufficient to saturate opportunities for major fires created by irregular summer drought, resulting in fire regimes of multi-decadal to century-scale frequency (Table 5.1, Fig. 5.3). Limitation of fire regimes by ignition (I) may generally be more common with increasing moisture in forested communities. The degree of limitation will depend on rates of human ignition as a function of context (e.g. proximity to towns/cities; Bradstock and Gill 2001).

## 5.4 Global change effects

Influences of global change on fire regimes are potentially varied (Fig. 5.4), due to the complexity of underlying processes. The influence of climate change can be distinguished from other influences such as elevated CO2 and land use (Fig. 5.5a, Fig. 5.5b). Climate change has the potential to affect B, A, S and I (Fig. 5.4). For example, changes in available moisture (via rainfall, temperature and evaporation) may directly affect B and A, while changes to temperature and wind will affect S. Elevated $CO_2$ may mostly affect B and possibly A, through alteration of the growth and competitive performance of plant functional types. Human activity can affect B and A via the introduction of new plant functional types. Activities such as grazing and clearing may directly affect B, while there is ever-present potential for human activity to affect ignition (I).

Responses of differing ecosystems can therefore be expected to vary according to the balance of these influences and the nature of fire regime limitation in each instance. Woodlands should inherently be sensitive to factors that affect B, whereas forests should be sensitive primarily to factors influencing A and S.

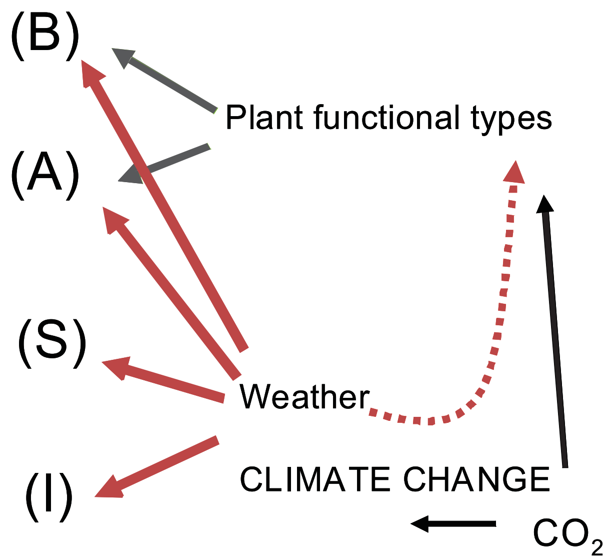



**Fig. 5.4. Potential effects of climate change and elevated atmospheric $CO_2$ on drivers of fire regimes.** Biomass growth (B), availability for burning (A), ambient weather conducive to fire spread (S) and ignitions (I).

### 5.4.1 Climate change effects

In woodlands, changes in available moisture have potential to affect B. Rainfall at stations relevant to the AW and TGW biomes is predicted to either increase or decrease under a range of 2070 scenarios (CSIRO and Bureau of Meteorology 2007). The median predictions for the biome stations (Table 5.2) are for a decrease in all cases. The consequent decline in available moisture will be exacerbated by predicted increases in evaporation in all cases (Table 5.2). Increases in summer rainfall and various levels of decrease in cooler season rainfall are generally predicted. Consequent changes to fuel (B; Table 5.2, Fig. 5.5a) may include decreased grass/herbage in AW and TGW. Increases in summer rain could favour the growth of the important C4 grass component in each of these cases (Murphy and Bowman 2007). While fire danger may tend to increase in these regions due to climate change (Table 5.2), such effects may be less significant due to inherent fuel limitation in woodlands (i.e. fire-weather is non-limiting). Thus there is potential for area burned to either increase or decrease in woodlands due to effects on B, but the balance of these effects suggests a decline (Fig. 5.5a).

By contrast, forests are expected to be sensitive to changes in fire danger and frequency of severe drought under climate change (A + S; Fig. 5.5b). General increases in drought and fire danger indices are predicted in most forested regions (Table 5.2) but the magnitude of predicted change is variable. Hennessy *et al.* (2005) and Lucas *et al.* (2007) predicted substantial increases in drought frequency and days of Very High to Extreme fire danger across broad regions of the south-eastern mainland, but very little change for southern Tasmania (Hobart – Table 5.2). These studies predicted an increase in severity of drought and fire danger, rather than a change in drought frequency, as they utilise the contemporary pattern of daily weather as their basis. Changes to ENSO severity are predicted (e.g. CSIRO and Bureau of Meteorology 2007) but detail of potential change in ENSO frequency is lacking. These could be crucial to changes in fire activity in many forested regions. Current estimates of change to fire danger may therefore underestimate the potential for climate change to alter fire regimes.

Declining moisture may result in reduced rates of litter fuel accumulation in DSF and WSF (Fig. 5.5b, Table 5.2; see also section 4.3) as demonstrated by comparative trends in litter fuel accumulation under differing contemporary moisture scenarios (Walker 1981; Raison *et al.* 1983). Such effects may tend to reduce area burned. Notwithstanding this possibility, the prognosis for forests is for increasing area burned due to effects of elevated fire danger. The TF may be an exception, due to the weak limiting effects of both fuel and fire-weather. The potential for climate change to affect either of these influences is more constrained; however, a reduction grass growth due to drying (Table 5.2, Fig. 5.5b) has potential to decrease the area burned (Harris *et al.* 2008).

There are few detailed predictions of changes to ignition rates from lightning under climate change (e.g. Hennessy *et al.* 2005). Earlier work suggests that there may be a trend for increasing lightning ignitions (Price and Rind 1994; Goldammer and Price 1998) due greater atmospheric instability under global warming.

**Table 5.2. Global change scenarios in case studies in differing Australian ecosystems.** Climatic predictions are 2070 90p scenarios from CSIRO and Bureau of Meteorology (2007) for Darwin (TF), Alice Springs (AW), Dubbo (TGW), Adelaide[a] (TGW Mediterranean), Sydney (DSF), Perth[b] (DSF Mediterranean) and Hobart (CTSF). Fire danger scenarios based on Williams et al. (2001)[1] and Lucas et al. (2007)[2] for 2050.

| Global change attribute | Tropical open forest (TF) | Arid woodlands (AW) | Temperate grassy woodlands (TGW) | Temperate dry sclerophyll forests (DSF) | Cool temperate wet sclerophyll forests (WSF) |
|---|---|---|---|---|---|
| Rainfall (% change) | -1 | - 9 to -17 | - 4 to -7<br>-7 to -13[a] | - 4 to -8<br>-11 to -19b | -3 to -6 |
| Temperature (°C change) | +1.7 to +3.2 | +1.9 to +3.7 | + 1.7 to + 3.3<br>+1.5 to + 2.8[a] | +1.6 to +3.0<br>+1.4 to +2.7[b] | +1.1 to +2.1 |
| Evaporation (% change) | +5 to +10 | +4 to +7 | + 4 to + 9<br>+ 3 to + 6[a] | +5 to +9<br>+4 to +7[b] | +5 to +10 |
| Fire Danger (very high /extreme days p.a.) | Increase[1] | Increase[1] | + 4.4 to +20.8[2]<br>+1.6 to +11.52,[a] | +0.4 to +6.6 (Sydney)[2]<br>Increase likely[1] (Perth) | 0 to +0.2 vh/ext. days p.a. (Hobart)[2] |
| Sensitivity (direction of change in mass) of main fuel types to: (1) climate change; and (2) elevated $CO_2$ | Annual grasses (1) decrease (2) decrease | Perennial grasses and annual herbs/ grasses (1) decrease (2) decrease | Perennial grasses and annual herbs/grasses (1) decrease (2) decrease<br>Woody plant litter (1) decrease (2) increase | Woody plant litter and shrub crowns (1) decrease (2) decrease | Woody plant litter (1) no change (2) decrease |
| New plant functional types | Gamba grass | Buffel grass | Tree plantations | Exotic grasses – Mediterranean areas | |
| Ignitions | | + anthropogenic | - anthropogenic | + anthropogenic | + anthropogenic |

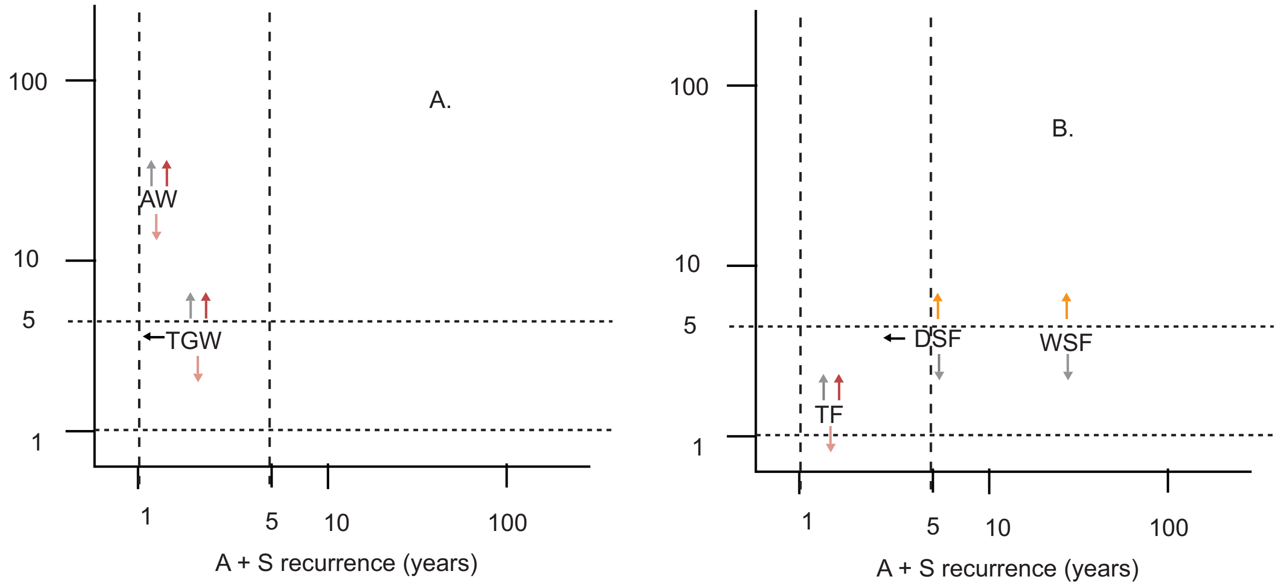



**Fig. 5.5. Effects of global change on factors governing fire regimes in (A) woodland and (B) forest** (See Fig. 5.3). Arrows indicate direction of effects on potential and contemporary fire regimes in relation to biomass growth (B) and fire danger (A + S). Climate change effects are indicated by various arrows: fire danger – black; grass/herb growth – red; litter fuel, accumulation – orange. Other global change effects are indicated by: elevated $CO_2$ – grey arrow; exotic species – pink arrow.

### 5.4.2 Other global change effects

Changes to fire regimes stemming from new plant functional types, altered plant growth, land use and ignition rates range from those that are well known (e.g. effects of exotic grasses) to speculative and highly uncertain effects (e.g. consequences of elevated atmospheric CO2).

In TF and AW, there is evidence that new plant species (exotic grasses) may affect fire regimes. In TF, introduction of gamba grass (*Andropogon gayanus*), an African exotic, has resulted in the elevation of fuel loads and consequent fire intensity (Rossiter *et al.* 2002). The rate of expansion of gamba grass is relatively rapid in places (e.g. the Darwin hinterland), although the bulk of TF remains unaffected. The potential for further spread and consequent changes to fire regimes across northern Australia is high. This situation is paralleled in arid areas of northern and central Australia (AW) by buffel grass (*Pennisetum ciliare*; Clarke *et al.* 2005a = *Cenchrus ciliata*). This species has high drought tolerance, thus providing levels of biomass and spatial connectivity that may exceed that contributed by native grasses and herbage (Clarke *et al.* 2005a).

Reafforestation of agricultural land in the south (TGW and some DSF regions) for timber, paper or fuel, carbon credits, salinity control and habitat restoration (e.g. exotic conifers – *Pinus* spp; native and exotic *Eucalyptus* spp.) may enhance landscape surface fuel connectivity, particularly in heavily fragmented landscapes, where cleared land may otherwise be seasonally fuel-free due to cropping and/or grazing. Further fragmentation of natural vegetation through urbanisation and agriculture (see Hobbs and Yates 2003) may counteract such trends.

Elevated atmospheric concentrations of $CO_2$ may differentially affect plant growth, via enhanced effects on woody species relative to herbs and grasses – particularly C4 grasses (Wang 2007; Osborne 2008). Such effects may be dependent on resource availability (e.g. water and nutrients) and this may be particularly important in Australia, given predominantly low-fertility soils and aridity (Hughes 2003; Steffen and Canadell 2005).

*Hakea eyreana* (Corkwood) basal resprouting after intense Buffel Grass Fire, western NSW
Source: Peter Clarke

Elevated $CO_2$ could affect woodlands and TS by altering the balance between grasses, herbs, shrubs and trees (e.g. Banfai and Bowman 2005; Berry and Roderick 2006). Major changes in grass–tree ratios in southern African savannas, with resultant effects on fire regimes (i.e. area burned positively related to grass cover), are postulated to be driven by fluxes in atmospheric $CO_2$ concentration. Increases in tree and shrub density at the expense of grass/herb cover are implicated in historical declines in area burned in TGW (e.g. Noble 1997; Noble and Grice 2002). Elevated $CO_2$ may increase accumulation of forest litter fuels via an increase in growth and accession of litter and decreased decomposition. Increases in the C:N ratio of leaves (reduced palatability for vertebrate and invertebrate consumers) could slow decomposition (e.g. Gleadow *et al.* 1998; Wang 2007; Box 4.1).

Rates of fire incidence are known to be positively related to population density and proximity to the urban interface (Bradstock and Gill 2001; Keeley and Fotheringham 2002). Continuing expansion of existing major urban centres and population drift from cities to rural areas may alter ignition patterns in south-eastern and south-western forested regions in particular (DSF, WSF).

### 5.4.3 Future trends in fire regimes

Factors that impinge on the performance of herbaceous/grass functional types (B) will have major effects on Australian fire regimes, because systems dominated by herbaceous fuels (e.g. woodlands) cover the bulk of the continent. Given that a range of effects on herbaceous growth and spatial continuity (B) are plausible, the potential for major changes to Australian fire regimes is high. Nonetheless, synergistic effects of all differing global change factors on fire regimes appear unlikely in many ecosystems. Rather, global change factors are more likely to have antagonistic effects (Fig. 5.5a, Fig. 5.5b). Both the direction and magnitude of changes to Australian fire regimes are therefore uncertain, given the difficulty in quantifying the magnitude of certain vectors.

The outcome for future fire regimes in woodlands (TGW and AW) will depend largely on the interplay between moisture and elevated $CO_2$ effects on B. If a decline in moisture occurs as predicted under the median 2070 scenarios (Table 5.2) then a decline in grass/herb biomass will result. This may complement the negative effect of elevated $CO_2$ on area burned and fire frequency (Fig. 5.5a). The spread of exotic functional types such as buffel grass could play a contrary role over significant areas of AW due to their effect on landscape connectivity (Fig. 5.5a).

Enhancement of woody plant growth in woodlands may also be context-specific. In arid and semi-arid regions, woody cover is dominated by *Acacia*, *Casuarina* and *Callitris* species, which form litter beds of low flammability (Bradstock and Gill 1993; Bradstock and Cohn 2002 ). In higher rainfall TGW areas, eucalypts with high flammability are more prominent. Negative effects of elevated $CO_2$ on fire, through stimulation of woody growth, are therefore likely to be more pronounced in semi-arid and arid woodlands. Resolution of the outcome of opposing effects of increasing summer rainfall (favourable for C4 grasses) and elevated $CO_2$ (favourable for woody plants) may be particularly significant in TGW. In moist TF regions (grassy woodland–open forest), encroachment of rainforest from enclaves due to $CO_2$ effects may also diminish the area burned (Fig. 5.5b).

The balance between fire-weather/danger effects (A+S) and fuel effects (B) will determine the nature of future fire regimes in forests (Fig. 5.5b), in contrast to woodlands. Because of inherent limitation of A+S in forests, predicted increases in fire danger make an increase in average area burned plausible. It is arguable that forecast fire danger changes are conservative and do not account for crucial changes in major drought frequency that may determine the magnitude of any forcing of fire regimes by climate change. Potential increases in ignitions (I) from urban expansion may be synergistic with fire-weather effects in many forested regions.

### 5.4.4 Conclusions

A general conclusion from these biogeographic analyses of Australian fire regimes, their drivers and the factors that constrain them (the 'limiting switches') is that, broadly, in grassy fuel systems – especially those in arid/semi-arid environments – fire follows rain. In contrast, in ecosystems where woody fuels dominate the fuel bed, such as the temperate sclerophyllous forests and scrublands of the south-east and the south-west, fire follows drought.

Climate change and elevated $CO_2$ may lead to a decrease in area burned and fire frequency in systems dominated by herbaceous fuels, due to effects on limiting 'switch' (B). Factors such as spread of exotic grasses have potential to cause contrary effects in these ecosystems in some instances. In forests dominated by woody litter fuels, alterations to B through $CO_2$ or other effects may be less critical than changes to fire danger (M+S). Climate change therefore has a strong potential to increase area burned in temperate forested regions through an increase in severity of fire danger, including the contribution of drought.

These examples do not represent fire regime trends in all Australian ecosystems, or all the possible effects of global change. Further extension and refinement of this approach may be appropriate, though adequacy of understanding of fundamental processes (e.g. fuel dynamics, fire behaviour, ignition patterns and sources) may impose constraints. For example, shrub-dominated communities (e.g. various shrublands, heaths and shrubby woodlands), though relatively small in area at a continental scale, are of great significance in terms of biodiversity (Hopper and Gioia 2002; Keith *et al.* 2002; Russell-Smith *et al.* 2002).
These communities span a range of environments comparable to TGW through to WSF. Fire regimes in some cases may be governed not only by *in situ* processes, but also characteristics of adjacent communities (e.g. heath patches within tropical savanna or temperate DSF). Differing approaches may be needed to predict future fire regimes in such cases. Ultimately, given the complexity of responses and drivers of fire and potential feedback processes, appropriate numerical models may be required to fully explore the problem.

This overview has also focused on factors governing area burnt by large fires and resultant effects on fire frequency, rather than effects on fire intensity and season. The approach could be extended to explore these components of fire regimes, particularly as information on spatial variation in fire severity accumulates. While these deficiencies are considerable, there is sufficient evidence to indicate that fire regime variation among Australian ecosystems can be explained by variations in the rates of four basic drivers. This provides a basis to guide the development of quantitative fire regime models in the future, as well systematic data collection to document the nature of the processes that shape these drivers.

The concept of spatial connectivity of the drivers of fire underpins the ‘four-switch’ concept presented here. Further exploration of the way that landscapes develop ‘flammable connections’ through interactions between fuel, weather and terrain (Falk *et al.* 2007) will provide an incisive basis for predicting future fire regimes. The concept of spatial and temporal fluctuation in flammable connectivity also provides a basis for understanding potential influences of management. Fire management in its varied forms attempts to impose discontinuity. Given that flammable connectivity is dynamic in most ecosystems, the effectiveness of fire management will depend on the rate at which disconnection can be imposed. The relative strength of these two opposing influences on connectivity (natural versus imposed) will ultimately determine ‘effectiveness’. Such considerations may enhance our ability to understand fire regimes and their consequences in the future.

## 5.5 General consequences for biodiversity

The scenarios sketched in Fig. 5.5 provide a speculative basis for consideration of general consequences for biodiversity. A number of possibilities stand out:

- *TF* – Elevation of fire intensity and possibly area burned due to invasion of gamba grass may result in structural change to savanna through diminution of tree cover; this may have wide-ranging habitat consequences. Such changes in fire regimes would run contrary to the current thinking about appropriate TF fire management, which is actively seeking to reduce net area burned and average fire intensity, due to perceived deleterious impacts.
- *AW* – As with TF, potential elevation of area burned due to invasive species may pose the most important adverse effects, although effects of moisture and elevated $CO_2$ could tend to have contrary effects. Woody plant communities consisting of long-lived obligate seeders (e.g. *Callitris* and *Acacia* spp.) may be the most vulnerable to any elevation in area burned, though adverse effects on other biota such as mammals could also follow.
- *TGW* – A possible diminution of area burned may have the most far-reaching consequences due to possible negative consequences for herbaceous diversity (Prober *et al.* 2007, 2008). Increases in woody plant cover (e.g. recruitment of *Callitris*) may exacerbate such changes (i.e. act as a positive feedback).
- *DSF* – A substantial increase in area burned and fire frequency in temperate sclerophyll open forests – and associated shrubby woodlands, shrublands and heaths – could have major consequences for community composition and structure. Many of these plant communities are relatively diverse, and sensitive to changes in fire frequency and intensity. Linkages with vertebrate diversity mean that shifts in plant community composition and structure have wider consequences.
- WSF – As with other sclerophyll forests, an increase in area burned and fire frequency may have consequences for community composition. Such changes may acutely affect cool, temperate rainforest enclaves within the sclerophyll plant community matrix.

# 6. CLIMATE CHANGE, FIRE REGIMES AND BIODIVERSITY: FOUR CASE STUDIES

## 6.1 What are the issues and potential approaches?

Questions regarding the prediction of the effects of climate change and fire regimes on biodiversity can be profitably addressed at multiple scales but caveats on the conclusions reached are essential given the uncertainties of prediction. At a national, broad-biome scale, the directions in which fire regimes are likely shift in relation to climate change have been indicated in the previous section. Species may move and communities change composition in response to changes in climate and fire regimes given the time to do so. Directions of movement of different plants and different animals may vary. Barriers to movement may be present in the landscape and their effectiveness will depend on distances of dispersal of individual species.

Our stated approach to examining the climate change–fire–biodiversity problem has been to focus on what climate change may mean for fire-weather, and how this might translate into changes in fire regimes. This has been examined in sections 1 to 4. Translating these potential changes in fire regimes into potential changes to the dynamics of ecosystems is the next component of the analysis that we consider. We outlined a potential national framework for this in section 5. In this section, we now consider the climate change–fire regime–biodiversity interactions via a series of case studies in regions for which we have data, models and appropriate insights into community dynamics. The aim is to evaluate relative sensitivities of biodiversity to change in fire regime components at more regional scales, in select parts of Australia.

The four case studies are: (1) the alpine ash forests of south-eastern Australia; (2) the Mediterranean ecosystems of south-west Western Australia; (3) the tropical savannas of northern Australia; and (4) the temperate sclerophyllous vegetation of the Sydney Basin. We have chosen these particular case studies because they represent different biomes of the country, and provide a framework within which to contrast climate, vegetation structure, fuel types, fire regimes and different prognoses for climate change impact on fire regimes and landscapes. They explore potential impacts based on research spanning multiple scales of resolution and approaches ranging from ecophysiology to population and landscape modelling. Case studies 1 and 2 use empirical lines of evidence, based on a consideration of ecosystem physiology and demography. Case studies 3 and 4 are based more on simulation modelling.

The case studies, although regionally focused, are not comprehensive, or even fully representative, treatises of regional issues. They do not use the same methodology to explore climate change–fire regime–biodiversity interactions. They are designed to be illustrative of the diversity of methodological approaches that could be used to understand biodiversity, fire and climate change. Case study 1 is aimed explicitly at evaluating potential change in a single (albeit reasonably widespread) plant community type; the other three are conducted at regional scales, within which there are numerous plant community types.

The Sydney Basin case study is the one where the main research infrastructure needed to undertake evaluations of climate change, fire regime and biodiversity is best developed. That is, there is a regional, spatially explicit fire model that can be parameterised for climate change scenarios. The fire model is, in turn, linked to a dynamic vegetation model. This combination of models also allows us to evaluate, at a regional scale, sensitivity of risk to both property and biodiversity assets, to various management options. The approach, data, models and lines of evidence used in the Sydney Basin study therefore represent a potential generic approach for evaluating the consequences of climate change-induced changes to regional fire regimes and biodiversity.

## 6.2 Case study 1: Alpine ash forests in south-eastern Australia

*Summary*

The tall alpine ash (*Eucalyptus delegatensis*) forests of south-eastern Australia are relatively well known ecologically and so provide a benchmark for what we know and what we need to know if realistic predictions about biodiversity are to be made in the context of fire regime change. Alpine ash is an obligate seeder, and populations of the species are killed by fires of sufficient intensity to scorch the crown completely. If two canopy-scorching fires occur within the time it takes for seeds to form and mature then local extinction of the species may occur; this is not unprecedented. Under enhanced atmospheric CO2 concentration, the productivity of the forest may be expected to rise. This may be reflected in higher fuel loads of litter and grass, and perhaps shrubs, but the extent is uncertain. If fuel loads rise, then potential fire intensities may be expected to increase. With changes in weather towards drier and warmer conditions, and perhaps an increased lightning frequency, the chance of fire occurrence would be expected to rise. All these changes point towards an increased possibility of local extinction of alpine ash if fire return intervals fall below the period needed for juveniles to flower and seed. However, the species may be able to migrate along with appropriate geographic shifts in weather. Such shifts are likely to be upward in altitude; if movement does take place in line with current domains of weather variables, population shifts may be thwarted in part by altitudinal limits. Alpine ash is an ideal target species for monitoring, and reacting to, any changes observed in the distribution of the forest as a consequence of changes to climate and fire regimes. Scientists, land managers and naturalists all have a part to play in monitoring, and reacting to, any changes observed in the distribution of the forest.

### 6.2.1 Introduction

In this section we explore the climate change, fire and biological interactions in a relatively well-known species, alpine ash (*Eucalyptus delegatensis*), a tree of south-eastern Australia. We begin these case histories using a single species as it is at this level of detail that the complexity of the fire, climate change and biodiversity issue is revealed.

Climate change is also predicted to affect ecosystems via its impacts on individual species (Dunlop and Brown 2008), so a case study that explicitly examines potential impacts on a single species is apposite to the aims of this project. Moreover, individual species can be expected to behave in individual ways, not only in relation to climate change, but also to fire interval, fire intensity, type of fire (surface, crown, peat) and season of fire. Examining alpine ash allows us to illustrate this more holistic approach to fire regimes and their effects under climate change (temperature, rainfall, wind speed, relative humidity and $CO_2$ fertilisation).

The present case study is designed to be one that touches on many aspects of fire and climate change, not just a few; exemplifies a particular plant functional type; sketches a case history of a vertebrate animal living in a closely associated vegetation; considers the consequences of a shift in distribution of the species; and makes recommendations for managerial action for an uncertain future. As such, it shows how the conservation manager might view the issues surrounding climate change, fire and biodiversity in his or her reserve. There is no quantitative model that deals with all these elements for any ecosystem, as yet. The alpine ash example comes from, but substantially extends, considerations of, Gill *et al.* (2008).

The detail given provides a benchmark for what we know and what we need to know if realistic predictions are to be made in a fire context. Uncertainty is a theme. Given such uncertainty, and the fire focus of this report, the need for enhancing our current understanding of fire regimes and their effects on Australia's biodiversity may be obvious.

Application of present knowledge takes place in the field at a regional scale or finer. At the broad scale, conservation of biodiversity can be informed by predictions of change, but it is the local scale that is the province of the land manager as a custodian of indigenous species.

### 6.2.2 The species and its distribution

Alpine ash is a tall forest tree that inhabits montane environments of south-eastern Australia including Tasmania (Hall *et al*. 1975, pp. 185–186). Forests of its close relative, mountain ash (*E. regnans*), or mixed-species eucalypt forest, tend to occupy lower altitudes on the Australian mainland when the two species are in the same area. At higher altitudes, alpine ash is usually replaced by woodlands of snow gum (*E. pauciflora*). The species is an important one in water catchments, recreation areas, forestry land and national parks. Its stature helps determine the vegetation class in which it occurs – tall open forest. The understorey, a major fuel source, varies from dense shrubbery to grass (Bowman and Kirkpatrick 1984) and litter.

Unburnt alpine ash (*Eucalyptus delegatensis*), Brindabella Range, Australian Capital Territory, mid 1990s.
Source: Geoff Cary

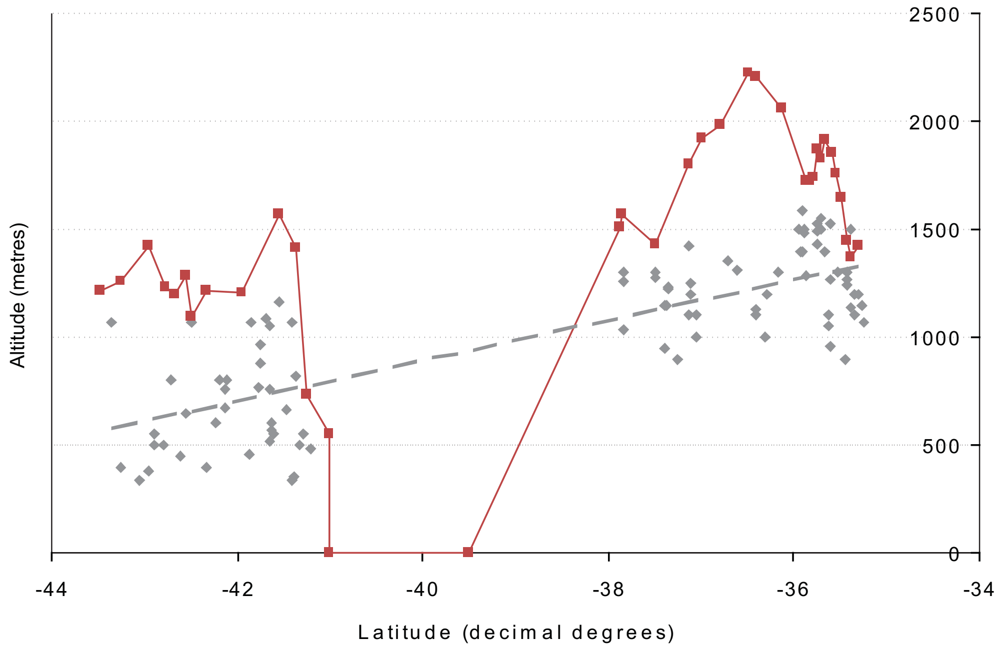



**Fig. 6.1. The distribution of alpine ash by latitude and altitude for the south-eastern Australian mainland and Tasmania.** Data were kindly supplied by the Australian National Herbarium (ME Nightingale). The red line represents the altitude of selected mountain peaks, the grey points are location records for alpine ash, and the grey dashed line represents the mid-point of the altitudinal distribution at any latitude. Source: After Gill *et al.* (2008).

Fig. 6.1 represents the distribution of alpine ash by altitude and latitude. The data points and associated trend lines represent one way to depict the realised niche (Austin 1992) of the species, i.e. where it grows now. However, altitude as a variable is really just a convenient surrogate for temperature and rainfall; these, in themselves, are simplifications of the way that weather and plants interact. Fig. 6.2 (see also *Gill et al.* 2008) shows the same data as Fig. 6.1 but uses interpolated datasets for temperature and rainfall from BIOCLIM, a bio-climatic splining program that enables annual as well as seasonal variables to be predicted for a collection of geocoded locations (http://www.cres.anu.edu.au/outputs/anuclim/doc/intro.html). The seasonal variables are perhaps more significant as discriminating variables than the annual variables. Graphs such as this cannot represent the importance of micro-climates or of disturbance niches.

Though the use of interpolated estimates of rainfall and temperature data to describe the realised niche can be useful, it could be simplistic because there may be edaphic factors of importance, as well as diseases (and dieback more generally; Ellis 1964) and herbivores. Furthermore, it will be shown later that there is a fire regime aspect to niche description also.

The realised niche is often used as a means of predicting the effects of climate change on species redistribution and the potential for extinction (e.g. Pearson and Dawson 2003; Thomas *et al*. 2004). Using the fundamental niche – where the plant could grow if free from all constraints – may be a better basis for prediction. The fundamental niche of forest tree species is generally known, as a result of commercial forestry species trials in Australia and especially overseas where, as far as possible, competition with other species is eliminated (e.g. Booth *et al*. 1988, 1991). With alpine ash, the fundamental niche from a fire regime point of view, however, is relatively well known (compared with most species in Australia).

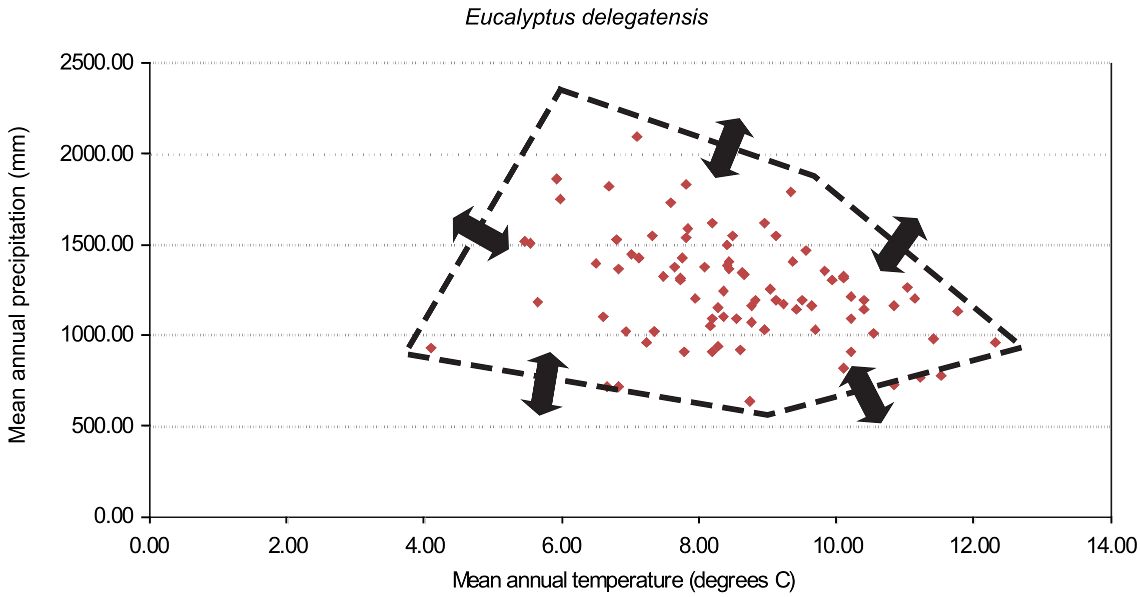



**Fig. 6.2. The realised niche of alpine ash as a function of average annual temperature and average annual rainfall.** Data points produced using BIOCLIM. (Source: http://www. cres.anu.edu.au/output/anuclim/doc/intro/html). The arrows indicate that the niche space could change as the result of new circumstances. The dots represent individual stands. Source: After Gill *et al.* (2008).

### 6.2.3 Alpine ash: a 'seeder', a functional type with susceptibility to certain fire regimes

Alpine ash is particularly interesting biologically because, like mountain ash (Ashton 1981), it is a species that has populations that die when their crowns are killed, i.e. when the crown has been scorched or flame-defoliated. When the population dies, the persistence of the species on the site depends on seed stored in the canopy of the tree at the time of the fire. These characteristics put alpine ash in the plant functional type known as a 'seeder'.

Seed is shed at far faster rates and in greater amounts after crown-killing fire than in the absence of fire (O'Dowd and Gill 1984). Because of this, seed-harvesting ants on the ground are satiated, and many seeds escape to germinate and grow on the newly-exposed soil (O'Dowd and Gill 1984); a new cohort of alpine ash is formed. Because of this sequence, stands are often one age or have a limited series of ages connected to previous fire events. If there is no fire within the life span of the trees, it is to be expected that as trees die they will not be replaced; however, this requires verification especially in situations where the understorey is not well developed.

Given that the trees are vulnerable to crown-killing fires, and dependent on seed stored on the trees, the timing of the first and the last seed production become important. While nothing is known about the latter, there is some information on the former. Cremer *et al.* (1984) suggested that even though seedset might start at 10 years of age, not much seed would be produced until stands were 20–30 years old 'unless they are widely spaced'.

Predicting the death of the trees, thereby enabling regeneration when there is seed on the trees, is an important feature of the life cycle. Death of the trees may be predicted from models of scorch height; when the modelled scorch height exceeds the target tree height, then the tree is deemed to have been killed. Tree height may be predicted from growth data such as that of Borough *et al.* (1984):

$HGT(m) = 48(1-e^{-0.035*AGE})$

where *HGT(m)* is tree height in metres and *AGE* is the age in years. This equation was extrapolated to age 400 years when the height, as expected from the equation, was close to 48 m (Fig. 6.3). Consistent with this, Walsh and Entwistle (1996) suggested that trees of this species reach 50 m, rarely 90 m, in Victoria.

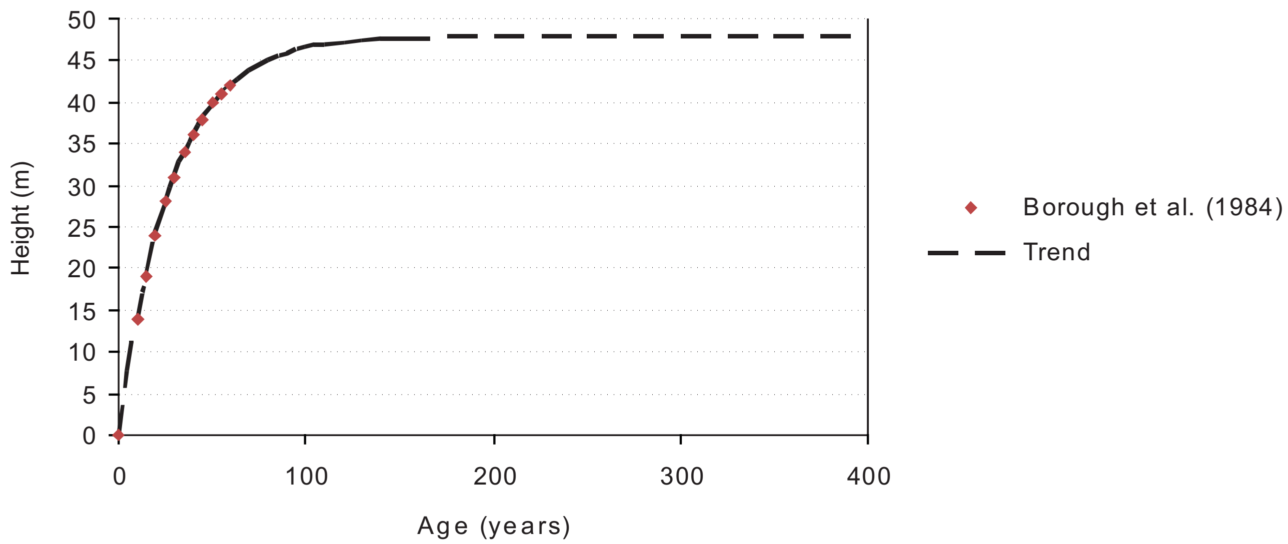



**Fig. 6.3. Height growth of alpine ash**. The data of Borough *et al.* (1984) are extrapolated to the possible life span of the trees.

Fire intensity may be predicted from rate of spread of the fire and fuel load (Byram 1959). Rate of spread of the fire may be predicted from McArthur's (1967) meter (Noble *et al.* 1980):

$ROS_o = a*FFDI*FUEL$

where $ROS_o$ is the rate of spread of the fire on level ground burning with the wind, *a* is a constant, FFDI is the forest fire danger index and *FUEL* is the fuel loading (McArthur 1967). Hence, the more severe the fire-weather, and the greater the fuel load, the faster is the fire. How litter fuels accumulate over time in alpine ash stands has been studied by Raison *et al.* (1983) and is shown in Fig. 6.4.

Fuel loads usually follow the form described by Olson (1963); see section 4.3.2:

$FUEL = FUEL_{MAX} [1-E^{-K*TSF}]$

where *K* is the decomposition constant, *TSF* is time since fire (years) and $FUEL_{MAX}$ is the quasi-equilibrium fuel load. In Raison *et al.*'s (1983) study they found that $FUEL_{MAX}$ was about 26 t/ha and K was equal to 0.16. However, fuel load can be subject to the effects of CO2 fertilisation as discussed later.

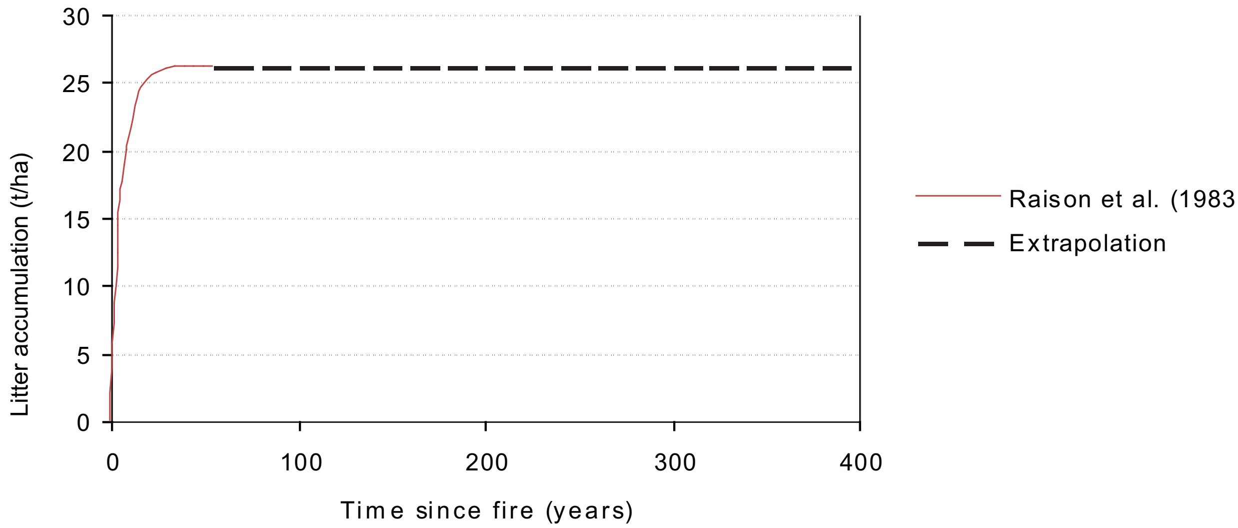



**Fig. 6.4. Litter fuel accumulation in alpine ash stands in the Australian Capital Territory.** Source: After Raison *et al.* (1983). The data are extrapolated to a potential life span of the stand.

FFDI is given by Noble *et al.* (1980), after McArthur (1967):

$FFDI = 1.25{*}D{*}(E^{(T-H)/30 + 0.234{*}V})$

where *D* is the drought factor, a measure of medium term drought, *T* is the air temperature in degrees Celcius, *H* is the relative humidity and *V* is the wind speed in km/h at 10 m above ground. What this equation states is that as drought deepens – (with a cap on the effect, through D) when temperature and wind speed rise, and as humidity falls – the 'fire danger' (FFDI) increases. The implications of an effect of climate change to FFDI are perhaps apparent.

As the FFDI rises, so too does the rate of spread of the fire perimeter where it is burning with the wind. In the field, slope of the terrain is very important, with the rate of spread doubling for every 10 degrees of windward slope (McArthur 1967) as it does with every 30 km/h of tailwind.

Fire intensity needs to be known so that scorch height can be predicted. This is given by Byram (1959) as:

$INT = H{*}FUEL_K{*}ROS_M$

where *INT* is the intensity of the fire in kW/m, *FUEL* is the fuel load in kg/m², and $ROS_M$ is the rate of spread of the fire in m/sec.

Finally, scorch height is a function of fire intensity. For scorch heights in jarrah (*E. marginata*) forest in Western Australia up to about 30 m, Burrows (1994) found:

$SH=0.41{*}INT^{0.55}$

where *SH* is the scorch height in metres. This equation shows that as intensity increases so too does scorch height, but at a rate less than expected if the relationship were linear. This relationship changes seasonally (Burrows 1994) perhaps due to seasonal changes in leaf temperature and moisture.

What these equations suggest, given that fuel accumulation stabilises at about 50 years (Fig. 6.4) and that height growth continues to about 100 years (Fig. 6.3), is that crown scorch – and death of the trees – is harder to achieve when the trees are >100 years old than when they are 50 years old, because fuel accumulation has remained the same but the trees are taller. While this may be so, the use of this set of equations to depict the system does not include the possible raising of scorch heights by fires reaching into and consuming shrubby understoreys. How the variables in the above equations may change as a result of $CO_2$ fertilisation and climate change is examined below.

A summary of the relationships outlined above is given by Fig. 6.5, which shows how a population of trees responds to changes in fire intensity with age. The approximate positions of critical life history markers are also given. Humus or peat fires could also kill the trees (Cremer 1962) but their effect is not considered in Fig. 6.5.

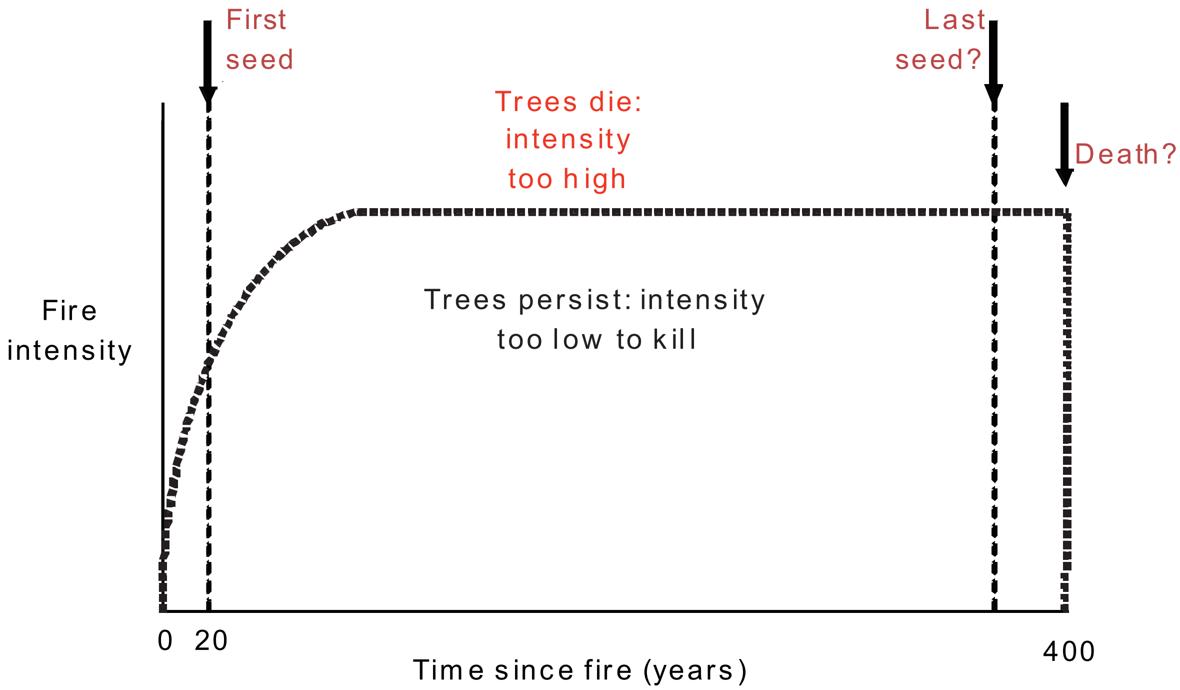



**Fig. 6.5. The responses of alpine ash to fire intensity during its lifetime.**

Once a new cohort of alpine ash has been established it grows rapidly (Fig. 6.3). If the trees are killed before seed production begins (Fig. 6.5), after it ends (Fig. 6.5) or during an 'off year' (Owen Bassett, pers. comm..), then there is no seed for further cohorts to develop and the species becomes locally extinct. This occurred for young alpine ash stands in parts of north-eastern Victoria in the 2006–2007 fires following the 2002–2003 fires. Thus, the interval between crown-killing fires is of critical importance in determining the local persistence or extinction of the species.

A possible insight into the more detailed nature of fire interval in alpine ash stands is provided by a possum, the rare Leadbeaters possum (*Gymnobilideus leadbeateri*), which inhabits forests of mountain ash (Mackey *et al.* 2002). The hollows of large trees of this species are an essential component of the habitat together with feeding trees of *Acacia* (Mackey *et al.* 2002). It can be demonstrated mathematically that variation in the fire interval around a mean interval will enhance the population of Leadbeaters possum (MA McCarthy, AM Gill and DB Lindenmayer, unpublished). Thus, for this ecosystem, the nature of the variation about mean levels of fire regime components could be an important part of the habitat.

Populations of Leadbeater's Possum are favoured by variation in the fire interval.
Source: Jean-Paul Ferro/AUSCAPE

### 6.2.4 Atmospheric $CO_2$ 'fertilisation' and its effect on fuels

Changes in the $CO_2$ concentration of the atmosphere are the basis for the global climate changes and their effects addressed in this report. Both the changes in $CO_2$ and changes in the weather could affect fuel types and their accumulation, thereby affecting fires and biodiversity.

The accumulation of litter fuels in stands of alpine ash shown in Fig. 6.4 could be affected by changes in atmospheric concentrations of $CO_2$. Because of the variety of understorey fuels in alpine ash, there could be different effects on fuels in different places but increases in accumulation are likely. In FACE experiments in grass steppe in the USA, Morgan *et al.* (2004, 2007) found increases in grass yields of 41% per year and that C3 grasses may gain an advantage over C4 grasses, and shrubs over grasses. Thus the situation is by no means simple; both species composition, and fuel structure and composition, can change. In a similar FACE experiment, Stokes *et al.* (2003, 2005, 2008) demonstrated enhanced biomass production of both C4 grasses and woody species in an Australian tropical savanna.

The effects of increasing $CO_2$ on vegetation communities in general, and forest understoreys in particular, will depend on what is limiting growth. Cowling and Field (2003), in reviewing the literature on control of leaf area production, pointed out that low nutrients, water deficits and self-shading may limit the potential for the expression of a $CO_2$ effect.

While the growth of trees may be enhanced by increases in atmospheric $CO_2$ (Norby *et al.* 2005), the way this increase is partitioned (McCarthy *et al.* 2006) is important in the fuel context. In hardwoods in North America, litterfall did not increase but for a conifer needles and bark it did (Finzi *et al.* 2001). Stage of forest growth and soil properties may affect the result (Cowling and Field 2003; McCarthy *et al.* 2006). Thus, there is the potential for changes in fuel type and for increases in fuel load, but there appears to be no published basis for predicting an increase in litter in *Eucalyptus* stands at present. There is no suggestion for a decrease in fuel load within any one type.

### 6.2.5 Changes in weather and its effects on fires and alpine ash

For major regional stations in south-eastern Australia, but all outside the area of alpine ash distribution, weather variables are expected to change so as to increase (Canberra and Melbourne) or maintain (Hobart) the present average FFDI (Hennessy *et al.* 2005, pp. 58, 61, 64); similar results were obtained by Lucas *et* *al.* (2007). Pearce *et al.* (2007), in their 2070 projections from GCMs, have summer temperature rising by 3.2, 3.1 and 2.2°C; annual relative humidity dropping 1.6, 2.3 and 0.6%; summer wind speed varying by +4, -1 and -6%; and summer rainfall varying +1, -4 and -10% for Canberra, Melbourne and Hobart, respectively. If the shift in annual average air temperature was 3.4°C upwards and the temperature lapse rate was 1°C per 100 m, then the midline for alpine ash distribution (Fig. 6.1) would rise to the extent that many populations would become extinct while others would have to establish at higher altitudes in order to persist. If this shift occurred, then the altitudinal spread of the species would narrow, as suitable heights would not be available. Of course, species normally found at higher altitudes could disappear – become locally, if not totally, extinct. It should be noted that this example does not include shifts in rainfall or consider the potentially new edaphic environment at higher altitudes.

Another, more direct source of information on climate change comes from trend analysis in the weather. For weather stations throughout south-eastern Australia, the trend in the historical record of FFDI is upwards or static (Esplin *et al.* 2003; Lindesay *et al.* 2004; Lucas *et al.* 2007; Fig. 6.6 for Canberra).

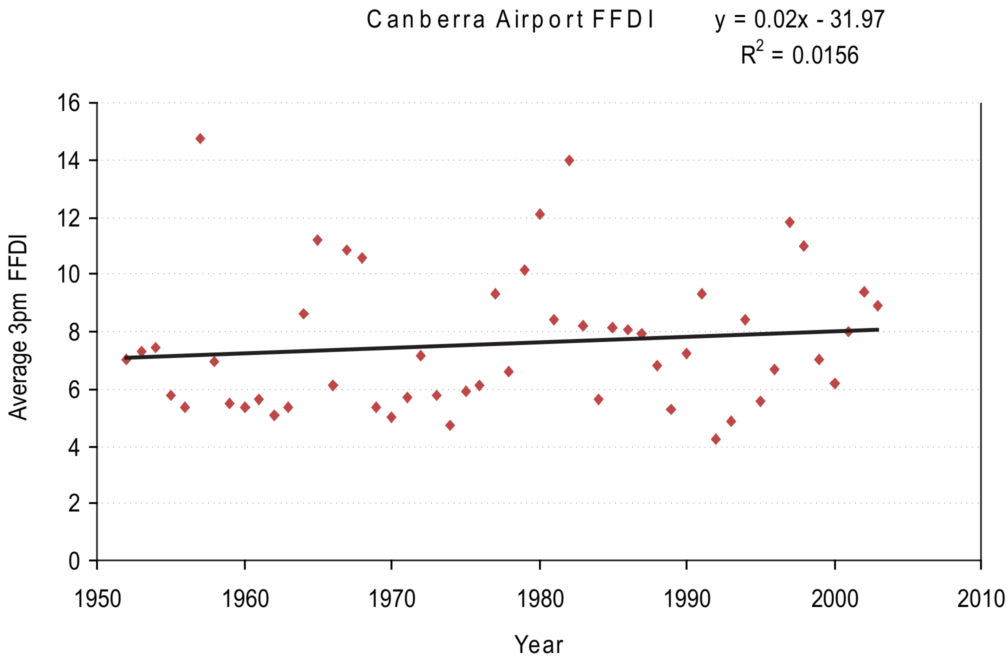



**Fig. 6.6. Trend analysis for yearly averages of 3 pm FFDI at Canberra Airport**. Data are from Lindesay *et al.* (2004). The trend is not statistically significant although it is consistent with analyses of other stations in south-eastern Australia (Esplin *et al.* 2003; Lucas *et al.* 2007). Source: After Gill *et al.* (2008).

Trends in the annual sum of FFDI from the weather record are apparent, but Hennessy *et al.* (2005, p. 11) and Lucas *et al.* (2007, pp. 17–19; see also section 4.2) have noted that some data may be missing from some records and data may vary in quality. Mackey *et al.* (2002, pp. 121–124) had reservations about the outputs of GCMs, especially due to the coarse scale of outputs. Outputs of wind projections vary between the two 'best' models (Hennessy *et al.* 2005).

Questions of variation in FFDI within a day and between years while conditions are changing are difficult to answer but could be significant. However, the implications of a general trend upwards in FFDI are that fire rates of spread will be faster and intensities higher. Thus, the chances of tree death will be expected to be higher; this is only significant for the alpine ash if trees without seed are killed (see previous discussion).

If rainfall declines, the decomposition rate of litter (K) is likely to decline, and increase the quasi-equilibrium level. A temperature increase, without a decline in rainfall, will increase the rate of decomposition and cause a decrease in fuel accumulation. If nitrogen contents of litter decline, in line with decreases in their concentrations in live leaves (Kanowski 2001), then decomposition rates may be lower and accumulation rates higher.

While trends in fuels and weather suggest that increases in potential rates of spread and intensity of fires are likely, albeit uncertain, this remains academic unless ignition rates remain the same or change. According to Goldammer and Price (1998) the incidence of lightning, and presumably lightning-caused ignitions, is likely to increase in Australia. Ignitions from people are an important source of fires in south-eastern Australia, and while their incidence may be expected to be increasing, the trend since the historically unprecedented fires of 2003 in the Australian Capital Territory, has been down (R McRae 2008, pers. comm.; Fig. 6.7).

In the simulations of Cary (2002), the effects of variations in weather due to climate change have been examined for the Canberra region using realistic models of fuel accumulation, lightning ignition and fire spread (section 4.4). The result was for a general decrease in the interval between fires; modelling of the effects of the trend in fire intervals on alpine ash distribution in the region has not yet taken place.

Caption: Burnt stand of alpine ash. Severe canopy scorching fires at frequent intervals can cause local extinctions of apline ash.
Source: Malcolm Gill

### 6.2.6 Opportunities to move geographically

The concept of the realised niche, introduced above, is useful in that it indicates where the species presently occurs. The realised niche can shrink if fire regimes become adverse and the species does not migrate. If areas near to the present distribution become favourable, the species can move into them if their dispersal mechanism is adequate and if competition there does not prevent establishment. In general, there may be favourable areas for the growth of the species well away from present distributions, but the species may not be able to reach them because of inadequate dispersal (Fig. 6.8). This may not seem likely for alpine ash, given its altitudinal distribution (Fig. 6.1).

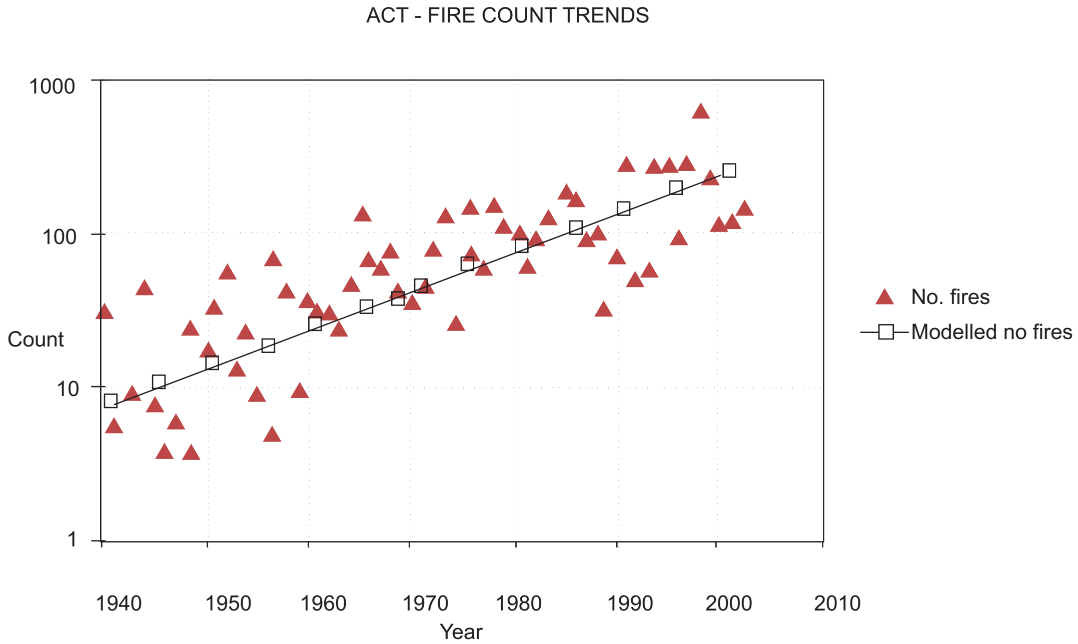



**Fig. 6.7. The numbers of ignitions, mostly human caused, in the Australian Capital Territory since 1940.** (Data of R McRae, ACT Emergency Services Agency).

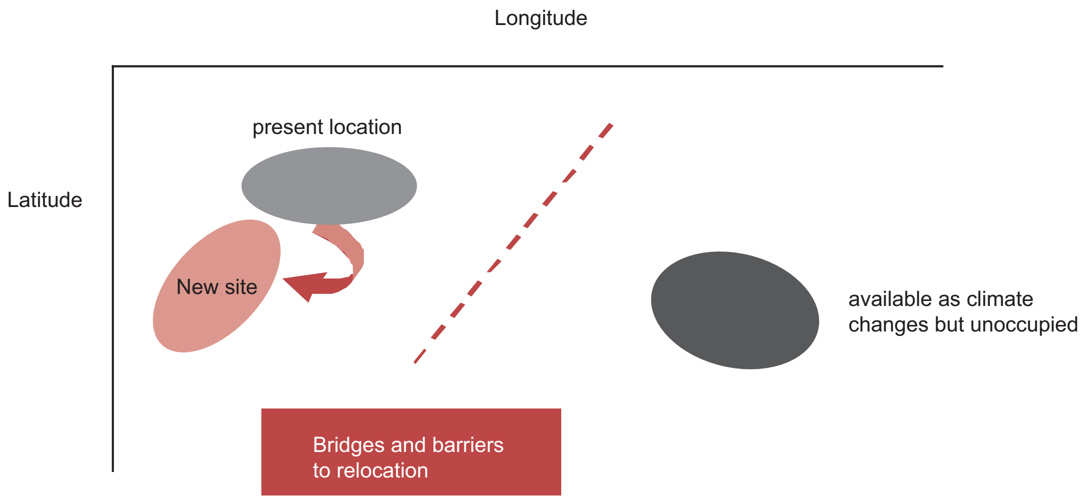



**Fig. 6.8. An illustration of the way a species may shift (red arrow) into another geographic location that becomes more suitable as the climate changes and is within range of the dispersal distance of the species.** The red dotted line represents a barrier to movement of the species to another site that may also be suitable under the changed climate but remain unoccupied because of the barrier.

What constitutes a barrier to dispersal will vary according to the mode of dispersal being considered and the behaviour of the vector as a result of climate change. Thus, for example, alpine ash can spread by seed, albeit in a very limited way. Hence a critical consideration in determining the fate of alpine ash is the rate of change of climatic and critical fire variables in relation to the potential rate of migration. If the species could move in concert with environmental changes then there would be no threat to the species, given that other aspects of habitat – such as soils of a suitable type – were available. Present models tend not to consider rates of change.

### 6.2.7 What can managers (and others) do when predictions are so uncertain?

Given the uncertainty in the prediction of climate change variables, their effects on fuels and fire regimes, and our limited knowledge of realised and fundamental niches even for a relatively well-known species like alpine ash, what can be done? A selection of suggestions for ecologists, reserve managers and naturalists follows.

There is an increasing need for ecologists to understand the effects of the components of fire regimes, and how ecosystems function as an interacting set of plant and animal species. While recognising plant species as taxa or as functional types with respect to fires, habitat elements or fuels, there are many interactions and other variables at work. However, some success is to be had by choosing the most vulnerable species to present fire regimes and monitoring it (or them, across localities) along with maps of fire occurrence and intensity. Where intensity is important to species response, knowledge of functional type of the species, alone, is inadequate for predicting the future course of the species; what individual fires do to the populations present is significant (Fig. 6.5). Functional types at present do not capture the effects of season or fire type (as used by Gill 1975) nor take into account possible interactions between components of the regime.

Monitoring of species thought to be the most vulnerable requires knowledge of functional types as a starting point; systematic recording of these in relation to fire is a feature of the management programs of most Australian states. Because of uncertainty, monitoring of 'wildcards' (species with unknown responses or responses at variance to chosen indicators) is also recommended as a subsidiary activity to the main program. Watching the effects of fire regimes on fungi and potoroos, or beetles (Doran *et al.* 2003), for example, could be revealing.

A formal program of monitoring along an altitudinal gradient has been established in Tasmania, which involves alpine ash forest among other types of forest (Doran *et al.* 2003). Such a program could be extended by its adoption on the mainland for the same or other groups of taxa. It could be run in association with a simple experimental program (at an appropriate scale) so that at each monitoring station there are two fenced and two unfenced plots, one of which is burnt at regular intervals and one which is kept unburnt.

By being aware of the fundamental elements of climate change, more strategic observation of landscapes is possible; intelligent observation of any type assists us all to react appropriately and so better preserve our biodiversity in the face of uncertain change. Naturalists, scientists and conservation managers could report unusual environmental events such as die-offs, out-of-season flowerings, failing seedling cohorts, animal invasions, plagues, mass defoliations, etc. Understanding of the dieback first studied by Ellis (1964), which appears climate-related, could be important.

All of these potential monitoring programs, if focused on adaptive management, would enable managers of conservation reserves to both observe change and react to it.

## 6.3 Case study 2: Sclerophyllous ecosystems of south-west Western Australia

*Summary*

South-western Australia is one of 25 global biodiversity hotspots, and its unique flora and fauna is considered to be particularly vulnerable to the likely deleterious impacts of climate change. Temperature increases and rainfall decreases recorded for south-western Australia during the latter part of the 20th century are among the fastest in Australia, with average annual river flows more than halving since the 1970s. GCM projections suggest further warming and drying for the region, with associated increased occurrence of high fire danger weather.

The shrublands of the Geraldton sandplain region are a plant biodiversity hotspot within the South-West Botanical Province (SWBP). Strong climate gradients and narrow species ranges constrained by substrate conditions make this vegetation type particularly susceptible to climate change impacts. Many large shrub species depend on groundwater mounds associated with deep sands, and are susceptible to loss if declining rainfall and increased evapotranspiration leads to a decline in soil water levels deep in the soil profile. Plant population modelling studies suggest that the combination of fire and decreased rainfall may lead to the loss of fire-sensitive (non-sprouter) woody plant species and decline even in fire-tolerant (resprouter) species.

The moist forest and wetland ecosystems of south-western Australia are likely to be among those most reduced in extent by projected warming/drying under climate change as the high rainfall zone contracts southwards.Wet habitats contain disproportionately high numbers of fire-sensitive plant species, which are also slow to reach reproductive maturity, so that they are among the most susceptible to any increase in drying and the occurrence of fire.

A drying climate will reduce the areas of suitable habitat available for occupation by wet forest types such as tingle and karri. A direct consequence of climate change may be increased recruitment failure for wet forest tree species, and increased (drought-related) mortality in mature trees. Fire may combine with climate change to drive such systems more quickly from one state (e.g. karri forest) to another (e.g. jarrah) by increasing levels of mature tree mortality and opportunities for invasion by other species. Dense riparian zone vegetation important as refuge for animal species may become more vulnerable to fire, presenting new challenges for conservation management.

Recent tree health decline among several *Eucalyptus* (especially *E. gomphocephala* and *E. wandoo)* and *Banksia* species in south-western Australia has received much attention, with drought and pathogens as potential causes. Decreasing rainfall may further deplete soil water availability, while increasing temperatures and more frequent fire might facilitate the spread of soil fungal (dieback) pathogens such as *Phytophthora cinnamomi*, thus exacerbating these and similar problems.

The potential effects of projected climate change on fire regimes and biodiversity for major ecosystems in south-western Australia are of great concern. Steep climate gradients and limited extent of sensitive ecosystems, such as wetlands and wet forests, makes them highly vulnerable to loss under a hotter and drier climate future. Any increase in the frequency of fire, associated with more high fire danger weather days, may drive systems more rapidly to new states. However, potential responses are complicated by uncertainty concerning the extent to which increased plant water use efficiency associated with higher $CO_2$, and potentially limiting soil nutrients, might affect biomass production and fuel loads.

### 6.3.1 Introduction

The South-West Botanical Province of Western Australia is one of 25 global biodiversity hotspots recognised by Myers *et al.* (2000), and the only one in Australia. It covers an area of approximately 300 000 km$^2$ and is home to at least 7380 recognised plant taxa – with about 50% endemic to the region (Hopper and Gioia 2004) – and including many highly speciose genera (Cowling and Lamont 1998). Malcolm *et al.* (2006) modelled potential biodiversity change under doubled $CO_2$ for major global vegetation types, and found high variability among hotspot regions. One of the most sensitive regions was south-western Australia, with a projected loss of up to 2000 species over the next 100 years (results depend on scenario assumptions). The SWBP is thus intrinsically important as a reservoir of biodiversity with significant conservation values. Its flora also has high economic value as an ecotourism asset. Threats from climate change – as well as those as a result of altered fire regimes, land clearance and fragmentation associated with agriculture, minerals exploration and exploitation – highlight the need for sound scientific understanding and management of these plant species and communities as a priority for monitoring and research on climate change impacts on Australian vegetation.

In their review of climate change implications for Australia's national reserve system, Dunlop and Brown (2008) identified potential key threats for each of the major climate–vegetation regions of the continent. They placed south-western Australia wholly within a single, Mediterranean-type region based on the landscape classification approach of Hobbs and McIntyre (2005). The region encompasses wet and dry forests, woodlands, mallee and shrubland vegetation. The major threats from climate change were suggested as increased impacts from altered fire regimes, invasive species, and declining water availability – leading to potential changes in vegetation structure, function and composition, and loss of sensitive species.

Climate trends for some parts of south-western Australia have seen maximum temperatures increase by 0.15–0.20°C per decade over the period 1900–2007 – one of the fastest rates in Australia – and rainfall decrease abruptly by 10–20% since the 1970s (CSIRO and Bureau of Meteorology 2007). Climate change projections to 2070 – based on a number of GCMs and assuming a medium emissions scenario – project further temperature increases of 1–3°C across all seasons of the year, and a 10–20% decrease in rainfall, with greater likelihood of decrease in winter than in summer (CSIRO and Bureau of Meteorology 2007). These projections imply a hotter, drier climate with more high fire danger days and shorter intervals between fires (Cary 2002), a projected scenario that is apparent for southern Australian in general.

Higher temperatures are projected to lead to increased plant stress in summer – but warmer winters and an earlier start to spring – with potentially longer growing seasons, if water is available. Projected decreases in winter rainfall (continuing a strong trend identified in the region over the past 30 years) may mean that soil drying through spring is faster, with greater impact on late spring/summer growth and survivorship. Summer rainfall projections are very uncertain, but indicate either little change or a possible increase in rainfall due to greater frequency of rain events associated with tropical lows (decaying cyclones tracking southwards down the west coast and upper atmospheric low pressure troughs). However, likely increases in evapotranspiration due to higher temperatures are estimated to be equivalent to 10–20% of annual rainfall so that, overall, water availability is likely to decrease substantially over the course of the 21st century. Severe drought may also become more common, leading to plant mortality events (as observed in the Geraldton sandplain region after two years of below-average rainfall in 2006 and 2007) and increasing the risk of fire propagation in very young vegetation as dead fuel load is rapidly increased.

Complicating further the future scenarios for the region are uncertainties concerning the role of $CO_2$ fertilisation. While increased $CO_2$ may increase water use efficiency, with potential increases in plant productivity, there is very little field-based experimental evidence for native tree and shrub species from southern Australian sclerophyll plant communities, such that the effects of increased $CO_2$ in water-limited and nutrient-poor ecosystems is unknown and in urgent need of investigation.

While little research has explicitly addressed the likely impacts of climate change and fire regime change on south-western Australian ecosystems, the high biodiversity values of this region make at least a preliminary assessment essential in order to identify potential major impacts and areas for urgent scientific investigation and management policy development. The IBRA ‘biogeographic regions’ (Thackway and Cresswell 1995; Australian State of the Environment Committee 2001) defined on the basis of climate, landscape and vegetation properties, are used here to delineate major environments and their characteristic ecosystems within south-western Australia. Particular attention is focused on the shrublands of the Geraldton Sandplain region, and forests of the Jarrah Forest and Warren regions (Fig. 4.13).

### 6.3.2 Mediterranean-type shrublands of the Geraldton Sandplain region

The biodiverse northern sandplain shrublands (kwongan) represent a biodiversity hotspot within the broader SWBP hotspot. This ecosystem lies in the ‘transitional’ rainfall zone (sub-humid to semi-arid; 400–600 mm/year rainfall), characterised by a particularly high estimated speciation rate, probably relating to selective pressures exerted by major shifts in rainfall over the glacial–interglacial cycles of the past few million years (Hopper and Gioia 2004). The region is characterised by a strong richness gradient – decreasing from south-west to north-east – in response to steep rainfall (decreasing) and temperature (increasing) gradients, so that many species reach their northern (hot, dry climate) geographic limits in this zone. Plant species distributions are also constrained by substrate type, with sandplain shrublands bounded by communities on coastal limestones (to the west) and laterites (to the south, east and north), which ultimately limits their capacities to migrate in response to changing climate. Thus, it is a region that is very vulnerable to future climate change.

Low shrublands on literitic substrate near Eneabba WA
Source: Neal Enright

Shrubland fuel dynamics differ from those of other major structural vegetation types. In forests and woodlands, surface fuel load is a key driver of fire regime; while in grasslands, grass biomass and curing is critical. In shrublands, surface fuel loads are often low and discontinuous, and surface fires generally cannot be sustained. Rather, the live above-ground biomass represents the critical component of fuel for fires, providing continuity of fuel load under appropriate (particularly dry, windy) weather conditions. Biomass levels vary markedly between shrubland types, ranging from around 5 t/ha in low heaths (<1 m height) to 15–20 t/ha in tall shrublands on deep sands (Bell *et al.* 1984). The dominance of resprouter species (60–85%; Enright *et al.* 2007) means that cover and biomass levels return to pre-fire values within 5–10 years (Bell *et al.* 1984). Surface fuel load is slow to accumulate relative to live biomass due to conservative strategies of nutrient use that result in sclerophyllous leaves retained for at least several years for most perennial species (Midgley and Enright 2000). Thus, mean fire interval is typically short, e.g. averaging 13 years since the 1970s for shrublands of the Eneabba sandplain (Miller *et al.* 2007) – and large fires can occur at intervals <10 years under extreme fire-weather conditions. This is particularly true for tall shrublands on coastal limestones, where higher soil nutrient levels support faster growth rates and higher plant cover than for shrublands on low nutrient acid sands and laterites. Recent experiments in young shrubland stands have shown that fires can propagate in three- to four-year-old calcareous shrublands, and in four- to five-year-old sandplain shrublands (at least for the small areas treated in these experiments) under warm, dry and windy conditions (N Enright, V Westcott and J Lade, unpubl. data). At these very short fire intervals, most fire-sensitive (perennial, non-sprouter) species either had not yet flowered, or had flowered but not set seed. In addition, most woody resprouters had not begun to set seed again, or had set few seeds.

Ecophysiological studies have illustrated that the large shrub genera (e.g. *Banksia*), which make up most of the biomass in these shrublands, require access to water over summer from deep in the (sandy) soil profile in order to survive and grow (Lamont *et al.* 1989; Enright and Lamont 1992a). Where sand depth is shallow (<1 m), soils dry out over summer and cannot support large shrubs, such that they are excluded, and a low heath dominates such parts of the landscape. Seedling survival in the first year after fire in *Banksia attenuata* has been correlated with levels of soil moisture in the near-surface soil over summer, with lower survivorship in sites where soils were already very dry by early summer (Enright and Lamont 1992b). Enright and Lamont (1992b) concluded that recruitment in this resprouter banksia was most likely in years when above average rainfall (especially summer rain) occurred in the first year after fire, and was unlikely in years of below average rainfall (later defined as <20% below mean annual in Enright *et al.* 1998a) – such years typically also having dry summers. In addition, lignotuber mass developed more slowly in seedlings (up to five years) in the driest sites, perhaps because resource allocation required continued tap root growth at the expense of lignotuber growth in order to ensure adequate access to water. In a similar study in the Californian chaparral, Fraser and Davis (1988) reported greater first-year mortality in seedlings of a common resprouter species, *Rhus laurina,* than of the non-sprouter *Ceanothus megacarpus,* concluding that the resprouter was more sensitive to drought stress. Richards *et al.* (1997) showed that survival of needle- and terete-leaved *Hakea* species was better than that for broad-leaved species under two consecutive years of drought, with plant death associated with higher transpiration rates, and little evidence of compensatory trade-offs such as higher fecundity. *Enright* and *Lamont* (1992a) and *Lamont et al.* (1993) similarly reported higher survivorship of needle-leaved *Banksia leptophylla* and *Hakea.psilorrhyncha* seedlings than of co-occurring broad-leaved *Banksia* seedlings in their first year of growth (in a post-mining rehabilitation site, and after fire in a natural shrubland site, respectively), confirming that such species are likely to fare better than broad-leaved species under a drying climate. Groom (2004) has shown that shallow-rooted small shrubs can vary markedly in their summer water use, with some species likely to suffer markedly from decreased soil water availability while others show considerable tolerance to drought. Groundwater-dependent, deep-rooted shrub species also show decline where groundwater levels have dropped. Groom (2004) reported that such species are increasingly at risk once groundwater depth exceeds 8 m, and Enright and Lamont (1992a) identified rapid rates of seedling root elongation to keep pace with seasonally retreating groundwater level as critical in the establishment and survival of such species.

The few published demographic modelling studies of non-sprouter, large shrub species are consistent in reporting potential declines in population viability in relation to decreased rainfall and increased frequency of fire. Burgman and Lamont (1992) reported increased first-year seedling survival and growth in *Banksia cuneata* when watered over summer, and suggested that population maintenance was closely linked to fire regime and post-fire-weather conditions. Based on the results of a single-species, non-spatial computer simulation model, they concluded that 'if, over the next 50 years, there is a reduction in average rainfall, the chances of persistence of the population are low, even in the absence of prescribed fires' (p 719). Enright *et al.* (1998a, b) explored the potential impact of below average (80% of median annual) rainfall in the first few years after fire in combination with different fire intervals on the demography of the non-sprouter *B. hookeriana*, and resprouter *B. attenuata*, showing similar negative impacts on population size and viability, with rapidly declining populations of the non-sprouter at fire intervals ≤8 years. Short fire intervals also led to decline in the resprouter species, with recruitment insufficient for parent replacement in the long term.

While studies such as those above indicate something about the possible non-interactive effects of changes to rainfall and fire frequency on shrubland species, they are wholly based on present-day responses of plants to drought and/or fire. However, future responses will incorporate a full range of interaction effects between all of the changing components of climate (including $CO_2$) and fire regime. Rates of seedling growth and resprouter regrowth after fire may decline under more frequent fire due to lower soil moisture availability and resprout vigour (bud bank and energy resources), causing reduced rates of biomass accumulation, and possible reductions also in seed production. Llorens *et al.* (2004) reported contrasting growth changes in two dominant species of a Mediterranean shrubland submitted to experimental drought and warming, and found decreased stem elongation and diameter in *Erica multiflora* and *Globularia alypum* as a result of decreased soil moisture (-22%), but varied response between species to increased temperature (0.7–1.6°C). Overall, decreased biomass production and potential shifts in competitive hierarchies were suggested as a consequence of climate change. Combined with predicted lower rainfall, any increase in fire frequency would further exacerbate reductions in recovery/growth of resprouters and of growth/survival of non-sprouters. For example, Croft *et al.* (2007) found that the combination of fire followed by drought depleted the vigour of lignotubers in seven tree and two shrub species in eastern Australia, leading to increased mortality. In 2006–2007 there was massive cohort die-off of *Banksia hookeriana* on sandy dunes in the Eneabba sandplain shrublands of south-western Australia, possibly due to depletion of groundwater. Such events may become more common.

*Banksia hookeriana* experienced a cohort die-off in 2006-2007 on sandy dunes in the Eneabba sandplain shrublands of south-west of Western Australia from a combination of fire followed by drought.

Source: Neal Enright

Study of the water relations of woody plant species on sandplain dunes and their demographic responses for dunes of different size (a surrogate of different soil water availabilities; Enright and Lamont 1992b) may provide important data on likely species responses to a drying climate. However, the interactive effects with fire regime are more difficult to explore and may need to be approached through the use of a variety of research methodologies, including simulation models incorporating new information on the ecophysiological responses of different plant functional groups to the various components of climate change.

The effects of increases in temperature have received less attention than those for decreases in rainfall. Musil *et al.* (2005) studied the effects of increased summer daytime temperature on survival of succulent perennials in South African Karoo, and revealed greatly increased mortality (two to five times above control), and lower plant starch and chlorophyll levels, over a four-month period in summer. They suggested that current thermal regimes may be close to critical limits for many species. Many shrubland species reach their northern range limits in the northern sandplains of south-western Australia, suggesting climate rather than substrate as the controlling factor on distribution; or that they are 'new' species which have not spread far from their points of origin – in which case both their environmental tolerances and dispersal capacities may be limited. Increases in temperature and decreases in rainfall alone may result in significant range contraction in some such species, reducing overall biodiversity in the northern part of the sandplain.

Elsewhere in southern Africa, Foden *et al.* (2007) presented evidence that a changing climate is eroding the geographical range of the Namib Desert tree *Aloe* through population declines and dispersal lags, with higher mortality in *Aloe dichotoma* trees near its northern range limits and healthy populations at southern limits, but no evidence of compensatory southward migration. Analyses implicate climate change as the driving factor and support conservative estimates of species migration rates for the species. In another study, de Dato *et al.* (2006) observed increased night-time temperature and decreased annual rainfall in a Sardinian shrubland over a three-year period, and reported decreased growth and decomposition rates in relation to reduced moisture, but no effect for temperature.

There are several implications of these findings for the northern shrublands of Western Australia. Any interaction of climate change with fire regime that leads to more frequent fire in the northern shrublands, or slower recovery (vegetative and reproductive) following fires, may both hasten, and increase the magnitude of, such range contractions. In the case of the northern shrublands, a lack of suitable substrate conditions will constrain any southward migration, which might allow species to track shifts in the distribution of appropriate climates. This increases the urgency for development of a sound understanding of the projected changes to fire-weather as a consequence of climate change in south-west Western Australia, and the likely fire regime consequences that may accompany changes to fire-weather.

Fuel reduction burn in shrublands of south-west Western Australia

Source: Neal Enright

The interactive effects of changes in temperature, moisture and $CO_2$ on nutrient supply, mineralisation and nutrient use efficiency are unclear. Biomass accumulation in these shrublands is strongly dependent on the growth of tall shrub species (mostly species of *Proteaceae* and *Myrtaceae*) and their access to adequate supplies of soil water. Maximum stem elongation in *Banksia hookeriana, B. attenuata* and *B. menziesii,* the three most abundant large shrub species of the shrublands, occurs in summer (Lamont and Bergl 1991), while surface fine root growth occurs in winter when the surface soil is kept continuously moist by winter rains. This summer growth pattern reflects a tropical evolutionary heritage that is likely shared by many other genera of *Proteaceae*. Any decrease in water availability caused by a combination of lower winter rain and higher summer evapotranspiration is likely to impact more heavily on summer-growing species than on spring-growing species – a circumstance that may enhance threats by the incursion of spring-growing invasive species.

Patterns of resource capture and allocation among plant parts in *B. hookeriana* show that relative resource allocation to reproductive effort is highest for phosphorus (P) (91 – for proportion of nutrient allocated to reproductive structures (mostly in seeds) versus dry mass of those structures) and nitrogen (N) (44), declining to <6 for potassium, calcium, magnesium and sodium (Witkowski and Lamont 1996). This reflects the limiting role of P (and to a lesser extent N), but substantial capacity to build biomass when water is not limiting, in the very nutrient-poor Mediterranean ecosystems of Australia. Changes in nutrient supply and availability have been reported for experiments in relation to future climate change, but directions of change in plant available N and P are not consistent between studies. For example, Sardans *et al.* (2007) found differing effects of warming versus drought on plant and soil P dynamics in a Mediterranean shrubland;

warming led to increased plant P due to higher biological activity and decreased soil P, drying reduced P in plants (lowered demand) increasing soil P but much in non-available form. If levels of P and N tied up in biomass were to increase due to increased leaf longevity and slower decomposition under future climate, then the pulses of nutrients released by fire would increase the temporal variability in nutrient availability and again might favour winter-growing competitor species (including invasive species) over summer growing tolerator species.

Parsimony suggests that we should conclude only what is most reasonable given the evidence available to us at the moment, and at this stage we cannot say with any certainty how fire regime will change in the Mediterranean-type shrublands of Western Australia as a result of future climate change. A number of conclusions can be drawn concerning the potential effects of individual factors, but these may change once interactions (and new findings) are taken into account:

- A hotter, drier climate suggests reduced biomass production, which may reduce the frequency of fires (while rates of surface fuel load accumulation are of limited relevance).
- Such changes in climate will lead to more high fire danger days and a longer fire season, promoting more frequent fire.
- While winter rainfall is projected to decline, summer rainfall may remain unchanged or even increase; any increase in summer precipitation might be accompanied by more frequent lightning, bringing with it an increase in potential ignitions when fire danger is generally at its highest.
- Competitive hierarchies are likely to change, as a consequence of a hotter, drier climate, leading to shifts in species compositions and abundances, with heavy water-user species (e.g. large, broad-leaved shrubs) perhaps disadvantaged relative to species with stronger drought tolerator attributes. More frequent extreme drought events may exacerbate shifts in species composition.
- Higher $CO_2$ levels may increase water use efficiency – at least partly offsetting warming and drying impacts on biomass production. However, internationally, evidence on this point is contradictory. For example, in the USA, mesquite showed enhanced seedling growth rate under $CO_2$ fertilisation, leading O'Polley *et al.* (2006) to conclude that it may expand its range and abundance under climate change. Pataki *et al.* (2000) found that reductions in stomatal conductance under increased $CO_2$ may not occur in *Larrea* and *Ephedra* based on their Mohave Desert study.
- Nutrient use efficiency, decomposition rates and the cycling of limiting macronutrients (especially P and N) may change, but the evidence for this is unclear. Any increase in N and P in litter may result in higher pulses of nutrients after fires, which may encourage invasive species. Implications are unclear and dependent upon the balance of climate changes in terms of temperature versus rainfall.

### 6.3.3 Other ecosystems

The projected impacts of climate change on fire regime and biodiversity are not necessarily uniform across the seven biogeographic regions of the SWBP and major ecosystem types within them (Fig. 4.13). All are likely to be adversely affected by a drying climate – particularly those in the higher rainfall south-west of the region due to likely reductions in overall geographic range as the area of humid climate contracts. Even more vulnerable to fire regime shifts associated with climate change may be those vegetation types associated with specialist habitats – in particular, peatlands and other organic soil communities (including wetlands and wet heaths), and granite outcrop communities. These habitats support many rare, and often fire-sensitive, species and will be particularly susceptible to loss from any increase in fire frequency and/or intensity that might accompany a drying climate. Among 209 rare and threatened taxa from south-western Australia with known fire response, Yates *et al.* (2003) found the proportion of non-sprouter to sprouter species was extremely high (0.66) compared with reported proportions for all species of the major vegetation types including jarrah forest (0.25) and northern kwongan (0.25).

Species rich southern kwongan vegetation situated east of Lake King, Western Australia.
Source: Angas Hopkins

This may suggest a greater risk of extinction in fire-sensitive (non-sprouter) than in fire-tolerant (resprouter) species for the SWBP as a whole, and a greater risk of multiple losses (and ecosystem transformation) in plant communities with high proportions of non-sprouters, such as granite outcrop vegetation (0.6) and southern kwongan (0.5). Burrows and Abbott (2003) used a geographic information system to map the distribution of possible refugia for part of the Swan Coastal Plain Jarrah Forest near Perth, and for the Warren (karri forest) Region. They defined refugia to include wetlands of all types, granite outcrops, south-facing slopes >10° (moister due to lower heat load, and assumed less fire-prone), and reaches of rivers and streams where riparian vegetation was well protected from fire (e.g. by cliffs) (Fig. 6.9 a, Fig. 6.9b). Burrows and Abbott argued that similar parts of the landscape may have been important refugia for plant and animal species during past periods of aridity, and could act as such again in the future – and so are of extreme conservation significance; Low (2008) referred to such potential refugia as 'cool sites'. Byrne (2008) reviewed phylogeographic evidence for plant and animal refugia in southern Australia and concluded that, over recent glacial–interglacial cycles, responses to climate change have involved persistence and resilience in local refugia more than large-scale migrations in the face of changing environmental conditions, further highlighting their potential significance. Representing many small and fragmented patches across a large region, these refugia pose major management problems if such sites are to be managed to exclude/reduce fire and other potential impacts of climate change.

### *Karri and jarrah forests of the Warren and Jarrah Forest regions*

The karri (*Eucalyptus diversicolor*) and jarrah (*E. marginata*) forests of the Warren Biogeographic Region and Jarrah Forest Biogeographic Region (respectively) are characterised by moderate to extreme surface fuel loads; up to 58 t/ha after 25 years in wet karri and 20–25 t/ha in wet jarrah forests (Sneeuwjagt and Peet 1985; section 4.3.3). Rates of surface fuel load accumulation with time since fire decrease with both decreasing rainfall and increasing potential evapotranspiration from south to north and west to east. While the possible water use efficiency benefits of increasing $CO_2$ may partly offset the impact of lower water availability on biomass production, we must assume (for now) that fuel accumulation rates will relate to future rainfall in the same way that they do today. In south-western Australia, critical threshold levels for surface fuels in relation to fire suppression effectiveness are 8–9 t/ha in jarrah forest, and 19–20 t/ha in karri (McCaw and Burrows 1989; N Burrows, pers. comm.; Burrows 2008), reflecting the fact that karri fuels are typically wetter. In karri forest, the 19–20 t/ha level is reached within 3–10 years of last fire (depending on forest type and site productivity), and in jarrah forest, the 9–10 t/ha level is reached within 5–15 years.

Old growth karri (*Eucalyptus diversicolor*) forest of the Pemberton region, south-west of Western Australia.
Source: Angas Hopkins

Under a 20% rainfall reduction, these levels take 10 to >25 years to reach in karri (although based on annual rainfall we project that only the drier karri forest fuel load levels will occur), 8 to >25 years in jarrah, and probably never in wandoo (*E. wandoo*). The reduction in fuel load accumulation rates and its potential effect on fire interval is much greater than might be expected in relation to the magnitude of the hypothesised reduction in rainfall. If correct, this would markedly change the fire dynamics of the region, with high fire danger weather increasing the risk of conditions conducive to rapid fire spread and high intensity fires on the one hand, but slow fuel accumulation rates limiting the availability of fuel for fires, such that fires may not propagate or may be more readily suppressed on the other.

Wardell-Johnson *et al.* (2007) investigated the relationship between number of fires since the 1970s, time since last fire, habitat properties and biodiversity for a range of karri and jarrah forest types, and provided some insights into possible impacts of changed fire regimes. They found that time since last fire was more important than number of fires in determining species richness and composition in karri forest due to the strong effect of fire ephemerals. They considered karri forest to be relatively resilient to impacts from increased frequency of disturbance due to its intrinsically lower species richness, limited presence of invasive species and rapid recovery of dominant natives (Wardell-Johnson *et al.* 2004). In contrast, jarrah forest communities had higher species richness and showed decreased richness in sites with increasing fire frequency. Long-term monitoring of 0.5 ha plots in jarrah at Lindsay Forest since the 1970s showed a decline in the abundance of some fire sensitive species in the most frequently burned plots (seven fires in the past 31 years).

More frequent fire in jarrah and karri forests may also adversely affect habitat for ground-dwelling animal species, such as the quokka (*Setonix brachyurus*), which requires a mosaic of recently burned vegetation for browse, and longer-unburned thicket areas, typically riparian, for refuge (Burrows 2008). Current management seeks to minimise the risks of fire spreading through the riparian zone by burning upland areas under cool spring conditions, when fires will self-extinguish upon contact with moist riparian vegetation.

Under warmer, drier climate conditions, control of fires in riparian zones may become more difficult. Increased frequency of forest fires may also lead to a reduction in the size (stem diameter and height) of trees, impacting on the size and number of tree hollows for use by fauna. However, Whitford and Williams (2002) found that small- to medium-sized trees, at 40–80 cm diameter at breast height, held 50% of all hollows; hence a decline in large trees may not necessarily mean a reduction in the number of hollows, especially if more small trees occupy the space of lost large trees. Nevertheless, size of hollows may also be important.

The quokka, *Setonix brachyurus* requires a moasiac of recently burnt vegeatation for browsing and longer-unburnt thicket for refuge.
Source: John Cancalosi/AUSCAPE

The demographic attributes of plant species largely determine their vulnerability to changes in climate and associated fire regimes. As noted in section 3.2, fire sensitive (non-sprouter) species are particularly at risk to changes in the interval component in fire regimes, as individuals must reach reproductive maturity and accumulate a sufficient store of seeds (in either a soil or canopy seed bank) for self-replacement after fire, otherwise populations will decline. Most understorey perennial species in jarrah forest flower within two to three years, and all within four years, of fire according to Burrows and Friend (1998) and Burrows *et al.* (2008). Some non-sprouter species with juvenile periods of up to six to eight years are associated with restricted mesic habitats where fire is infrequent (Burrows and Wardell-Johnson 2003; Burrows *et al.* 2008). However, time to first flowering implies nothing about seed production and storage; Gill and Nicholls (1989) recommended a 'rule of thumb' of 2 x the juvenile period as the minimum fire interval likely to ensure sufficient seed production and storage for parent replacement. Burrows and Wardell-Johnson (2003) further noted that the length of the juvenile period for these species is longer in the drier northern parts of species ranges by up to 1.5 years. Provenance variation in jarrah shows trees of drier areas are smaller and slower growing, while those of wetter areas are faster growing in their own locale but suffer higher seedling mortality when grown in drier sites (O'Brien *et al.* 2007). A clear conclusion from this is that with reduced rainfall under future climate change, juvenile periods will be longer and non-sprouter species will be more vulnerable to population decline in relation to short fire intervals.

Pertinent also is the strong representation of serotiny (canopy seed storage; Lamont and Enright 2000) in the flora of south-western Australia – higher than anywhere else in the world (Lamont *et al.* 1991). Serotinous, non-sprouter species (particularly prevalent in the families *Proteaceae* and *Myrtaceae*) may be more vulnerable to shortened fire interval and poorer conditions for recruitment after fire (due to reduced rainfall) compared to species with soil seed storage. Although all seeds for serotinous species must establish in the first year following dispersal (usually after fire) or perish, this is not the case for soil-stored seeds – where some may remain dormant in the soil through more than one fire cycle, so that absence of the species from the extant vegetation does not necessarily mean propagule loss as well.

Both widespread (e.g. karri, jarrah, marri, wandoo) and narrow endemic eucalypts (e.g. tingle species) of the Warren Region recover vegetatively from crown or basal epicormic buds after moderate- and high-intensity fires, but patterns of recovery and regeneration differ, so that the impacts of climate change might also vary among species. Wardell-Johnson (2000) found recruitment was high in canopy gaps for yellow tingle (*E. guilfoylei*), high under the canopy in marri, no different in either microsite type for karri and red tingle (*E. jacksonii)*, and consistently low in Rates tingle (*E. brevistylis*). He concluded that low levels of regeneration in Rates tingle and high bole damage in red tingle (related to intensity of fires, especially under dry conditions) may be a concern under more frequent fire. However, under a drier climate, the wet forest habitats occupied by the local endemic eucalypts may be lost in any case, and changes in canopy openness may have relevance only to the relative abundance of the common species: marri, jarrah and karri. Rate of change may be most rapid for ecotonal areas where jarrah–marri communities are close to karri, providing a significant seed source for invasion following fires as drying proceeds. Where seed sources are distant the course of change may differ – with karri forests persisting longer, but becoming more open and depauperate as recruitment failure becomes acute.

Changes in the length of the fire season might also impact on recruitment among eucalypt species. McCaw et al (2000) noted little regeneration in red tingle after a low-intensity spring fire, but prolific regeneration after a hot autumn fire – probably due to better opening up of the canopy and understorey, and better seed bed conditions. Similar results were found by Burrows *et al.* (1990) for wandoo. However, Burrows and Friend (1998) reported that seedlings established after spring fires may not have sufficient time to establish to survive through the first summer relative to post-autumn fire recruits, which have the whole winter–spring period to establish. Thus, any increase in the average length of time between fire occurrence and seedling emergence (due to earlier arrival in spring of conditions for fire occurrence, and later arrival in autumn/winter of temperature and moisture conditions conducive to germination and establishment) would adversely impact upon seedling survivorship.

In a study of germination and seedling responses to water availability, Schutz *et al.* (2002) found that two larger-seeded, deep-sand *Eucalyptus* species (*E. macrocarpa, E. tetragona*) germinated earlier/faster compared with two smaller-seeded, laterite/loam species (*E. loxophleba, E. wandoo*). *E. tetragona* was the most drought tolerant overall, and root growth was faster in the sand species in moderately stressed soils (-0.5 megapascals, MPa) compared with moist soils (0.1 MPa). They concluded that large seed size may bestow an advantage on young seedlings in rapidly drying soils over summer. Changes in rainfall seasonality are also potentially important with a likely decrease in winter rain, but perhaps no change, or even a slight increase in summer rain. Burgess (2006) tested response to summer rain (as measured by sap flow) in species from four genera common to south-western Australia: *Eucalyptus wandoo* showed no response due to reliance on antecedent soil water, Christmas tree (*Nuytsia floribunda*) showed a delayed response with no change to water uptake rate for two weeks, *Allocasuarina campestris* showed a minor, twofold increase in water use (due to partial summer dormancy) and *Isopogon gardneri* (a small proteaceaous shrub) showed a rapid response with up to a fivefold increase in water use. Again, different species may be advantaged by such seasonal shifts, and in this example, shorter juvenile stage shrubs (able to cope with shorter fire intervals) were the species most readily able to respond to summer rains.

In relation to nutrients, an N and P addition experiment in jarrah forest by O'Connell and Mendham (2004) reported an increase in P stored in litter, but no change in decomposition rates or total litter store after five years. Thus, climatic factors appeared to be controlling the rates of decomposition and cycling. However, fire may release large pulses of P (and N) for uptake, and for use by other (invasive) species.

### *Swan Coastal Plain region*

Forest decline phenomena and invasive species are major concerns for the dominant plant communities of the Swan Coastal Plain – *Banksia* (particularly *B. menziesii*) woodlands and tuart (*Eucalyptus gomphocephala*) forest – in relation to climate change and fire regime. Decline in crown health of tuart (*E. gomphocephala*), wandoo (*E. wandoo*) and coastal woodland Banksia species over recent decades have become sources of major management concern and new research investigation in south-western Australia. Although the causes remain largely unknown, a drying climate and falling groundwater levels – in combination with soil and stem decay-causing fungi (Hooper and Sivasithamparam 2005) – are variously hypothesised as the major causes of mature tree dieback and death. At the same time, dense swards of invasive grasses limit seedling recruitment during both inter-fire and post-fire periods. The fuel continuity provided by winter-growing invasive grasses and high potential ignitions due to human population concentration in this region suggests that fires may become more frequent, deep-rooted large shrub and tree species will suffer from falling ground water levels, and many native small shrub and herb species will suffer increased competition from invasives for declining surface soil water resources in both inter-fire and post-fire environments.

### *Esperance Plains region*

Future reductions in growth rates and biomass may be less severe, on average, in the southern than in northern kwongan (shrublands), as lower temperatures in south coastal south-western Australia mean potential evapotranspiration has less impact on overall water availability. However, increased incidence of extreme heat or drought events may be important. Further, dieback of plant species susceptible to the root-rot fungus, *Phytophthora cinnamomi,* may increase if increasing temperature increases the duration in spring and autumn when soil moisture levels are conducive to fungal reproduction. Moore *et al.* (2007) reported increased impacts of *Phytophthora* dieback on high-diversity shrublands in the Stirling Ranges National Park for areas recently burned (within the last six years) relative to sites long unburned (>10 years). Rates of infection were 13–21% in recently burned sites compared with 1–11% in long-unburned sites, while plant densities in recently burned sites were 33–66% lower. Most species were non-sprouters (95 of 152 species; 63%) with many *Proteaceae*, *Ericaceae (Epacridaceae)* and *Papilionaceae*, along with *Xanthorrhoea platyphylla*, the most adversely affected, and *Myrtaceae* the most resistant. Results suggest that an increase in fire frequency may favour pathogen spread – particularly under mild, moist climatic conditions, which are more likely in southern shrublands and forests, than in central and northern areas of south-western Australia. Species restricted to mountain-top habitats in the Stirling Ranges may also be under threat from increasing temperature and decreasing moisture (including reduced frequency of cloud and mist), with no suitable refugia likely to be available in the future.

Bluff Knoll, Stirling Range National Park, showing death of jarrah as a result of *Phytophthora* dieback infestation.
Source: Ray Wills

Several rare bird species – including the bristlebirds (western and rufous), noisy scrub-bird and western ground parrot – all show a preference for shrubland and forest vegetation providing good cover, so that populations are threatened by any increase in frequency and extent of fire, which decreases the overall availability of dense vegetation. A combination of decreased rainfall and increased frequency of fire, producing on average younger vegetation and a slower recovery of vegetation cover, would adversely affect such species (Burbidge 2003).

#### *Mallee region*

This region extends beyond the semi-arid eastern edge of the south-western Australian botanical province. Rainfall records here do not show evidence of recent change, and a 350-year tree ring climate reconstruction for *Callitris columellaris* (Cullen and Grierson 2008) shows strong multi-decadal variability, but no trend in rainfall. Nevertheless, projected increases in temperature will likely result in reduced water availability for plant growth and more frequent high fire danger weather. Hopkins and Robinson (1981) reported the conversion of a *Eucalyptus* woodland (about 400 km east-southeast of Perth) to a mallee heath as the result of a single fire in a 40-year-old stand. They noted that these dry, open woodlands are characterised by very slow stand dynamics and cannot sustain fires at intervals $<100$ years without risk of structure change, concluding that fires at 50-year intervals or less will reinforce the mallee regrowth habit at the expense of tree habit. This to some extent supports the generalised impacts suggested for south-western Australia in Dunlop and Brown (2008), which suggests shifts from forest to woodland, woodland to shrubland, and shrubland to grassland in the Mediterranean climate-type regions of Australia in response to more frequent fire under a future drier climate. However, at this time there is no evidence to support the suggested changes for other vegetation types in south-western Australia.

#### *Avon Wheatbelt region*

Fragmentation is extreme in this region due to a long history of land clearance for agriculture, so that natural vegetation remnants such as salmon gum (*E. salmonophloia*) woodlands occur as small, isolated stands surrounded by agricultural fields. Natural fire is highly unlikely in these remnants – even if future climate increases the overall risk of ignitions and fire spread – as the probability of ignition within any remnant will remain extremely low.

The use of planned fire for biodiversity management across a region characterised by many small bush fragments is limited by cost and logistics, so that long intervals between fires may see the decline of some short-lived species. Most immediate concerns for these fragments are associated with the impacts of salinisation, weed species, grazing and loss of species interactions, e.g. pollinators (Hobbs 2002), all of which may be further affected by the direct effects of climate change – rather than on those of potential climate change-induced changes to fire regime.

### 6.3.4 Conclusions

The most likely consequence of climate change for south-west Western Australia is a climate that is warmer and drier, as is the case for most of southern Australia. On the basis of this, we predict that, for the ecosystems of south-western Australia, species will show reduced growth rates and slower rates of post-fire recovery. Communities are likely to show lower fuel accumulation rates, if temperature and moisture act alone, but may show more complicated responses if $CO_2$ fertilisation occurs and nutrient availability changes. In dry woodlands and shrublands, drought effects might more than offset $CO_2$ fertilisation effects. In wetter forest areas, growth and fuel conditions might be maintained due to greater water use efficiency, so that fire interval will shorten and impacts will be driven more by species responses to temperature than to moisture.

Table 6.1 provides an overview of possible climate change impacts on fire regime and other ecosystem properties for the major ecosystems of south-western Australia represented in the seven IBRA biogeographic regions (Australian State of the Environment Committee 2001; Thackway and Cresswell 1995; Fig. 4.13) of the SWBP defined on the basis of climate, landscape and vegetation properties.

In relation to fire we assume, on the basis of *Climate change in Australia* (CSIRO and Bureau of Meteorology 2007) and the implications for fire-weather of a warming/drying scenario in south-eastern Australia (section 4.2.3), more high to extreme FFDI days under a future climate scenario. The implications of this are longer fire seasons, a greater risk of escape of planned fires (due to higher likelihood of high FFDI in the days following prescribed fires) and fires able to propagate through younger, sparser fuels. Upward shifts in average FFDI and higher night-time minima may also allow fires to persist longer and so reignite under return to high FFDI conditions more often – unless post-burn mop-up is increased. There may be fewer planned fire days available in spring and autumn due to increased average FFDI (and most state agencies are already unable to meet their annual burn area targets; Esplin *et al.* 2003), but more days available in winter. However, there may be negative impacts on seed germination and seedling establishment (shortened winter/spring period for establishment), and on fauna. Thus, new approaches to landscape-level fire management may be required, e.g. more protection corridor and buffer burning, but less burning of blocks.

The considerable uncertainty surrounding climate change effects on fire regime, and consequently on biodiversity, highlights the clear and urgent need for well-designed and resourced monitoring (and experimental) programs at selected sites in major ecosystem types (and ecotones) of south-western Australia to provide baseline and change data that is comparable among sites to understand how systems will respond under climate change. Experiments investigating the effects of increased $CO_2$ on biomass production, nutrient cycling and fuel load accumulation in relation to rainfall in these ecosystems is particularly needed if the interactions among factors driving future fire regimes are to be understood.

Some suggested priority research needs in south-western Australia in relation to fire regime and climate change are:

1. Developing a better understanding of CO2 effects on vegetation
2. Meta-analyses of existing data to better understand the relationships between past fires and weather conditions in different vegetation types
3. Examination of the interaction between declining annual rainfall and elevated CO2 on forest productivity over the past 30 years
4. Long-term monitoring for major ecosystems (and ecotones) using a standardised set-up and monitoring approach – perhaps following a modified form of the Forestcheck system being used by Department of Environment and Conservation Western Australia in the Jarrah Forest region (Abbott and Burrows 2004). (http://www.dec.wa.gov.au/climate-change/climate-change-and-biodiversity/climate-change-wa.html)
5. Fire ecology of less well-known habitat type communities, especially vegetation of organic soils and granite outcrops
6. Demographic analyses of fire-interval sensitive local endemics in south-western Australia – especially non-sprouter species of restricted habitats and serotinous species
7. Analysis of functional type groupings, distributions, susceptibilities, etc. in relation to climate and fire for poorly understood plant groups such as the orchids and other geophytes (representing 7% of the south-western Australia flora)
8. Management response research – including Bayesian approaches to decision-making

**Table 6.1. Summary of potential fire regime (and other) consequences associated with future climate change in the major biogeographic regions (IBRA) of south-western Australia.** IBRA regions after Thackway and Cresswell 1995; Australian State of the Environment Committee 2001. The four fire 'triggers' or switches (section 5) are numbered (1–4). Potential shifts associated with climate change are indicated for all factors for the first region and are only shown for other regions where a different response is anticipated.

| | Geraldton Sandplains | Jarrah Forest | Warren | Avon Wheatbelt | Swan Coastal Plain | Esperance Plains | Mallee |
|---|---|---|---|---|---|---|---|
| Major ecosystem(s) | Northern shrublands | Jarrah forest | Karri forest | Wandoo and Salmon Gum woodlands | Banksia woodlands | Southern shrublands | Western woodlands |
| 1. Fuel Biomass | Decrease due to slower growth and survival rates | | Increase due to higher $CO_2$, moisture not limiting | | | May be little change | |
| 2. Fuel availability | Decrease of litter and live fuel continuity | | Increase, or no change | | | | |
| 3. Fire-weather | Hotter, drier, longer fire season, more summer rain? | | Hotter, drier | | | | |
| 4. Ignitions | Possibly more summer lightning | | No change | | | | |
| Invasive species | Limited risk, possible grass incursions from north under higher summer rain | Blackberry in riparian zones | Relatively resistant | Little change; already strongly impacted by agricultural weeds and edge effects | Grasses, e.g. *Ehrhardta* | | |
| $CO_2$ effect | Uncertain, but may not offset drying | | May offset drying | | | | |
| Phenology | Earlier flowering, drying, longer fire season | Earlier flowering, drying, longer fire season | Earlier flowering, drying, longer fire season | | | | |
| Disease | | | Increased *Phytophthora* risk due to longer reproductive period | | | Increased *Phytophthora* risk due to longer reproductive period | |
| Sensitive habitat types | Lake and swamp systems | Granite outcrops, wetlands, organic soil communities | Peatlands, wetlands, wet heaths, granite outcrops | Granite outcrops | Wetlands | | |
| Distribution | Reduced, constrained by substrate and fragmentation | | Reduced by loss of small outliers and edge erosion | | | | |
| Protected areas* | 14.0% | 8.1% | 8.1% | 0.5% | 8.1% | 19.3% | 19.3% |
| Over-all prognosis | Species losses and habitat range contraction | | Habitat range contraction for wet forest and wetland vegetation types | Little change due to strong control on fire regime of fragmentation | | | Shift in structure and composition from mallee to heath? |

* Estimates are averages for larger regions as listed in Australian State of the Environment Committee (2001).

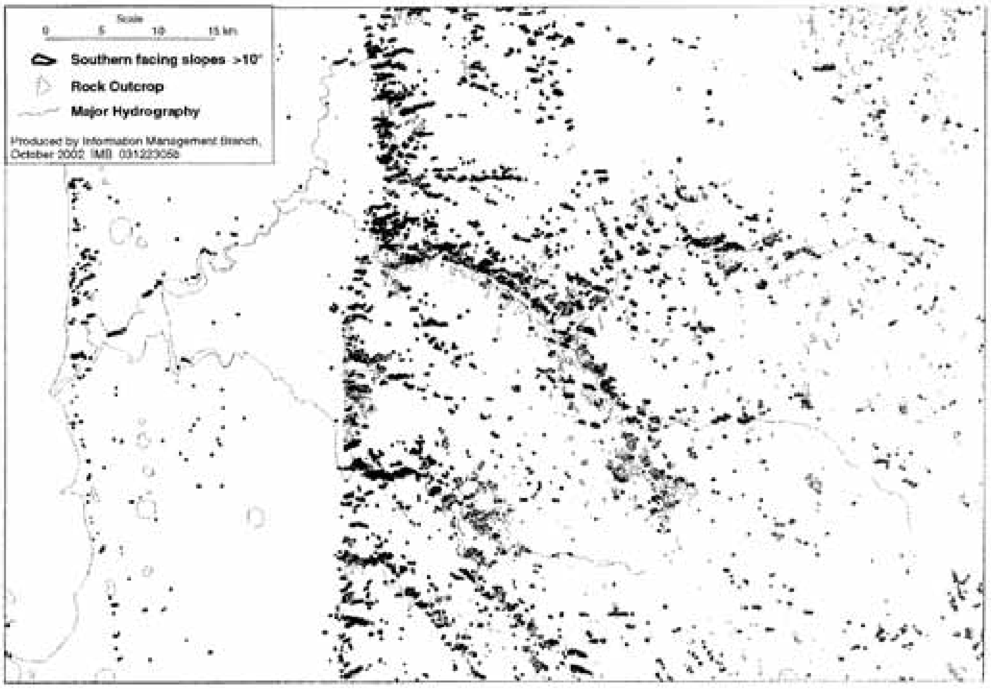



**Fig. 6.9a. Locations of potential plant species refugia on the Swan Coastal Plain region near Perth, and the central-western portion of the Jarrah Forest region of Western Australia.** Source: Burrows and Abbott (2003, p. 443). Potential refugia associated with south-facing slopes and riparian zones are concentrated along the edge of the Darling Scarp and major river systems, while granite outcrop refugia dominate further east in the Jarrah Forest region.

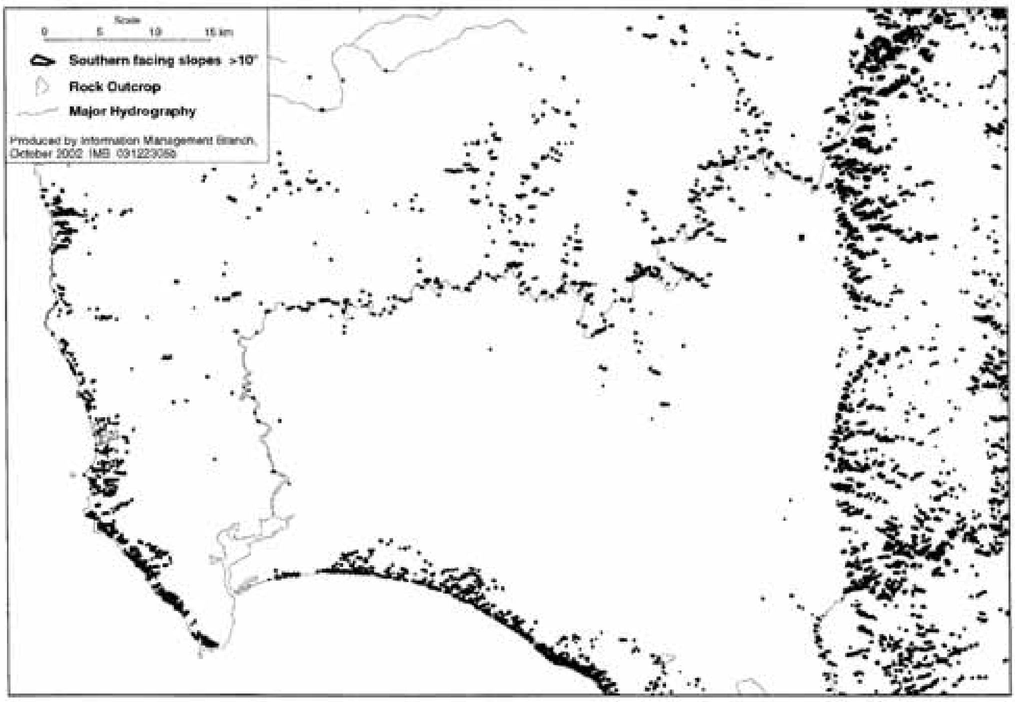



**Fig. 6.9b. Locations of potential plant species refugia in the Warren region of south-western Western Australia.** Source: Burrows and Abbott (2003, p. 444). Potential refugia associated with south-facing slopes are concentrated along the west and south coasts, and the edge of the Darling Scarp, and riparian zones along the Frankland River, while granite outcrop refugia are less frequent in this region.

## 6.4 Case study 3: Tropical savanna vegetation of northern Australia

*Summary*

The vast, sparsely populated landscapes of northern Australia are dominated by savannas – a more or less continuous cover of grasses and a variable cover of trees. The fire regimes in the savannas are driven primarily by the monsoonal, wet–dry tropical climate, which produces fine fuels annually during the wet season that then dry off each dry season. Fire is very extensive – over 300,000 km² are burnt on average each year across the savannas. The mesic savannas of the Kimberley, the Top End of the Northern Territory and Cape York Peninsula have an annual abundance of tall tropical grasses, where neither fuel nor fire-weather is limiting, and average fire frequency is about one year in two. In the semi-arid savannas, fire frequency is about one year in four in Western Australia and the Northern Territory; whereas in semi-arid Queensland, where properties are smaller and land use is intensive, fire frequency is less than one year in ten. In arid savannas, dominated by hummock and spinifex grasses, fires are infrequent and occur primarily in or following years of above-average rainfall.

Changes in mean annual temperature will lead to an increased incidence of days over 35°C, which will affect FFDI and increase the number of fire ban days. Changes in rainfall regime, and in particular the variability of rainfall, could alter fire regimes – particularly in the and arid semi-arid savannas – where fire follows years of above-average rainfall. The fundamentals of fire management in the savannas – the use of prescribed fire early in the dry season to mitigate the effects of more intense late dry season unplanned fires – is unlikely to change under climate change scenarios. Fire management in the savannas will continue to be geared towards reducing the incidence of fire, whatever the impact of climate change on fire regimes. Long-term experiments and monitoring suggest that a major biodiversity conservation objective for the savannas is to increase the amount of the savanna landscape that is unburnt for periods of five years or more. Prescribed fire will continue to be an important tool to achieve such an objective. Emerging opportunities presented by carbon markets may provide additional incentives and resources for such management, particularly on Indigenous lands. Simulation modelling indicates that, across the savannas, land use change (intensification of grazing, spread of exotic grasses) may exert a stronger influence on future fire regimes than climate change.

### 6.4.1 The tropical savannas

The tropical savannas of Australia occupy approximately 1.9 million km² or about 25% of the Australian continent (http://www.environorth.org.au), and represent over twice the area of New South Wales. This region includes the entire northern coastline and rangelands from northern Queensland to Broome. The remote nature of this area and low population density has resulted in a largely intact and continuous landscape containing a number of Australia's biodiversity hotspots, (Woinarsky *et al* 2007).

Land use varies across the savannas, with large areas managed by traditional owners and pastoralists. Other land uses include mining leases, military areas, horticulture, agriculture and protected areas such as Kakadu National Park. The savannas are governed by two states and one territory (Western Australia, Queensland and the Northern Territory).

There are two fundamental, overriding landscape features of this vast region. The first is the vegetation structure – sparse to moderate tree cover over a ground stratum of C4 grasses. The second is the wet–dry monsoonal climate – the annual occurrence of the wet season followed by a long dry season, driven by the waxing and waning, between hemispheres, of the monsoon. Both the biodiversity and fire regimes of the region are inextricably linked to these fundamental landscape features, and have been for at least 5–6 million years, following the worldwide appearance of C4 gasses during the Miocene (Osborne 2008).

Tropical savanna woodland in the Kimberley region of Western Australia
Source: Colin Totterdell (CSIRO)

Like tropical savannas in monsoonal climates elsewhere, fires are a pervasive feature of northern Australia. Fires are frequent and extensive especially in the north of the region; individual fires can cover an area of over a million hectares (Luke and McArthur 1978, p. 260) and the region has been characterised as 'perhaps the most extensive and flammable ecosystem in the world' (Liedloff and Cook 2007). Fires affect all landscapes and land tenures – pastoral properties, conservation lands, Indigenous freehold land, mining leases and, increasingly, the urban–savanna interface (Dyer *et al.* 2001; Williams *et al.* 2002). Potential fire intensity increases with the progression of the dry season (Gill *et al.* 1996, 2000; Williams *et al.* 1998; 2002; 2003b). Prescribed burning is widely used to mitigate the effects of late dry-season fires, as the control of fires – both prescribed and unplanned – becomes increasingly problematic as the dry season advances. Fire frequency declines with decreasing average annual rainfall and increasing pastoral intensiveness (Williams *et al.* 2002), although large fires can affect both mesic and semi-arid parts of the savannas in any given year (Dyer *et al.* 2001; Yates *et al.* 2008).

The biodiversity of this region has been shaped by the monsoonal climate, low fertility soils and a regime of relatively frequent fire. The tropical savannas are a matrix within which are found a number of other important landscape types, including wetlands, monsoonal vine forests, rainforests and sandstone escarpments. These have high conservation values. For example, the sandstone escarpment landscapes of the Arnhem Land Plateau have the highest richness of both plant and terrestrial vertebrates in the Northern Territory (Woinarski *et al.* 2006, 2007).

There are clear indications that there have been, and continue to be, substantial changes in both the plant and animal diversity of northern Australia since European settlement of the area about 150 years ago. Most alarming are population declines and elevated threats in groups such as granivorous (seed-eating) birds, the assemblage of some small native terrestrial mammals, and some obligate seeder woody plants. Both fire and the grazing of domestic livestock have been implicated in such declines (Bowman and Panton 1993; Franklin 1999; Woinarski and Ash 2002; Russell-Smith *et al.* 2003a; Woinarski *et al.* 2007). Cane toads have also led to localised declines in some vertebrates (Woinarski *et al.* 2007).

The effect of climate change on the savannas may appear relatively small compared with other parts of Australia where altitudinal gradients are strong, topographic complexity is high and the dispersal capacity of the biota is limited. Climate-driven change in the savannas may impact upon both the savannas and the embedded ecosystems, which may contain fire-sensitive and isolated taxa with limited dispersal potential. The significant impact on the biodiversity of the savanna biome as a whole may result from climate change and changed fire regimes acting in tandem with other drivers of change – such as land use intensification, changes in water use, and the spread of exotic plants and feral animals. Because water is in fact a limiting resource for a considerable part of the year, any climate change-driven variation in the amount and seasonal distribution of rainfall in this region has the potential to significantly affect native flora and fauna.

### 6.4.2 Fire regimes and biodiversity in the tropical savannas

Large areas of the tropical savannas experience fire with high frequency and low intensity. High temperatures and ample rainfall during the wet season is conducive to plant growth primarily in the herbaceous layer (biomass of fuel, switch B section 5.3.1). Fuel cures during the subsequent annual dry season as is available to burn (availability of fuel, switch A section 5.3.1). This annual cycle of abundant growth and curing – in combination with suitable weather conditions each year in the dry season for fire spread (temperature, humidity and wind, switch S), and ignition from lightning and anthropogenic sources (switch I) – means fire frequency is high (potentially annual) in the tropical savannas. Importantly, the model predicts that the limiting switch for savanna fire regimes is ignition, not fuel or weather. The open nature of the tropical savannas means that most fires are grass fires and the area is not capable of supporting canopy fires (Williams *et al.* 1998). Like other savannas worldwide, fire has been used as a management tool by Indigenous people for thousands of years, resulting in a largely fire-adapted landscape containing pockets of fire-sensitive habitats in fire refuges.

Fire has an important role in structuring plant and animal communities of the tropical savannas of northern Australia (Andersen *et al.* 2003; Russell-Smith *et al.* 2003a,b,c; Williams *et al.* 2003a,b). Frequent, late dry-season fires are also detrimental to some elements of the flora, particularly obligate seeder shrubs. The pan-tropical, deciduous tree and shrub species appear to be more fire-sensitive than are the dominant eucalypts (Williams *et al.* 2003a,b). Nevertheless, the lowland savanna flora is relatively resilient to variation in fire regime (Andersen *et al.* 2003; Williams *et al.* 2003a,b).

With respect to fauna, different fire regimes produce substantial divergence in fauna (Woinarski 1990; Woinarski *et al.* 1999, 2004; Corbett *et al.* 2003; Pardon *et al.* 2003; Andersen *et al.* 2006), typically with the most notable polarisation in faunal communities being between long-unburnt treatments and those burnt frequently by intense, late dry-season fires. In the Kapalga fire experiment, small mammals in general increased in the unburnt treatment compared with both annual early dry-season fire treatment and late dry-season fire treatments, while relatively few species of frogs (1 of 11), lizards (5 of 16) or birds (5 of 25) were affected by experimental treatment (Corbett *et al.* 2003). Some species, such as the northern brown bandicoot (*Isoodon macrourus*), a small fossorial marsupial, decreased under all fire treatments, possibly because it is reliant on a mosaic of burnt and unburnt patches that is finer-scaled than that produced in the 15–20 km$^2$ landscapes of the fire treatments (Pardon *et al.* 2003).

Studies of the long-term (decadal) absence of fire are rare, as such patches are very uncommon in the savanna landscape. In mesic savannas near Darwin, Fensham (1990) documented declines in herbaceous species diversity; grass cover may also decline in such systems in the decadal absence of fire (Russell-Smith *et al.* 2003b; Dr Paul Williams, Queensland Department of Environment, pers. comm.). There may also be substantial increases in faunal diversity in long-unburnt sites compared with frequently burnt sites

(Woinarski *et al.* 2004). Indeed, the most notable feature of faunal response to variation in fire regime in the savannas has been the strong difference between being between long-unburnt treatments and those burnt frequently by intense, late dry-season fires (Gill *et al.* 2009).

Given these results from long-term experiments and monitoring, a major biodiversity conservation objective for the savannas is to increase the amount of the savanna landscape that is unburnt for periods of five years or more; a target of 5% of the landscape was suggested by Andersen *et al.* (2003).

There are also, not surprisingly, complex interactions between fire regimes and other environmental factors, particularly occasional drought (Fensham *et al.* 2003). Such interactions may place stress on populations of rare or threatened species. For example, Williams *et al.* (2004) reported that *Acacia ramiflora*, a rare shrub in north Queensland that normally resprouts and germinates seedlings after fires, was strongly affected drought following fire in 2001, causing post-fire sucker shoots to die, with no post-fire seedling recruitment.

### 6.4.3 Simulating the savanna environment using FLAMES

The computer simulation model, FLAMES, has been extensively tested in order to assess its capacity to capture and reproduce, via simulation, key ecological features of the savanna that vary as a consequence of variation in moisture regime.

The FLAMES model tracks the fate of individual trees in a stand over decades to centuries (Liedloff and Cook 2007). The model includes components that allow for competition for water between trees and grasses – as well as the tracking of fuel loads, tree dynamics (mortality, recruitment) and stand basal area – in relation to drought and fire. It can also be used to explore the economics of management options, and track above-ground carbon stocks and flows. It simulates a single hectare stand of trees upon which we can impose a range of different management and climate change scenarios. However, it is not spatially explicit and is not linked to biodiversity models. Nevertheless it can be used to evaluate the impacts of fire regime change on tree basal area and habitat structure, and hence be used to make qualitative assessments of fire regime change on biodiversity.

There are strong rainfall gradients in northern Australia, from coastal to inland regions. For example, in the Northern Territory, rainfall decreases from about 1987 mm on the Tiwi Islands around Darwin, to about 363 mm at Tennant Creek, 1000 km inland (Williams *et al.* 1996; Cook *et al.* 2002). There is a concomitant decrease in tree basal area from about 15 $m^2$/ha to <2 $m^2$/ha along the gradient. Simulations using the FLAMES model for sites along this rainfall gradient, using historic rainfall records, predicted tree basal area along the gradient very well, with good agreement between model output and field measurements (Liedloff and Cook 2007; Fig. 6.10).

Other areas of the tropical savannas, such as north Queensland, do not experience the regular monsoon-based rainfall of the northern coast. Annual rainfall in these regions is influenced more by occurrence of low-pressure depression rainfall, which is less reliable, resulting in greater inter-annual variability in rainfall, a higher proportion of winter rainfall, and the occurrence of periodic droughts that are intense enough to kill trees (Fensham and Holman 1999; Fensham *et al.* 2003).

The FLAMES model was used to simulate a stand of ironbark (*Eucalyptus crebra*) located around the Charters Towers region, Queensland, based on local soil properties and daily rainfall records. The simulation results show that the FLAMES model predicts the historical drought patterns reported by Fensham and Holman (1999) that were based on the drought index, historical reported records and field studies (Fig. 6.11).

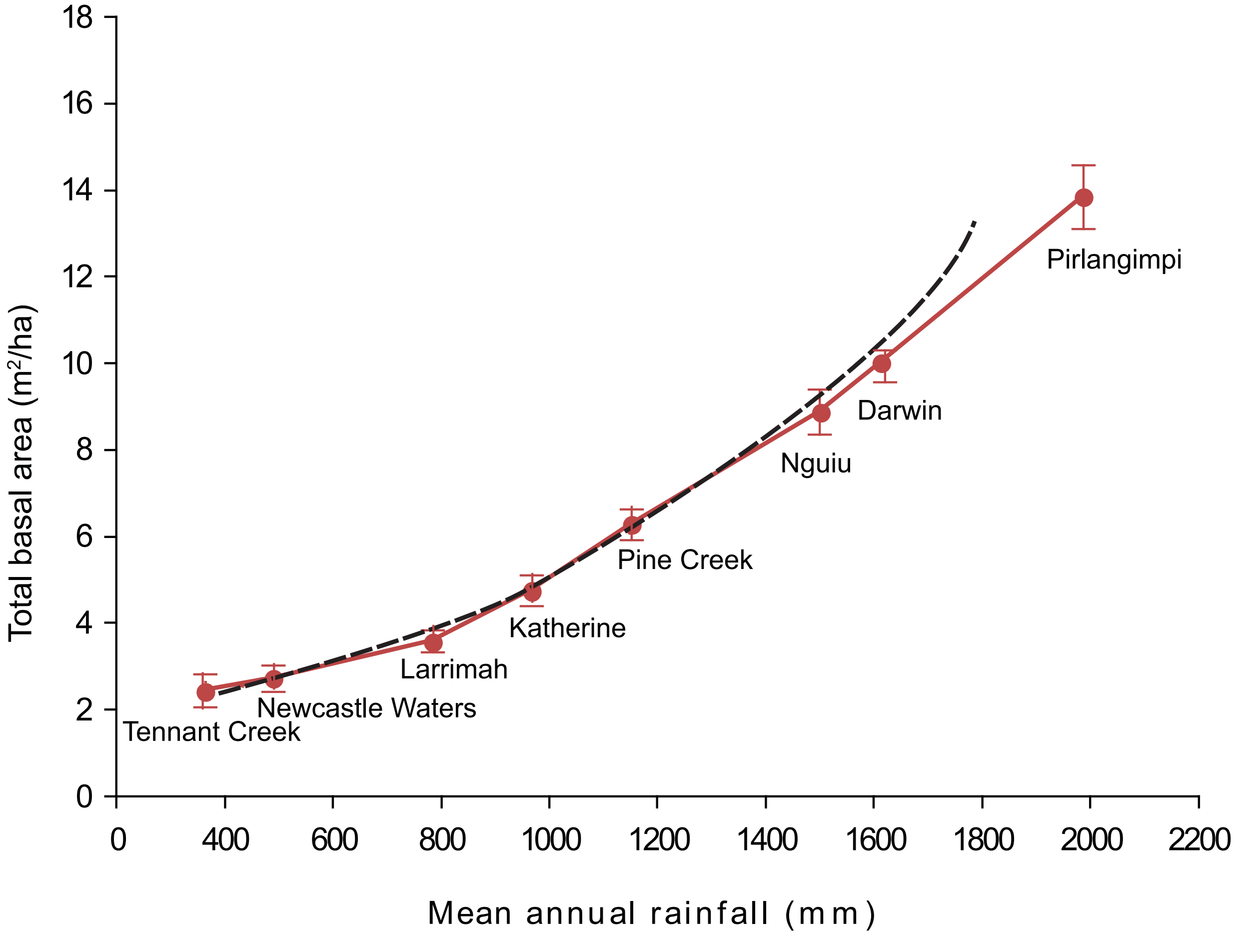



**Fig. 6.10. Comparison of FLAMES model outputs with empirical data.** Shown are tree stands predicted by the FLAMES model (solid red line) and the actual populations measured by Williams *et al*. (1996) and Cook *et al*. (2002) (dashed black line). Source: Redrawn after Liedloff and Cook (2007).

These results show that FLAMES can simulate the behaviour of tree populations in response to long-term variation in annual rainfall, and suggest that any increase in drought occurrence can significantly affect tree populations, and potentially the associated flora and fauna. We can also infer that if climate change results in the occurrence of droughts to areas that do not currently experience this variability (e.g. north western Australia), this may result in changes to tree populations, with more periods of dieback and population recovery. This influences the amount of biomass in coarse woody litter, tree biomass and grass biomass (switch B) for fires.

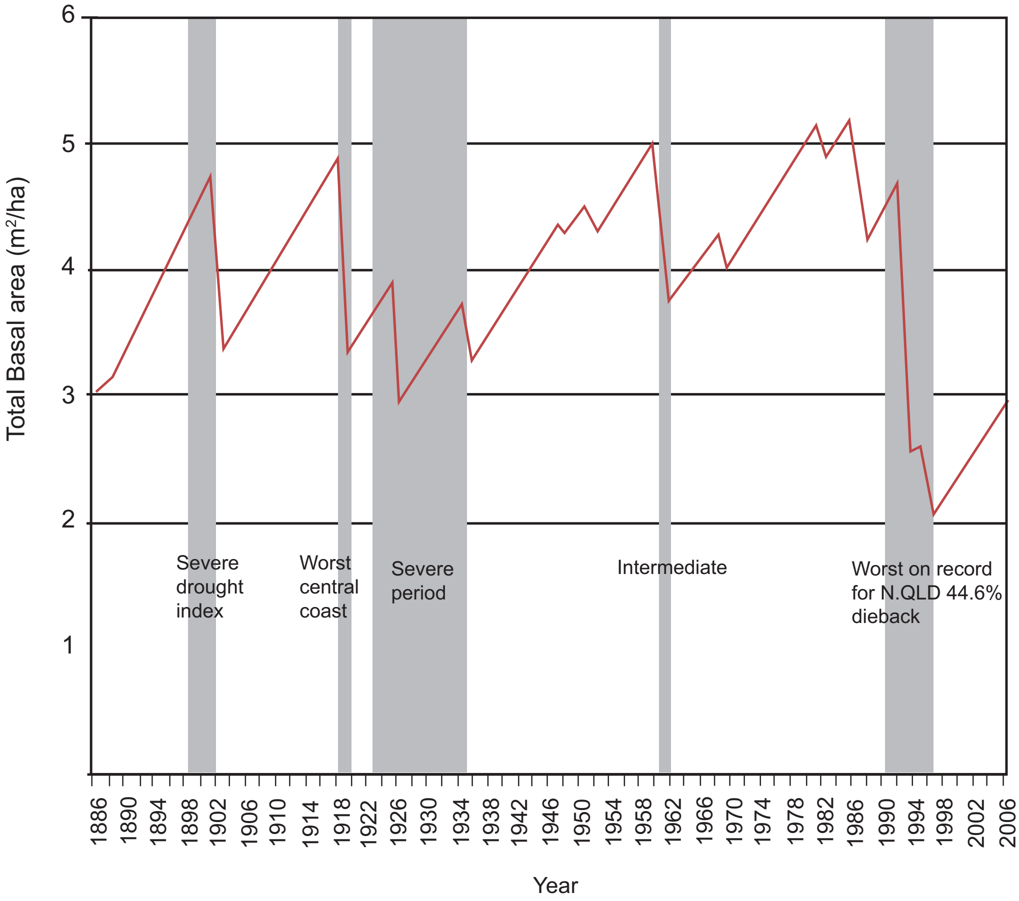



**Fig. 6.11. Flames simulation of an ironbark stand for Charters Towers,** Queensland. Model runs use historic rainfall records and the occurrence of reported droughts (grey shading) from Fensham and Holman (1999).

FLAMES can also be used to simulate the influence of variation in rainfall seasonality as well as amount. Changes in rainfall seasonality may lead to changes in tree population structure and fire seasonality, which can impact on trees by burning them during more fire-sensitive periods of growth and reproduction (Williams *et al.* 2003a) and have detrimental flow-on impacts on fauna.

While northern Australia has long been considered extremely fire-tolerant, current research into plant and animal populations suggests the system can be influenced greatly by different fire regimes, and associated fire seasonality and intensity. Fire is known to kill trees as a function of fire recurrence, fire intensity, and the size and age of the tree. Relationships between fire intensity and seasonality, type of ignition (line or point), and tree mortality and fire intensity have been built into the FLAMES model and the long-term consequences of variation in fire regimes on savanna tree populations can be tested. Fig. 6.12 shows a 250-year simulation of savanna tree stands around Darwin. In the absence of fire, the tree populations are relatively stable and vary relatively little with variation in annual rainfall. As fire frequency increases, the tree populations decline. The simulation with the highest fire frequency represents the most extreme fire conditions late in the dry season, with a front-style management fire on average every three years. It must be recognised that it is unlikely that this regime could be maintained for a century under current conditions, but these simulations highlight the ability of fire management to drastically alter tree stand composition and structure through adult tree mortality and suppression of recruitment. This impact of fire on the tree and shrub component of the savanna has impact on other flora and fauna with structural cover and habitat requirements.

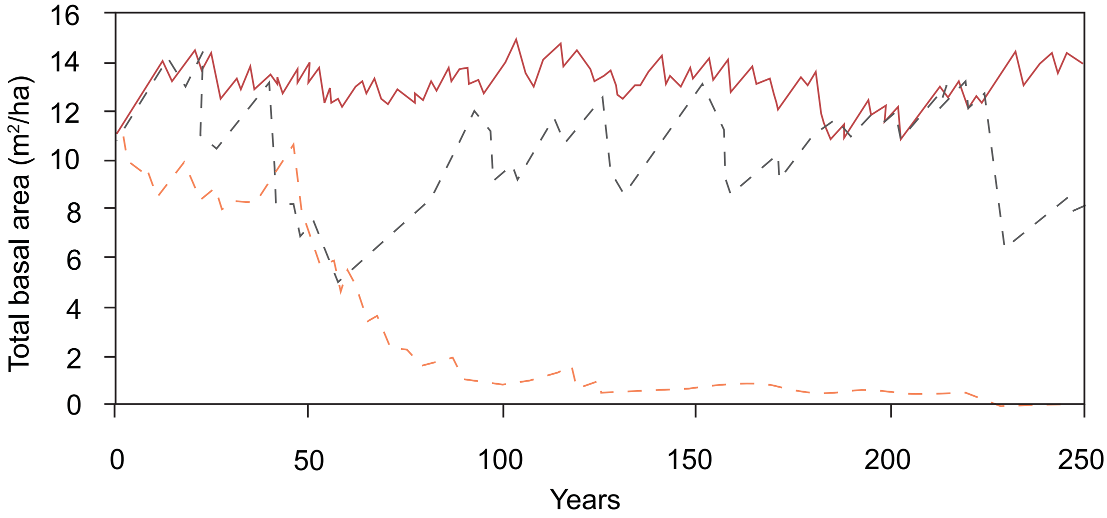



**Fig. 6.12. Total basal area (m²/ha) of a stand of savanna trees in relation to variation in fire regime**. Simulations for Darwin region, Northern Territory, exposed to no fire (red line), a low intensity fire on average every five years (dashed grey line) and a high intensity late dry season fire every three years (dashed orange line).

Highly seasonal rainfall can also result in periods of 'drought' within the year. Alternatively, not all the rain that falls is necessarily available to the tree — once the soil is fully saturated, any additional rain will move into deep drainage, or flow to creeks and rivers.

Fig. 6.13 shows the results of a simulation, where each year a theoretical eucalypt species receives 1000 mm of rainfall but the distribution within the year varies from extremely seasonal (e.g. Katherine, Northern Territory) to more uniform rainfall throughout the year (e.g. Sydney, New South Wales). These results show a rapid decline in the tree basal area of trees as rainfall becomes more seasonal. The preliminary analyses suggest that in southern temperate forests – where, as a consequence of climate change, the proportion of summer rainfall is projected to increase – the introduction of as little as one seasonal year in five can halve the resulting tree basal area of areas currently supporting open forest. This could potentially result in the introduction of grasses, and alter fire regimes. Such changes could be very important to fauna relying on trees and associated habitat, and fire regimes, given additional within-dry season variation in fine fuel moisture.

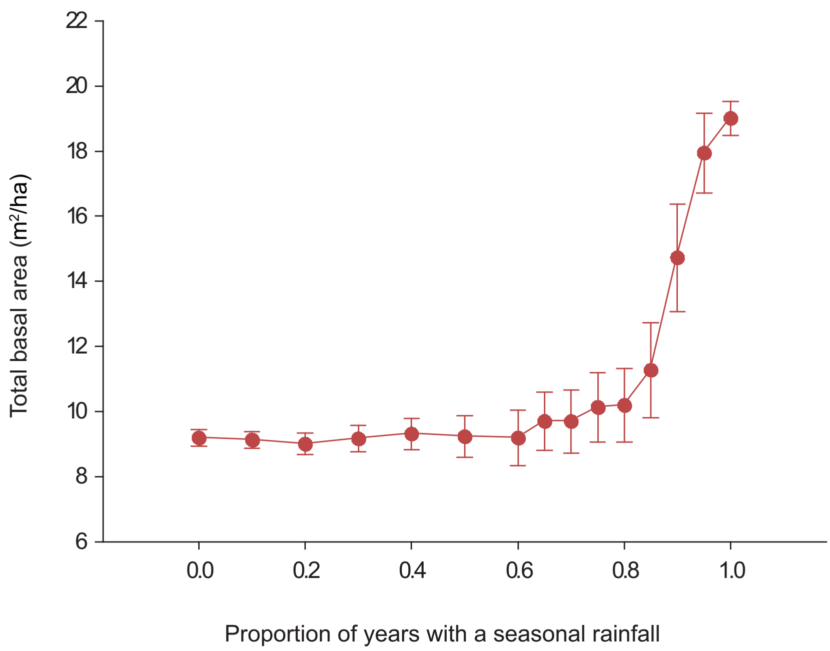



**Fig. 6.13. The effect of seasonality of rainfall on a simulated tree population in an area with 1000 mm annual rainfall.** Source: Redrawn from Liedloff and Cook (2007).

### 6.4.4 Implications of climate change for fire regimes in the tropical savannas

Current climate change scenarios for the tropical savannas and northern Australia include an increase in temperature, increases in total rainfall, changes in rainfall seasonality and the frequency of severe events such as cyclones, and potential sea level rises (CSIRO and Bureau of Meteorology 2007).

Given that fire frequency is effectively running at or near its maximum in the tropical savannas (section 5.3.1), and that the limiting switch is ignition rather than fuel or fire-weather, then the impact of climate change on fire regimes of the savannas may be relatively muted. However, the impact of climate change itself on savannas will not necessarily be muted. There is evidence for the role of $CO_2$ fertilisation in the expansion of the rainforest boundaries into savannas (Banfai and Bowman 2005, 2006), and the increase in tree density within the savannas over the past half century. Such 'bush thickening', as a consequence of $CO_2$ fertilisation, has also been postulated in the savannas of southern Africa (WJ Bond, pers. comm.). Moreover, rising temperatures may place physiological stress on dominant trees. For example, the temperature optimum for *Eucalyptus miniata* is 32°C, and photosynthetic capacity declines rapidly at temperatures above 35°C (L Prior *et al.*, unpublished data). Hence, an increase in the number of days above 35°C (see further on) may constrain tree productivity in the savannas.

In this case study we used FLAMES to explore some of the potential consequences of climate change on components of the fire regimes and biodiversity in northern Australia. We focused on the mesic savannas of the Darwin 'Top End' region of the Northern Territory, and the savannas of the Townsville–Charters Towers region of north-eastern Queensland. We investigated the likely impacts of climate change and other land use change (e.g. the impact of exotic pasture grasses) on fire regimes in northern Australia. Modelling of potential impacts on biodiversity is not possible at present and requires further model development.

#### *Temperature*

Climate change projections for northern Australia show an increase in temperature of between 1.0 and 1.2°C by the year 2030 (CSIRO and Bureau of Meteorology 2007). Small increases in daily temperatures can result in increases in the frequency of extreme events, or the number of very hot days per year. The Darwin region currently experiences 11 very hot days per annum (greater than 35°C), which is projected to increase to 44 days per year in 2030 and up to 227 by 2070 (CSIRO and Bureau of Meteorology 2007). This increase may affect the number of total fire ban days, and therefore the length of the permitted burning season and the ability to undertake management burns. These changes could lead to more intense fires and therefore a greater area burnt. Other determinants of fire spread, such as relative humidity and soil moisture, have not been modelled. However, FLAMES is not parameterised to deal with variation in daily temperature, and its consequent effects on tree growth or grass growth.

#### *Rainfall amount*

Projected changes in annual rainfall for the mesic, near-coastal regions of northern Australia are generally relatively small (Fig. 4.10). Other projected changes include increased evapotranspiration, but relative humidity is likely to remain unchanged or decrease slightly.

With the vast majority of rainfall around Darwin occurring during the wet season (November to April), trees have ample water and the excess is directed to river flow after the soil profile is full. As the dry season commences and soil dries out, grasses cure and trees rely on stored soil water to survive the dry season. The store of soil water is therefore more important than the total rainfall amount, assuming adequate rainfall occurs to fill the soil profile (Liedloff and Cook 2007).

#### *Rainfall seasonality and variability*

The variability of annual rainfall generally increases with distance from the coast, as rainfall becomes dominated more by storms and tropical depressions than by monsoonal rains. While temperature and total rainfall projections are available, there is currently limited ability to predict what change will happen with respect to rainfall distribution and inter- and intra- annual variability from GCMs. Nevertheless, the incidence of severe cyclones is likely to increase (Walsh *et al.* 2004) and the proportion of rain that falls during severe storm events is expected to rise by 20–30% (Pittock 2003). The biggest change to ecosystem structure is likely to be seen in variation in the patterns of daily precipitation, for which there are no projections at present.

### $CO_2$ fertilisation

Elevated $CO_2$ levels and associated $CO_2$ fertilisation is predicted to increase grass production by 15% (Stokes *et al.* 2003, 2008) and tree production by 10%. This increases fuel production for fires (switch B, section 5.3.1), which may result in increased fire intensity for a given set of fire weather conditions. However, the effect of $CO_2$ fertilisation on fire frequency is unknown.

### Lightning ignitions

Globally, lightning ignition of fires may increase due to climate change (Goldammer and Price 1998). Any increases in lightning frequency (reference) will potentially provide additional ignitions. However, lighting occurs in conjunction with dry landscapes only at the end of the dry season or very early wet season, by which time much of the savanna landscape has already been burnt, generally by fires resulting from human ignition sources. Thus, the impact of changed lighting incidence on the fire regimes of the savannas is uncertain.

### Regional variation

Northern Australian fire regimes are affected by several environmental and management gradients (Williams *et al.* 2002). The first is the north–south gradient in rainfall amount; the second is the east–west gradient in seasonality, with the east having a greater likelihood of dry season rainfall; and the third is the cadastral gradient from Queensland to the Northern Territory and Western Australia.

The Kimberley, Northern Territory Top End and Cape York Peninsula are relatively lightly stocked with cattle, have relatively large property sizes and an annual abundance of tall tropical grasses. These areas have a fire frequency of about 0.25 (i.e. on average one fire every four years). The fire regime is unlimited by time and fire-weather. Changes in windiness may have a greater impact than changes in temperature or humidity. However, climate change scenarios for this region do not specify windiness in their data. Any increase in rare events such as dry-season rainfall, or in the temporal distribution of rainfall events at the start and end of the wet season, may have important fire x phenology interactions. It may also change the viability of annual sorghum, a major driver of fire regimes where it occurs. An increase in dry-season rainfall could eliminate sorghum if the probability of fires occurring after germination but before seed set increased. Conversely, decreases in dry-season rainfall in the eastern Top End and Queensland could see an expansion of annual sorghum.

Low intensity fire in annual *Sorghum* grass fuels in a tropical savanna, Territory Wildlife Park, Darwin Region, NT
Source: Dick Williams

In the semi arid savannas, fire frequency is about 0.25 in Western Australia and the Northern Territory. In Queensland, where property sizes are small and management intensity high, fire frequency is very low (about 0.1). The future may bring intensification of land use and management in the Northern territory and Western Australia, with a consequent decrease in fire frequency. It may bring consolidation of properties in Queensland and perhaps an increase in fire frequency. The trade-off of forage for fuel will remain strong for this region. The spread of buffel grass (*Cenchrus ciliaris)* in this region could alter fire regimes. In this region, management has the overriding influence on fire regimes.

In arid savannas, extensive landscapes dominated by spinifex burn at a low frequency in response to fuel availability (determined by preceding years of above-average rainfall) and lightning strikes. Changes in rainfall regime, in particular the variability of rainfall, could alter fire regimes here. The area is likely to remain very sparsely settled and human intervention in fire regimes will probably remain small and spatially restricted. The spread of buffel grass in this region could also alter fire regimes, bringing high-intensity fires to areas where shorter grasses had dominated. It also has the potential to outcompete spinifex in many areas, with consequent changes to fuel dynamics.

### 6.4.5 Other drivers of fire regimes

*Impact of invasive grasses on fire regimes*

One of the major landscape changes that has occurred in northern Australia has been the introduction and spread of exotic pasture grasses, which generally have a high potential for invading savanna ecosystems (Rossiter *et al*. 2003). A number of such grasses are currently spreading around the Top End of Australia. Gamba grass (*Andropogon gayanus*) is one such species, with a potential range covering much of the tropical savannas (Setterfield *et al*. 2005). Gamba grass produces up to 20 tonnes per hectare of biomass (Douglas and Setterfield 2005) influencing the fuel mass (switch B; section 5.3.1). Gamba grass also cures later in the dry season than native grasses, resulting in changes to fire season (switches A and S). These changes to fuels result in fire intensities not experienced under current native grass fires in northern Australia. Such intense fires increase tree mortality compared with savanna fires burning in native grass fuel loads. A further pressure is that gamba grass also competes with trees for water and nutrients. Areas infested with gamba grass are likely to require additional management to protect life and property.

A 50-year simulation of the FLAMES model (Fig. 6.14) with gamba grass shows how repeated, intense fires can reduce the tree component of a stand. This will have consequences for habitat structure and complexity, and the management of biodiversity of the savannas if the range of gamba grass continues to expand.

It was not possible to produce a similar result using the FLAMES model based on only estimates of temperature change from climate change scenarios. The FLAMES model predicts that regimes with the highest fire frequency in Fig. 6.12 (and other simulations with annual management fires) take between 100 and 150 years to reduce the tree populations, to levels reached after 50 years of fires in gamba grass fuels. This suggests that the current threats to biodiversity posed by introduced grasses in the savannas (Douglas and Setterfield 2005) may actually be greater than those posed by future climate change. The current threats posed by invasive grasses are unlikely to abate and such changes are likely to add to any threats posed to biodiversity by climate change.

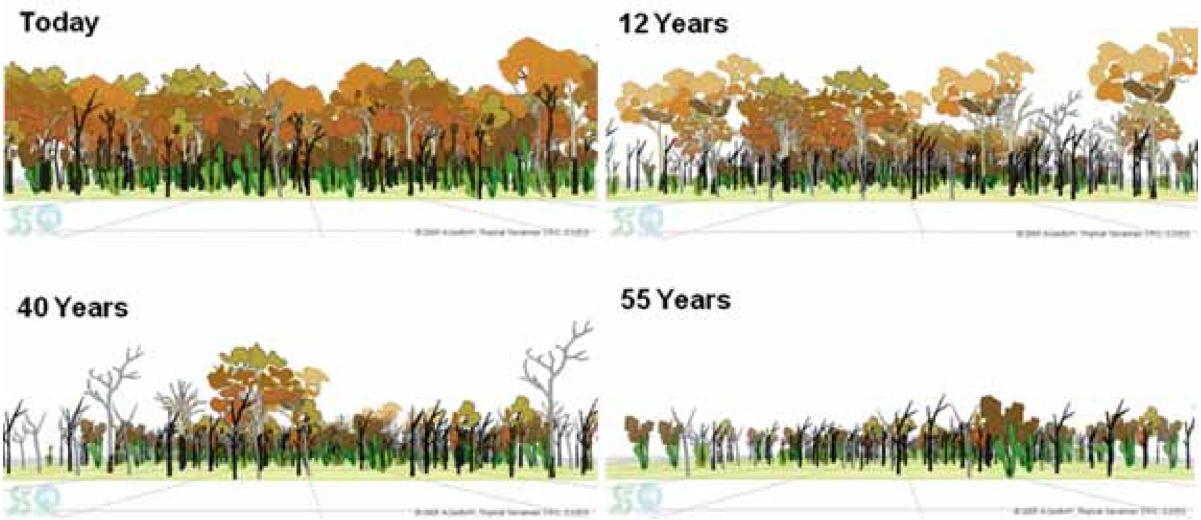



**Fig. 6.14. The simulated impact of annual fire in gamba grass (*Andropogon gayanus*) fuels on a stand of trees in the Top End, Northern Territory.**

High intensity fire in gamba grass fuels, Darwin district, NT. Gamba grass is an exotic species that increases fuel loads, and hence fire intensity, in savanna fires

Source: Samantha Setterfield, Charles Darwin University

### *The carbon economy*

The expanding carbon economy, with prospects for landscape-based carbon offset products, may also affect the tropical savannas. Greenhouse gas mitigation and offsetting carbon emissions has already led to investment in Indigenous fire management to reduce large-scale, late dry-season fires in Arnhem Land (Russell-Smith *et al.* 2009a,b; Williams *et al.* 2009), with potential biodiversity benefits. The same carbon market may also be driving changes in land use, such as the increasing interest in silviculture in northern Australia.

Such economic drivers of a carbon economy could also influence fire regimes, as fire is used as a management tool to offset emissions or sequester carbon. It could also be that the carbon released by management fires will have to be accounted for, thus placing economic constraints on fire management and altering anthropogenic means of fire ignition.

### 6.4.6 Implications of climate change and changed fire regimes for biodiversity management

Fire management in the savannas of northern Australia is predicated on two fundamental factors: the potential for large landscape fires as a consequence of the annual occurrence of continuous fuels and high fire danger; and a long history of Indigenous fire management (Bowman 1998; Russell-Smith *et al.* 2009b). This co-evolution of the savannas in the face of a monsoonal climate, frequent natural fires and active use of fire by Indigenous people for tens of thousands of years has resulted in a landscape consisting of a resilient biota, but containing islands of fire-sensitive habitats and species.

The modelled outputs from FLAMES of future fire regimes based on climate change scenarios are limited at present basically to inferring biodiversity impacts from changes in the primary outputs from FLAMES – changes to tree basal area as a function of variation in fire regime. However, variation in tree basal area at a site over time is an important component of landscape heterogeneity – declines in tree basal area over time are likely to be associated with declines in large tree abundance and vertical structure, both of which are important habitat variables. Further research is needed into the development of biodiversity models that can be coupled with FLAMES.

Lessons from experimental fire–biodiversity studies, such as at Kapalga, is that there is a degree of resilience in the savanna biota to variation in fire regimes (Andersen *et al.* 2003, 2005). With respect to plants, there was little detectable variation in floristic composition between annual early dry-season regimes, annual late dry-season regimes and unburnt regimes (Williams *et al.* 2003a). With respect to fauna, different species were affected by fire treatment in a variety of ways, within and between treatments. In general, the greatest difference was between burnt and unburnt treatments. There was also variation in species responses within treatments – for example in the severe, annual, late dry-season treatment, populations of some species declined, whereas populations of other species increased (Corbett *et al.* 2003). Such variation indicates that no single fire regime serves all conservation objectives, and highlights the importance of: (i) conserving the species rather than maximising populations; and (ii) varying fire regimes within limits appropriate to management aims, a general aim for fire management in areas managed for biodiversity conservation (Andersen *et al.* 2003, 2005; Gill 2008).

Biotic resilience notwithstanding, results from fire and biodiversity experiments have indicated the need to reduce incidence of fire (Andersen *et al.* 2003, 2005; Williams *et al.* 2003b), with the concomitant effects of increasing the intervals between fires. Current fire management aims to achieve this by reducing the incidence of late dry-season fires, by using prescribed burning during the early dry season. Research on fire and biodiversity has also indicated the need to have an explicit management goal of increasing the amount of country that is unburnt for 4–5 years. Achieving these goals is likely to be more difficult under a climate change scenario of elevated temperatures and an increased incidence of days >35°C. These fundamentals of fire management in the savannas are unlikely to change in the near future. Fire management in the savannas will continue to be geared towards reducing the incidence of fire, whatever the impact of climate change on fire regimes.

The threats posed by introduced grasses are likely to continue to have major impacts on fire regimes and biodiversity. The threats to biodiversity from these grasses (via their direct effects on habitat structure, and via their effects on fire regimes) are likely to be much greater than the potential threats to biodiversity brought about by climate change. This is similar to the conclusions of Clarke *et al.* (2005a) for the arid zone – exotic grasses exerted a greater influence on ecosystem function than rainfall variability. Any increase in the risk of elevated fire danger will be compounded by the spread of these grasses.

Climate change and the need to reduce greenhouse gases does, however, present opportunities to improve fire management in the savannas. Nationally, savanna burning contributes about 3% of Australia's national

GHG emissions (Russell-Smith *et al.* 2009a,b). The opportunity to reduce GHG emissions by fire mitigation brings biodiversity as well as greenhouse gas benefits. Moreover, fire mitigation using prescribed burning has stimulated landscape research into the overall effectiveness of prescribed burning in delivering fire mitigation and biodiversity benefits (Price *et al.* 2005, 2007). This research is supported by significant institutional support, from both private and public sector (>$1m per annum; Russell-Smith *et al.* 2009b), and this model will have application in other savanna areas of the world and may have application in temperate fire-prone ecosystems.

## 6.5 Case study 4: Sclerophyllous vegetation of the Sydney Basin region

*Summary*

The Sydney Basin has a high diversity of vegetation types and species, and includes high numbers of obligate-seeding plants. The distribution of plant functional types is partly affected by gradients of temperature and moisture. Increasing temperatures (the most certain outcome of climate change) could disadvantage that component of the vegetation (resprouters with a persistent seedbank; shrubs in particular) that is most resilient to variation in fire interval. Higher temperatures may favour obligate seeders with persistent seedbanks – the functional type most sensitive to change in fire frequency (length of inter-fire interval). Thus, climate change has the potential to shift vegetation composition toward functional types that are more sensitive to any shift in length of between fire interval.

Simulation modelling indicates that increases in FFDI could result in increased area burned. The consequences of such changes in fire regimes for biodiversity are, however, mixed. Shifts in inter-fire interval (IFI) were predicted to be insufficient to significantly change landscape-scale extinction probability of IFI-sensitive plant species, assuming prescribed burning is held at current levels. In contrast, the probability of crown fires may increase by up to 20% under the high emissions scenario. Elevated risks to people and property,to  intensity-sensitive populations of some taxa (e.g. arboreal mammals), and to soil stability may result.

The modelling suggests that, under the broad climate change scenario of warming and drying, the effects of warming on fire-weather – and consequently fire intensity and area burned – may outweigh any diminution of surface fuel loads caused by declining moisture.

Prescribed burning is one tool used to manage the conservation estate in the Sydney Basin. Prescribed burning can have major effects (positive and negative) on biodiversity, catchments and human protection. Moderate increases over current levels of prescribed burning (e.g. two- to threefold increase) are unlikely to increase risk to the integrity of plant diversity in the widespread, species-rich dry sclerophyll vegetation. Commensurate risk reduction to urban and other ecological values sensitive to fire intensity is, however, likely to be small. Larger increases in prescribed burning (i.e. > fivefold) would be needed to counteract effects of the high climate change scenario. Such an increase may not be feasible on the basis of cost and resources. Benefit-cost analysis of various prescribed burning options (treatment strategy, area burnt) as a function of risk to various landscape values (life, property, biodiversity, water) is an urgent research need.

### 6.5.1 Introduction and background

The Sydney Basin bioregion 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 Dividing Range in the west, to the Clyde and Shoalhaven river catchments in the south (Fig. 6.15). The terrain is diverse and rugged across the bulk of this region, reflecting the predominant sedimentary geology and elevation >1000 m on the highest parts of the Great Dividing Range. The region contains one of the largest tracts of relatively intact native vegetation in eastern New South Wales. It also contains the largest city in Australia (Sydney) along with satellite developments on the Central Coast, the Illawarra and the Blue Mountains.

Hanging Rock overlooking the Grose Valley, Blue Mountains National Park, NSW
Source: Mark Langford/AUSCAPE

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 of the region. Embedded within this sclerophyll matrix are pockets of warm temperate and sub-tropical rainforest, often associated with gullies (topographic fire refugia) and/or volcanic intrusions. The rare gymnosperm, *Wollemia nobilis*, lies in such rainforest pockets. A significant tract of grassy woodlands is situated on the shale-based soils of the Cumberland Plain in western Sydney – now heavily fragmented by agricultural and urban development. Elements of drier plant communities (e.g. *Callitris, Brachychiton*) are found as dominants on the western margins of the regions.

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 plants of which about 100 are eucalypts. The bulk of Sydney's water supply catchments are situated within these landscapes, to the west and south of Sydney.

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. 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). The sclerophyll plant communities exhibit relatively rapid rates of fine fuel accumulation (Conroy 1996). Major episodes of fire resulting in large areas burned and loss of property in the region occur once or twice a decade (Cunningham 1984; Gill and Moore 1996).

Most natural vegetation within the region, including the GBMWHA, lies within national parks, nature reserves and conservation areas that are managed by the NSW National Parks and Wildlife Service within the NSW Department of Environment and Climate Change (Fig. 6.15). Other significant areas of fire-prone bushland are managed by the Sydney Catchment Authority (a New South Wales statutory authority) and local government. This configuration of vegetation, terrain, biodiversity and human assets provides a challenging arena for fire management.

Policy and planning for bushfire management must therefore deal with protection of lives, property, infrastructure, biodiversity conservation, air pollution and catchment stability. Currently, the basis for management of risk to these values is heuristic and speculative. The prospect of climate change adds further uncertainty and heightens the management challenge.

In this case study we examine approaches to understanding risks posed by fire to key values – particularly biodiversity and urban assets – and the way these may vary under climate change. The following questions are addressed:

- How will climate change affect the ability of species-rich vegetation to cope with fire in the future?
- How sensitive are current fire regimes to the drivers of fire?
- How will fire regimes change in the future due to climate change and other influences?
- How do differing management approaches affect risk, and how will this be altered in the future?

Examples and results from recent research (completed and in progress) utilising a variety of approaches and methods are used to address these questions.

Dry Selerophyll Open Woodland in the Blue Mountains National Park.

Source: Kate Hammill

### 6.5.2 Climate change projections for the Sydney region

Projections of climate change for the region include an increase in temperature and evaporation across all modelling scenarios for 2030 and 2070 (CSIRO and Bureau of Meteorology 2007; Table 6.2). Projected trends in rainfall, wind, relative humidity and solar radiation are more variable (Table 6.2). The median projection is for declining rainfall overall, with a shift toward summer and autumn dominance. Increases in evaporation are projected, hence available moisture is likely to decline. These influences underpin a predicted increase in daily and annual fire danger (see also Lucas *et al.* 2007 and section 4.2).

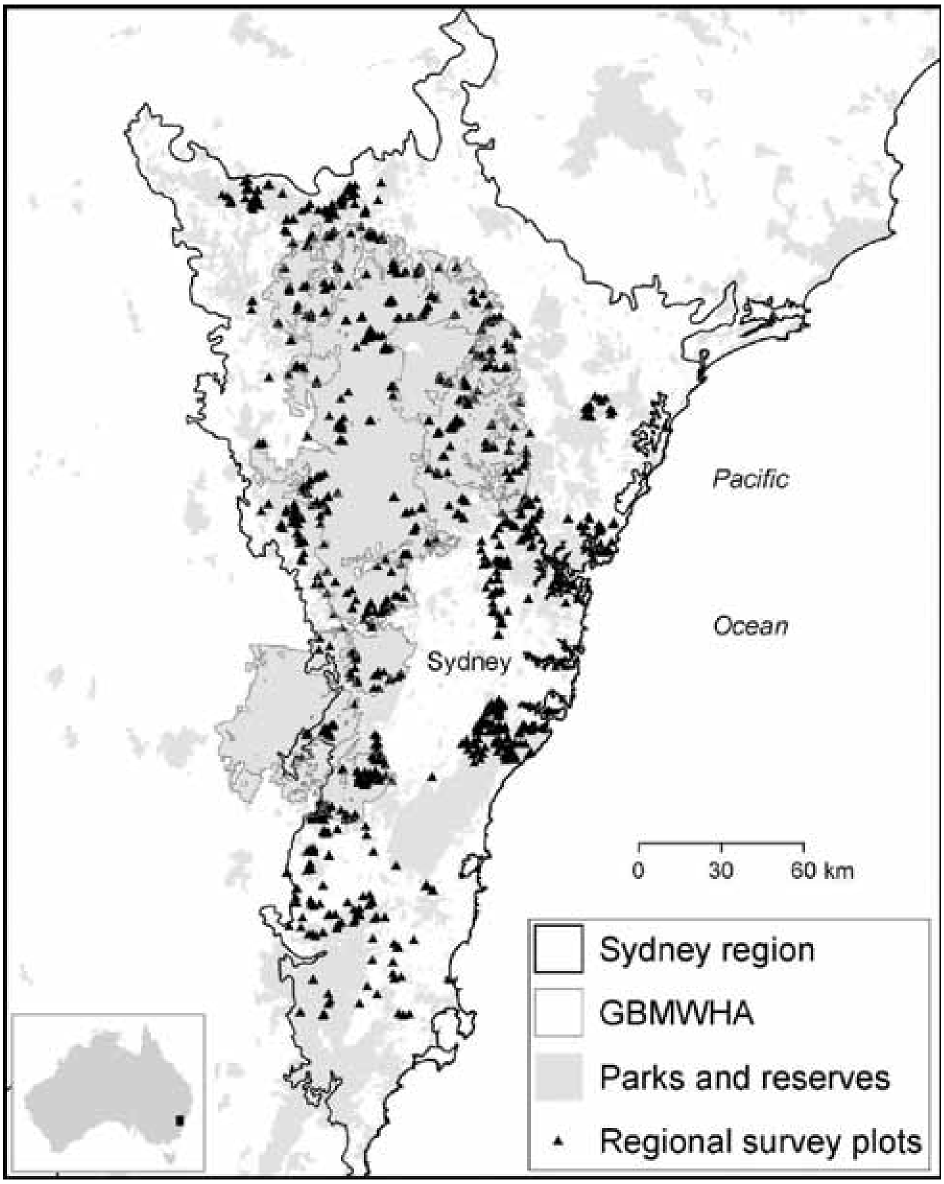



**Fig. 6.15. Map of the Sydney bioregion.** Map shows the boundaries of the Greater Blue Mountains World Heritage Area, the conservation reserve system and the location of vegetation survey plots used in analyses of plant functional types in relation to environmental gradients. Source: Bradstock and Hammill (2008).

**Table 6.2. Scenarios of climate change for Sydney, representing different 21st centry emission pathways.** Scenerios are: (A1B – mixed technology and high population growth; B1 – constrained under altered economic development and population growth; A1FI – fossil fuel intensive and high population growth), for the years 2030 and 2070, relative to 1990. The 50th percentile (50p), or median, is the 'central estimate' of the projected change. The 10th and 90th percentiles (10p and 90p, respectively) are given as a guide to the uncertainty range (CSIRO and Bureau of Meteorology 2007; Chapter 5 'Regional climate change projections'; p. 53). Source: CSIRO and Bureau of Meteorology (2007; Table B13; http://www.climatechangeinaustralia.com.au/documents/resources/TR_Web_AppendixB.pdf.

| Variable | Season | 2030 A1B 10p | 2030 A1B 50p | 2030 A1B 90p | 2070 B1 10p | 2070 B1 50p | 2070 B1 90p | 2070 A1FI 10p | 2070 A1FI 50p | 2070 A1FI 90p |
|---|---|---|---|---|---|---|---|---|---|---|
| Temperature (°C) | Annual | 0.6 | 0.9 | 1.3 | 1.1 | 1.6 | 2.2 | 2.1 | 3.0 | 4.3 |
| | Summer | 0.6 | 1.0 | 1.5 | 1.0 | 1.6 | 2.5 | 2.1 | 3.1 | 4.7 |
| | Autumn | 0.6 | 0.9 | 1.4 | 1.0 | 1.5 | 2.3 | 1.9 | 3.0 | 4.3 |
| | Winter | 0.6 | 0.8 | 1.2 | 0.9 | 1.4 | 1.9 | 1.8 | 2.6 | 3.7 |
| | Spring | 0.7 | 1.0 | 1.5 | 1.2 | 1.7 | 2.5 | 2.2 | 3.3 | 4.8 |
| No. days over 35°C (currently 3.5) | Annual | 4.1 | 4.4 | 5.1 | 4.5 | 5.3 | 6.6 | 6.0 | 8.2 | 12.0 |
| Rainfall (%) | Annual | -9 | -3 | +3 | -14 | -4 | +5 | -25 | -8 | +10 |
| | Summer | -7 | +1 | +9 | -12 | +1 | +14 | -21 | +2 | +28 |
| | Autumn | -10 | -2 | +6 | -16 | -3 | +11 | -29 | -6 | +21 |
| | Winter | -15 | -5 | +4 | -23 | -9 | +6 | -40 | -16 | +12 |
| | Spring | -16 | -6 | +4 | -25 | -9 | +6 | -44 | -17 | +12 |
| Potential evaporation (%) | Annual | +2 | +3 | +5 | +3 | +5 | +8 | +5 | +9 | +15 |
| | Summer | +1 | +3 | +5 | +2 | +5 | +8 | +4 | +9 | +15 |
| | Autumn | +2 | +4 | +6 | +3 | +6 | +11 | +7 | +12 | +20 |
| | Winter | +2 | +5 | +9 | +3 | +8 | +15 | +6 | +16 | +29 |
| | Spring | 0 | +2 | +4 | 0 | +3 | +7 | +1 | +6 | +13 |
| Wind speed (%) | Annual | -5 | 0 | +4 | -8 | 0 | +6 | -15 | -1 | +12 |
| | Summer | -5 | +3 | +11 | -9 | +4 | +19 | -16 | +8 | +36 |
| | Autumn | -9 | -2 | +4 | -14 | -3 | +7 | -27 | -5 | +14 |
| | Winter | -7 | -1 | +5 | -12 | -2 | +8 | -23 | -3 | +16 |
| | Spring | -8 | 0 | +6 | -14 | -1 | +10 | -26 | -1 | +19 |
| Relative humidity (%) | Annual | -1.3 | -0.4 | +0.4 | -2.1 | -0.6 | +0.7 | -4.0 | -1.2 | +1.3 |
| Solar radiation (%) | Annual | -1.0 | +0.3 | +1.9 | -1.6 | +0.5 | +3.1 | -3.2 | +0.9 | +6.0 |

### 6.5.3 Potential responses of plant functional types to climate change

The vegetation of the region consists of sclerophyll-dominated plant communities, situated on shallow, infertile soils and highly varied terrain (Keith 2004). High plant diversity is apparent across life forms (e.g. overstorey trees – predominantly eucalypts), understorey shrubs (e.g. Casuarinacae, Fabaceae, Proteacae) and herbs (e.g. Cyperaceae, Restionaceae, Lilliacacae). Shrubs are prominent in most communities.

Empirical studies examining responses of typical sclerophyll vegetation to fire regimes (Keith and Bradstock 1994; Cary and Morrison 1995; Morrison *et al.* 1995; e.g. Bradstock *et al.* 1997; Keith *et al.* 2007; Tozer and Bradstock 2003) show that composition and abundance of this vegetation is sensitive to variations in the length of the inter-fire-interval (IFI). Such effects reflect life history attributes of species and demographic capacity to cope with differing fire regimes, as well as inter-specific competitive effects (e.g. Keith *et al.* 2007). Pausas *et al.* (2004) demonstrated that a minimum set of attributes, based on presence/absence of resprouting and persistent fire-cued seedbanks, was sufficient to account for responses of woody plant diversity to fire regimes. Four functional types can be defined on this basis. These are (Table 6.3): resprouters with persistent seedbank (hereafter R+P+); seeders with persistent seedbank (R-P+); resprouters with transient seedbank (R+P-) and; seeders with transient seedbank (R-P-).

**Table 6.3.** The functional type scheme used to examine vegetation sensitivity to climate change and fire regimes (after Pausas *et al.* 2004).

| | **Resprouting ability (R)** | |
|---|---|---|
| **Seed bank persistence (P)** | Species that resprout after fire, either from basal or epicormic shoots (may sometimes be killed by fire) | Species that are usually killed by fire, causing 100% crown scorch (may sometimes resprout, usually after low-intensity fire or incomplete crown scorch) |
| Species with some carry-over of viable seeds from year to year, including those with a serotinous seed bank (canopy-held, released by fire or drying out) or persistent soil (physical or physiological dormancy broken by heat or other fire-related cue) seed bank. | R+P+<br>Resprouters with persistent seed bank | R-P+<br>Seeders with persistent seed bank |
| Species with short-lived seed usually released soon after maturity. Viable seed persists for no more than one year. | R+P-<br>Resprouters with transient seed bank | R-P-<br>Seeders with transient seed bank |

Resprouting ability and seed bank types may be related to gradients of moisture and resultant community productivity (Vesk and Westoby 2004; Clarke *et al.* 2005b; Pausas and Bradstock 2007). In particular, dependence on resprouting may be positively correlated with moisture availability, while persistent seed banks may be negatively correlated. While the mechanisms underlying these trends are subject to debate, they provide a useful framework for understanding how functional responses of plant species may be affected by shifts in moisture and temperature under climate change (Pausas and Bradstock 2007).

Floristic data from regional vegetation surveys were compiled (Bradstock and Hammill 2008; Fig. 6.15). Information on fire response (NSW DECC Fire Response database) was used to characterise the resprouting (R) and seed bank persistence (P) status of species recorded in plots from *Eucalyptus* open forest and woodland on sandstone-derived soils (Table 6.3). These vegetation communities (the most common in the region – from the coast to the mountains) span a wide range of average annual temperature (11–17°C) and rainfall (600–1800 mm).

A total of 896 species (46% of all species present in the plot-based data) were completely characterised by functional type. Of those, the most common functional types were R+P+ and R-P+ (43% and 36% of species characterised, respectively), with R+P- and R-P- less common (16% and 5%, respectively). Shrubs comprised 55% of all species characterised. Generalised linear models (GLM) were used to explore relationships between functional types (proportion of species present), and temperature and moisture variables (ANNRAIN, MOIST, WARMRAIN, ANNTEMP, MINTEMP). Results for the most common functional types are presented here (R+P+, R-P+).

Moisture availability and temperature were found to have significant effects on the two most common functional types (R+P+ and R-P+) when examined for all species combined and shrubs only (Fig. 6.16). Proportional occurrence of R+P+ increased with increasing moisture (soil moisture – all species; warm season rainfall – shrubs) but decreased with increasing temperature. The opposite trends were found for R-P+, which decreased with increasing moisture (warm season rainfall – all species and shrubs) but increased with increasing temperature. Notably, the proportion of deviance explained by the preferred, significant models was low (about 15%), meaning that other factors (e.g. fire regimes) may have considerable influence on the distribution of functional types.

The results indicate potentially complex effects of climate change on vegetation and its capacity to cope with changing fire regimes (Fig. 6.16). Predicted increases in temperature will potentially disadvantage resprouters, particularly those species restricted to cooler habitats in the higher, mountainous parts of the region. Any decrease in rain may disadvantage resprouters in general, but the situation may be more complex for resprouting shrubs, which were positively related to warm season rainfall. Irrespective of the uncertainty surrounding predicted trends in overall rainfall, there is greater certainty of a predicted shift toward summer rainfall in the region (CSIRO and Bureau of Meteorology 2007). This trend would favour resprouting shrubs, contrary to effects of temperature. The influence of temperature appeared to be stronger, as indicated by the respective slopes of the fitted lines. A similar conclusion may apply to obligate seeders, where predicted trends in temperature would tend to be favourable (all species and shrubs) but trends in season of rainfall may be unfavourable. With obligate seeder shrubs in particular, the response to temperature was stronger than that to warm-season rainfall.

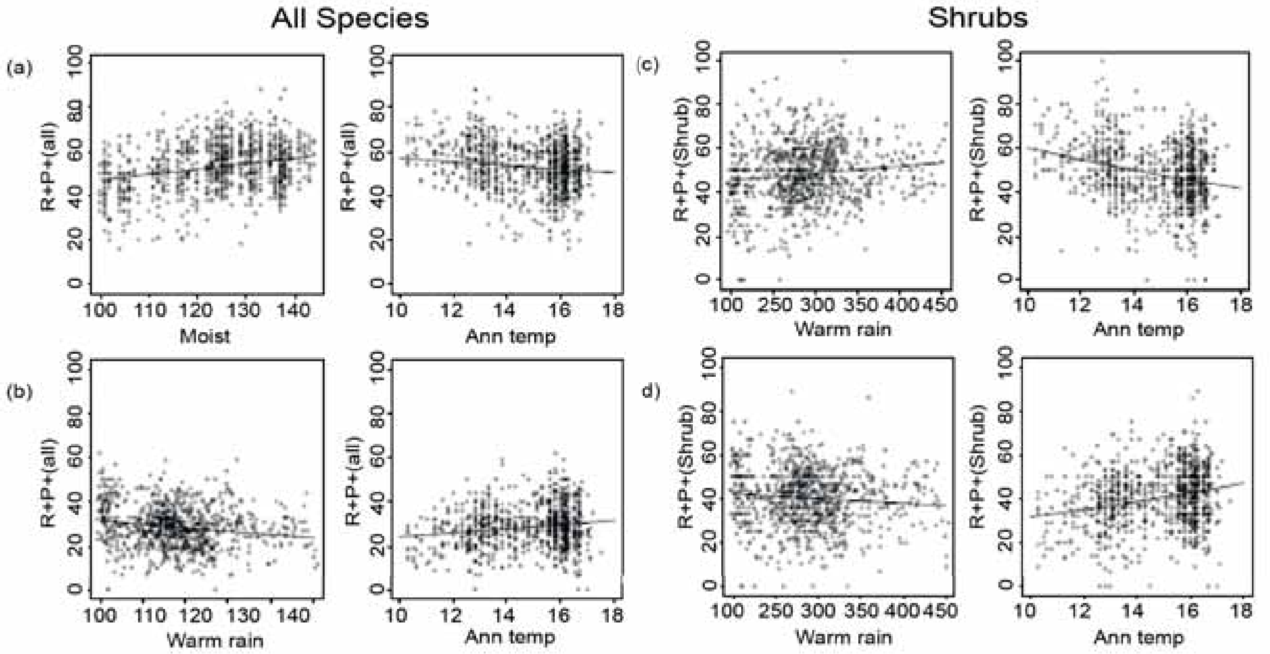



**Fig. 6.16. Effect of climate variables on the functional type response (y-axis), proportion (%) of species.** Data shown are for the most common functional types: R+P+ (resprouters with persistent seedbank; a, c) and R-P+ (seeders with persistent seedbank; b, d) for all species (top) and for shrub species only (bottom). Source: Bradstock and Hammill (2008).

In conclusion, increases in temperature (the most certain outcome of climate change) could disadvantage that component of the vegetation (resprouters with a persistent seed bank; shrubs in particular) that is most resilient to variation in fire interval. Higher temperatures may favour obligate seeders with persistent seedbanks – the functional type most sensitive to change in fire frequency (length of inter-fire interval). For shrubs, any such shift may tend to reinforce competitive dominance by obligate seeders. Responses to predicted rainfall changes illustrate the complexity and uncertainty of the problem. Nonetheless, for the pivotal shrub component, the temperature response may outweigh any contrary response to rainfall. Climate change has some potential to shift vegetation composition toward functional types that are more sensitive to any shift in length of between-fire interval.

### 6.5.4 Drivers of contemporary fire regimes: effects of weather and fuel

Recent research (M.G. Turner *et al.* 2003; Moritz *et al.* 2004; Westerling *et al.* 2006; Seydack *et al.* 2007; Thompson *et al.* 2007; ) indicates that variations in fire area and intensity may be strongly affected by weather variations in a range of ecosystems. These studies indicate that such effects may outweigh the influence of fuel variations on area burned. This suggests that fire regimes will be acutely affected by climate change in the future, via its effects on weather, and that management via manipulation of fuel will be challenging.

Local insights on the relative influence of drivers of fire were explored at differing spatial and temporal scales via differing methodologies (i.e. remote sensing of fire severity and analyses of contemporary fire records in the Sydney Basin).

#### *Effects of weather, terrain and fuel on fire severity*

Recent work on mapping of fire severity via remote sensing (Chafer *et al.* 2004; Hammill and Bradstock 2006) provides opportunities for exploration of relative effects of weather, fuel and terrain on inferred fire intensity. Data on fire severity within a large fire (about 50,000 ha) in December 2001 in the lower Blue Mountains (Hammill and Bradstock 2006) were analysed to investigate effects of these influences. This fire burned under a wide range of weather conditions ranging from Moderate to Extreme FFDI.

Logistic regression analyses were used to explore effects of weather (Extreme or Non-extreme FFDI; EX and NEX, respectively), time since last fire (a surrogate for fuel) and terrain (topographic position – proximity to ridges and gullies; aspect; and slope) on fire severity. These analyses contrasted differing combinations of fire severity, representing thresholds in fire intensity that relate to the probability of suppression and key ecological effects. In particular, crown fires (i.e. partial or complete combustion of tree crown foliage) burn at intensities where suppression is unlikely, whereas understorey fires fall into an intensity range where suppression is possible (Cheney 1981; Gill *et al.* 1987).

Analyses indicated that there were significant effects of all independent variables on crown fire probability, but weather had the paramount effect (Table 6.4). Predictions of probability of crown fire from the final model, using Akaike Information Criterion (AIC) model selection criteria, illustrate the interactions and relative effects of all independent variables (Fig. 6.17).

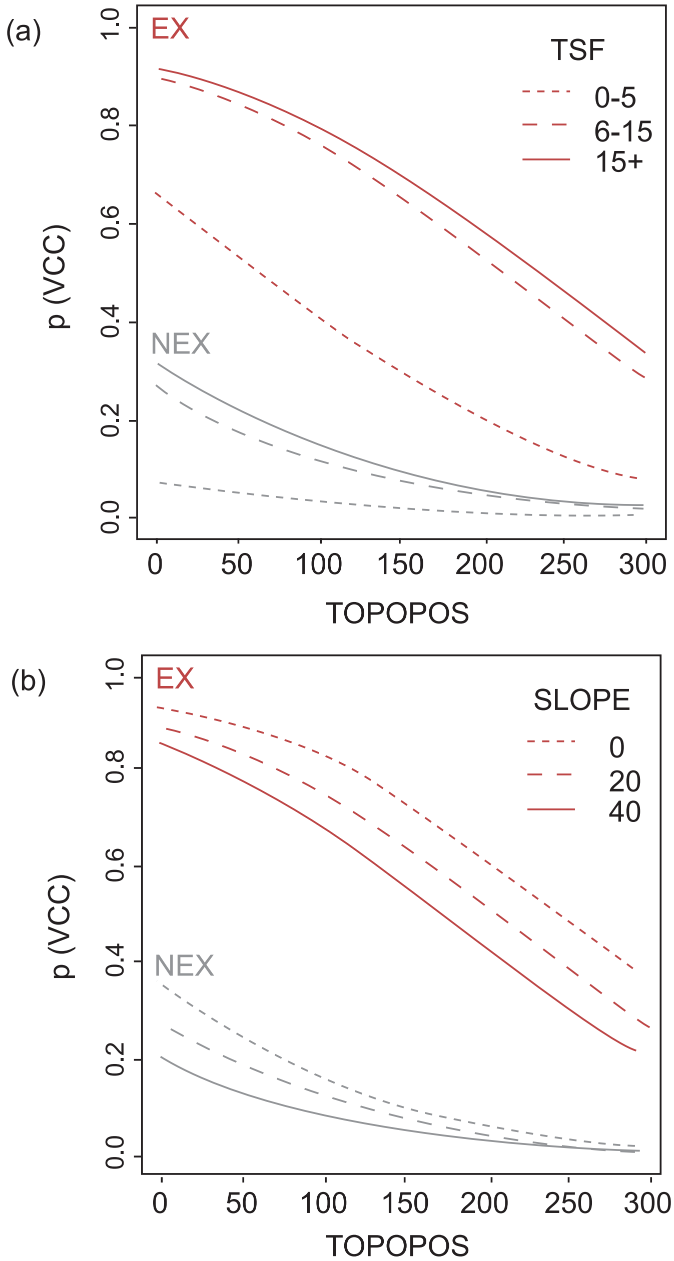



**Fig. 6.17. Probability of occurrence of crown fire [p(VCC)] relative to topographic position.**
TOPOPOS; m below ridge; (i.e. 0 indicates ridges, 300 the deepest gullies) under Extreme weather (EX; red lines) and Non-extreme weather (NEX; grey lines). Solid, dashed and dotted lines indicate the additional effects of: (a) time since fire (TSF; 0–5, 6–15, 15+ years since last burnt); and (b) slope (SLOPE; 0, 20 and 40 degrees).

**Table 6.4. Results (maximum likelihood parameter estimates) for the preferred model of crown fire probability in response to multiple variables.** These variables are: (a) weather (NEX – Non-extreme; EX – extreme), topographic position (TOPOPOS, m below ridge), time since fire (TSF; 0–5, 6–15, 15+ years since last burnt); and (b) slope (SLOPE; degrees). Aspect did not have a significant effect. Since WEATHER and TSF are factors, coefficient estimates are for each level relative to the reference level (in parentheses). P indicates the significance of the estimates (using the z distribution). Odds multipliers are the multiplicative effects of a one-unit/level increase of that variable on the odds of crown fire occurring. The larger the odds multiplier, the larger the effect of that variable on the probability of crown fire (Agresti 2007).

| Response | Predictor | Coefficient estimate ± SE | *P* | Odds multiplier |
|---|---|---|---|---|
| VCC | Intercept | -2.297 + 0.314 | <0.001 | |
| | WEATHER (NEX) | | | |
| | EX | 3.171 + 0.204 | <0.001 | 23.830 |
| | | | | |
| | | | | |
| | | | | |
| | | | | |
| | TOPOPOS | -0.010 + 0.002 | <0.001 | 0.990 |
| | TSF (0–5) | | | |
| | 6–15 | 1.505 + 0.349 | <0.001 | 4.504 |
| | 15+ | 1.716 + 0.307 | <0.001 | 5.562 |
| | SLOPE | -0.0196 + 0.0097 | 0.043 | 0.980 |

Under moderate (NEX) weather the probability of crown fire was relatively low, but under extreme (EX) weather it was high (Fig. 6.17). Topographic position had strong effects on crown fire probability, which declined from a maximum on ridges to minimum values near gullies. The effect of topographic position on crown fire probability under EX weather was equivalent to the magnitude of the effect of change in weather on ridgetops. Crown fire probability was also sensitive to fuel age, although to a lesser degree than either weather or topographic position. Crown fire probability was diminished by low fuel ages (0–5 years) relative to older fuel classes (>5 years). There was little effect on crown fire probability of fuels aged 6–15 years compared with fuels >15 years. Effects of fuel age on crown fire probability diminished on lower slopes under NEX weather (but not EX). Slope had minor but consistent effects on crown fire probability, with negative effects of steeper slopes.

These results indicate that the intensity of fires as interpreted from measures of severity (i.e. profiles of vertical damage in the vegetation) is likely to be highly sensitive to changes in FFDI. Elevation in FFDI will increase the probability of intense fires, as well as a possible commensurate increase in area burned due to an elevation in rate of spread. Changes to rates of fuel accumulation could temper this effect, although the effects of fuel on fire severity and intensity were more limited and confined to a short temporal window (five years time since fire). Effects of the non-weather elements of climate change (e.g. drought or elevated $CO_2$) on early post-fire accumulation of litter and near-surface fuels may therefore be crucial to understanding future implications for fire regimes.

The ability to manage fire through prevention and suppression is substantially influenced by weather, terrain and, to a lesser degree, fuel. The results indicate that the length of the temporal window of effectiveness of fuel manipulation is relatively short under a wide range of weather conditions, and is strongly influenced by terrain. This has important ramifications for strategic fire management, given the rugged terrain and accessibility for suppression resources in the Sydney region. While the probability of high-intensity fires is at its maximum on ridges, these may be the most practical parts of the landscape in which to strategically treat fuel, given the nature of roads and trails in typical dissected terrain.

*Effects of weather based on fire history analyses*

Large fires are important because they account for the bulk of the area burned in most ecosystems over time, even though they may be relatively rare (Reed and McKelvey 2002; Boer *et al.* 2008). Bradstock *et al.* (2009; in press) found that >95% of the area burned in representative landscapes of the Sydney Basin region was accounted for by fires >1000 ha (sample period 1960–2003). Such large fires comprised about 5% of total recorded fires for the sample areas. They found that a model based on the Drought Factor (DF) – summed with an 'ambient' index of weather, representing effects of maximum daily temperature, windspeed and humidity – provided the best predictor of the day of incidence of large fires (i.e. ignitions yielding a final area burned >1000ha).

The probability of large fire incidence increased as a function of the sum of these indices in case study landscapes within the region, e.g. the Blue Mountains (Fig. 6.18), with an even or greater than even chance of large fires occurring at maximal values of each index. Large fires are unlikely in the absence of either drought (i.e. high values of DF) or severe ambient conditions. The result may reflect the structure of landscapes in the region. Rugged terrain provides sheltered aspects where fuel remains moist under 'normal' conditions (low DF). Such aspects provide natural, meso-scale barriers to fire spread and fires tend to remain small (<1000 ha). Strong winds may allow some fires to bridge these discontinuities. Drought may provide widespread opportunities (i.e. increased landscape connectivity) for development of large fires, through removal of these natural barriers to fire spread.

In summary, results from studies conducted at different spatial scales (patch and regional scales) lead to a common conclusion. Weather conditions have a strong effect on fire severity, inferred intensity and area burned by large fires. Severity and inferred intensity were less sensitive to variations in fuel (at the patch scale) and it is likely that area burned will be similarly affected. Thus, fluctuations in weather over time may account for the bulk of variation in area burned, as found in other temperate shrub dominated systems (e.g. Seydack *et al.* 2007) and more generally (Marlon *et al.* 2008). Further regional-scale analyses of area burned in the Sydney region have reinforced this conclusion (Price and Bradstock unpub.). Fire will therefore be strongly sensitive to climate change in these ecosystems.

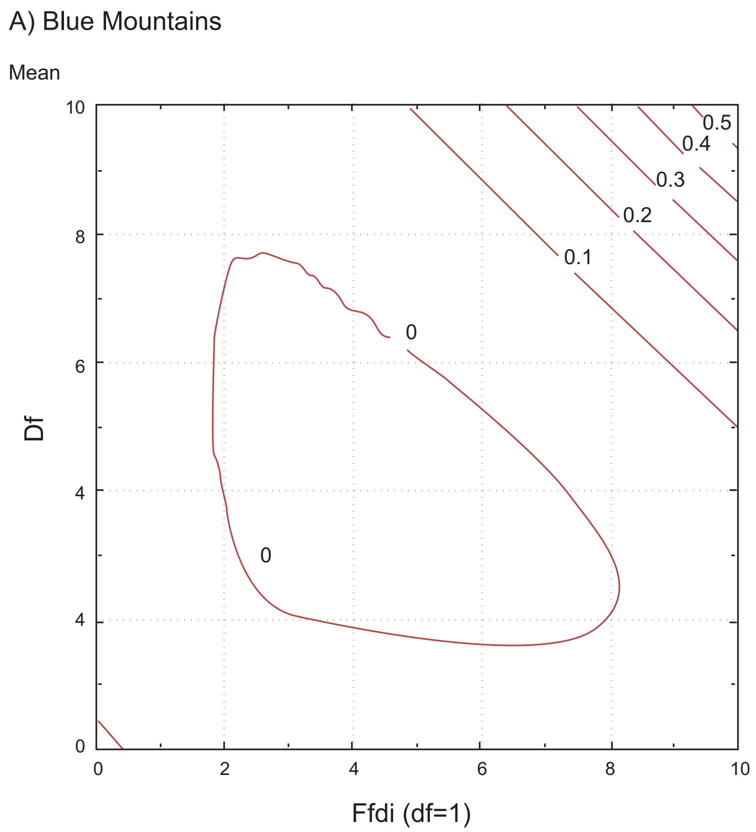



**Fig. 6.18. Predicted mean probability of large fire ignition days (isolines) in relation to drought and weather.** Drought Factor (DF) and ambient weather (FFDI (DF=1) indices based on historical data from the Blue Mountains (1960–2003). (After Bradstock *et al.* 2009 in press).

### 6.5.5 Effects of climate change on fire regimes

Cary (2002) used the FIRESCAPE fire regime simulation model to examine responses of fire regimes to climate change in the Australian Capital Territory region (see also section 4.5). FIRESCAPE is a landscape-scale simulator that ignites and spreads fires using models of fundamental processes underlying fire behaviour. This approach explicitly deals with effects of weather, fuel and terrain at a variety of spatial and temporal scales (Keane *et al.* 2003). As a result, simulations dealing with large spatial and temporal scales can be performed, providing an experimental basis for exploring the effects of variation in these drivers.

The FIRESCAPE model was used to examine changes in climate and management strategies in the Sydney region via three case studies involving representative landscapes on the coast: Hornsby/Ku-Ring-Gai; Woronora/Illawarra; and Blue Mountains (Bradstock *et al.* 2008). Results for the Blue Mountains case study are presented here, representing an area of about 190,000 ha (of which about 150,000 ha is available to burn), centred on the Grose Valley (Fig. 6.19). This area contains the major urban corridor within the Blue Mountains local government area and a major portion of the Blue Mountains National Park, Blue Mountains Heritage Area and the Warragamba water supply catchment.

The 2050 scenarios of change in FFDI produced by Hennessy *et al.* (2005) were used to simulate effects of change in fire-weather on area burned, IFI and fire intensity (Bradstock *et al.* 2008). The maximum and minimum scenarios of change in fire danger were used (Hennessy *et al.* 2005), and are referred to here as 'High' and 'Low', respectively. The model was calibrated to produce diagnostics of area burned and spatial pattern that matched the contemporary fire history for the case study. The model was initially run for a 26 year time-series (50 replicates) of contemporary, observed fire danger (1977–2003). Ignition rates (lightning plus human) were set at the observed level for the period. Simulations were then repeated using predicted weather for 2050, with ignition rates held constant.

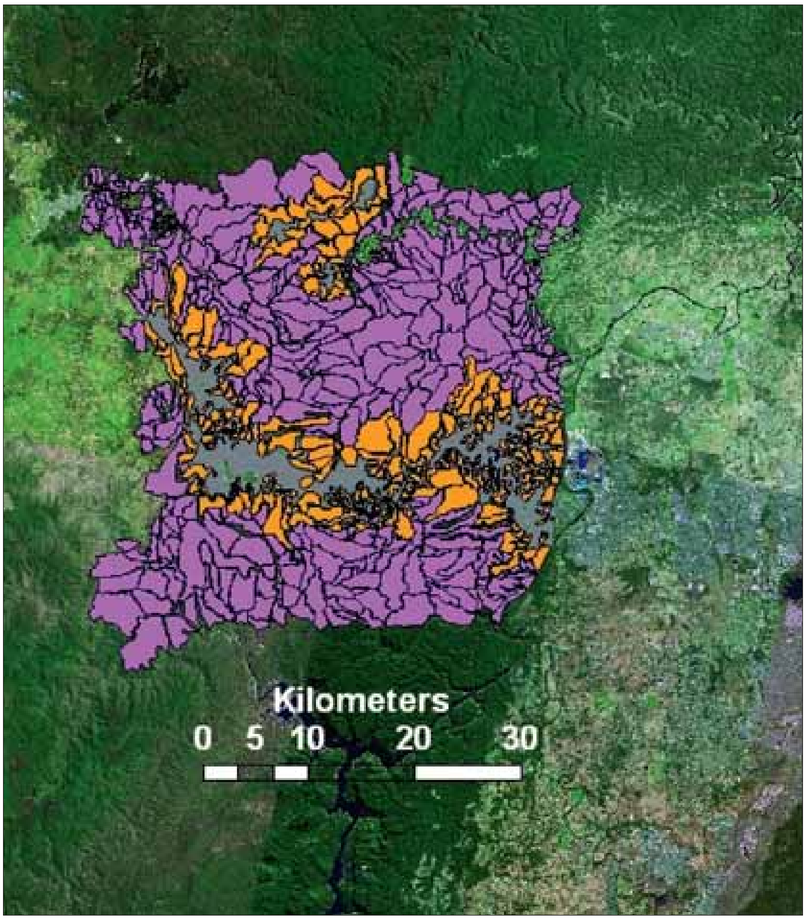
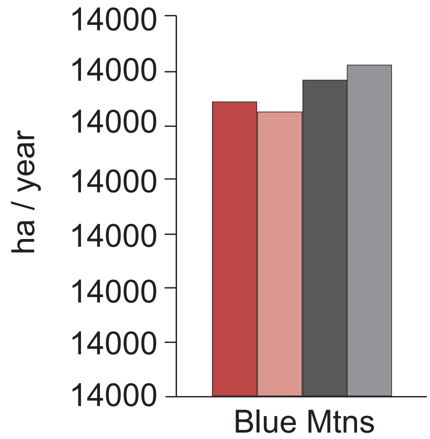



**Fig. 6.19. Blue Mountains case study area for exploration of effects of climate change on fire regimes and risk using the FIRESCAPE model.** Grey shading is urban area, prescribed burning blocks are demarcated by lines, orange blocks are urban edge blocks, and purple blocks are non-edge (remote). Source: Bradstock *et al.* 2008.

The results predicted an increase in area burned of 13–19% (Fig. 6.20), which exceeded the corresponding level of change in FFDI (4–11%) estimated for Katoomba (Bradstock *et al*. 2008). It should be noted that the scenarios of Hennessy *et al*. (2005) are conservative. These involve an adjustment of daily weather parameters, using the contemporary time-series of daily weather (1977–2003) as a base. The approach therefore conserves the temporal sequence of days, but does not account for any rearrangement of the sequence of days that could occur under future climates. Thus contemporary intra- and inter-annual patterns of drought are conserved in the 2050 scenarios.




| | |
|---|---|
| Observed | 10719 |
| Current | 10450 |
| Low | 11593 |
| High | 12136 |

**Fig. 6.20. Simulated change in annual area burned in the Blue Mountains.** Case study area is approximately 190,000 ha – see Fig. 6.19. Predictions simulated using FIRESCAPE and alternative fire-weather scenarios for 2050 (Low and High) under climate change (Hennessy *et al.* 2005). Obs = observed data for 1977–2003; Current = modelled for 1977–2003; Moderate = moderate 2050 climate change scenario; High = high climate change scenario. (After Bradstock *et al.* 2008).

### 6.5.6 Effects of fire management and climate change on risks to biodiversity and people

Strategies of prescribed burning were explored within the Blue Mountains case study using FIRESCAPE. This involved varying levels of 'effort' (i.e. area treated per annum) and spatial pattern of treatment. These were chosen to represent contemporary approaches and commonly debated alternatives. Outcomes of differing strategies were explored for using the fire danger information under contemporary climate and the 2050 scenarios of Hennessy *et al.* (2005).

Four strategies, which represent the main alternatives, were compared (Bradstock *et al.* 2008): (i) Edge – treatment of blocks abutting the urban interface only (Fig. 6.19); (ii) Unconstrained – treatment of blocks at random; (iii) Constrained – treatment of both 'edge' and 'non-edge' in a 1:1 ratio; and (iv) Linear – strategic treatment of blocks in the landscape in linear arrays running approximately north–south and east–west. Variations in annual level of treatment effort in each case were achieved by varying the number of blocks treated. The range of effort explored under differing spatial strategies varied due to differences in the number of available blocks available in each strategy (Bradstock *et al.* 2008). Blocks were only available for treatment at five years after last fire (planned or unplanned). Simulations were carried out, using ignition rates, level of replication and the climate scenarios described previously. Various diagnostic outputs were used to assess risk to biodiversity and people.

Biodiversity effects were evaluated using a 'thresholds of potential concern' (TPC) (Bond and Archibald 2003) approach based on life history attributes of species and their responses to fire regimes. Bradstock and Kenny (2003) provided a spatial/temporal TPC approach for assessing the status of vegetation in relation to IFI. This was developed locally (IFI is known to affect the persistence of obligate seeders – see above). It 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% of the area) then extinction may be likely. Critical inter-fire intervals for dry sclerophyll woodland/open forest covering >80 % of the study landscape are <7 years and >30 years (e.g. Bradstock and Kenny 2003; Kenny *et al.* 2004). These respectively represent significant immaturity and senescence risk to obligate seeders. The landscape IFI distribution is therefore a direct indicator of risk to 'IFI-sensitive' plant species.

Fire intensity has important consequences for biodiversity conservation and catchment protection. Fires >10,000 kW/m are high-intensity crown fires in these dry sclerophyll forests and may be lethal to arboreal vertebrates, such as koalas (Lunney *et al.* 2007), which are common in the region, though explicit linkages with extinction probability are not available. Large fires may also be significant for such vertebrate taxa. Such species may depend on immigration for recovery within patches subjected to high intensity fire. Recovery from refugia may therefore be delayed or impeded if burned areas are large (Bradstock *et al.* 2005). Annual, mean probability of large fires in the case study (i.e. fires >1000 ha) provided an index of this effect.

High-intensity crown fires may also lead to temporary loss of soil surface water repellency. This increases the potential for movement of soil from upper slopes if high-intensity rainfall occurs in the first year after fire (Shakesby *et al.* 2007). Adverse consequences for water quality in streams and reservoirs can result. These effects may be amplified by fire size. Mean annual probability of occurrence of high-intensity fires and large fires are therefore indirect indicators of risk to these 'intensity-sensitive' ecological values.

The results (Figs. 6.21–6.23) predict that fire regimes (IFI distributions, high-intensity and large fire probabilities), may be affected by climate change and prescribed fire strategy (pattern and treatment level).

Under contemporary climate and contemporary management (prescribed burning average of about 1400 ha per annum or about 1% of the available case study landscape – approximating the 'Constrained' spatial strategy), the modelled IFI distribution was not adverse for obligate seeders in dry sclerophyll vegetation; i.e. the predicted sum of adverse short and long intervals was about 20% (Fig. 6.21). Fire regimes were within a landscape-level range likely to be compatible with the persistence of plant functional types sensitive to IFI.

Prescribed burning has the predicted potential to substantially alter the landscape-level IFI distribution (Fig. 6.21). Adverse, short IFI proportion increased with treatment level under all strategies under contemporary climate (i.e. there was a minimal effect of spatial pattern). At high levels of 'Unconstrained' treatment, the proportion of adverse, short IFI increased to high levels (i.e. about 40%). Long, adverse IFI were largely unaffected by either strategy or annual level of treatment, and affected about 10% of the area of these plant communities. As a consequence, the overall level of adverse IFI (i.e. sum of adverse short and long IFI; Fig. 6.21) reached a level (about 50% of landscape area) that corresponds with high probability of extinction of obligate seeder plant species, under high levels of Unconstrained treatment.

Climate change was predicted to have little impact on this scenario. A general decrease in IFI occurred (Bradstock *et al.* unpublished data) but this was mainly within the acceptable IFI domain (i.e. 7–30 year range). Overall the results predicted that high levels of Unconstrained treatment (>15% of the landscape treated per annum) under a contemporary climate (Fig. 6.21) may tend to be adverse to IFI-sensitive plant species, due to high levels of short IFIs in particular. Climate change did little to alter this (Fig. 6.21).

Predicted mean annual probability of high-intensity (i.e. >10,000 kW/m intensity) crown fires (at point or patch scale) was affected by prescribed burning strategies and climate change (Fig. 6.22). This probability declined with increasing prescribed burning effort, although spatial pattern had little effect. Large amounts of prescribed burning (about 10 % of the landscape treated per annum) halved this probability (cf. zero treatment). The high climate change scenario (i.e. maximum increase in FFDI) increased high-intensity fire probability by about 15–25% (Fig. 6.22), while the low scenario caused an increase of <10%. The change in probability caused by the high scenario of climate change would require a fivefold increase over contemporary levels of prescribed burning (about 1400 ha treated per annum – see previous discussion) to counteract it.

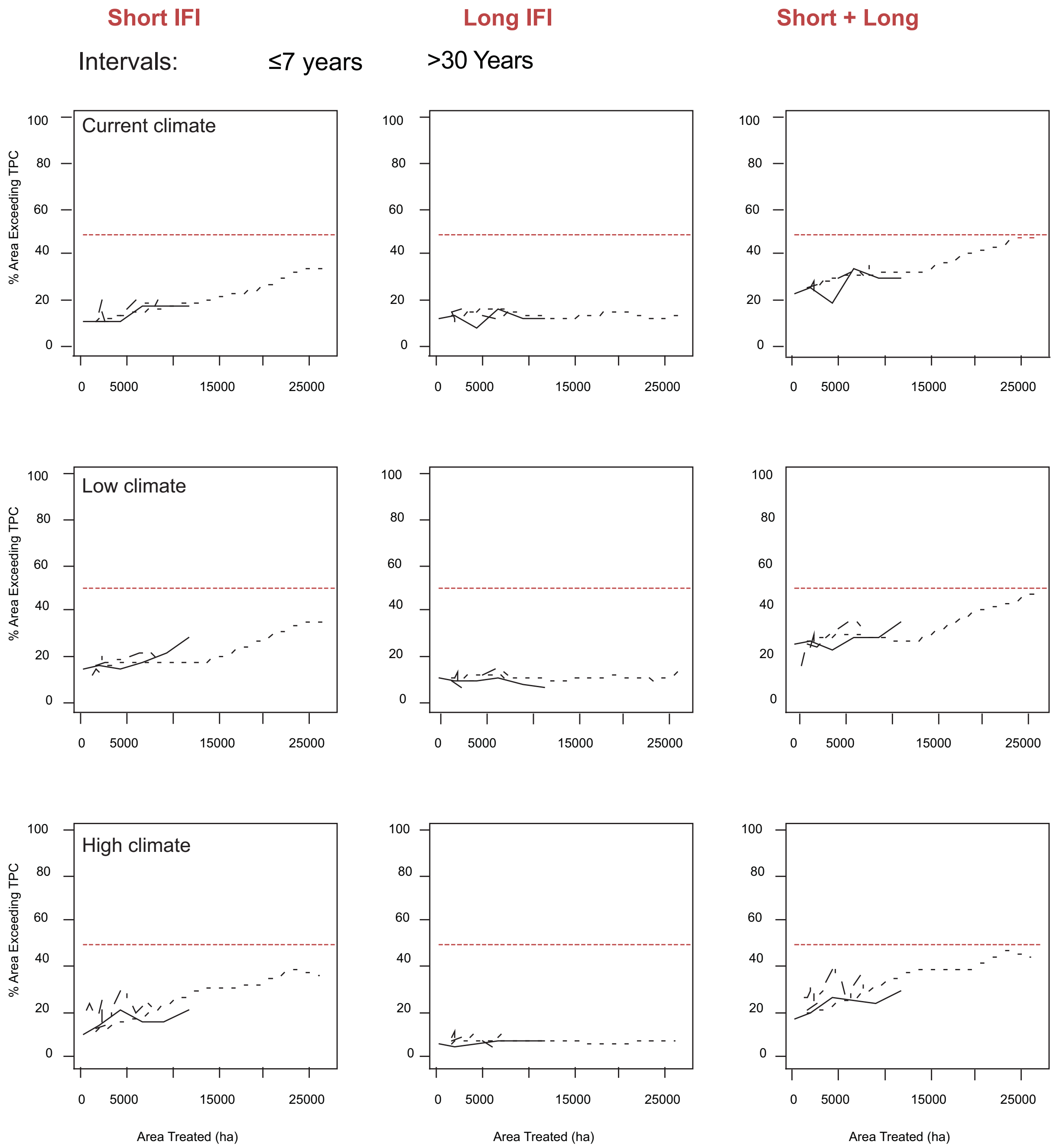
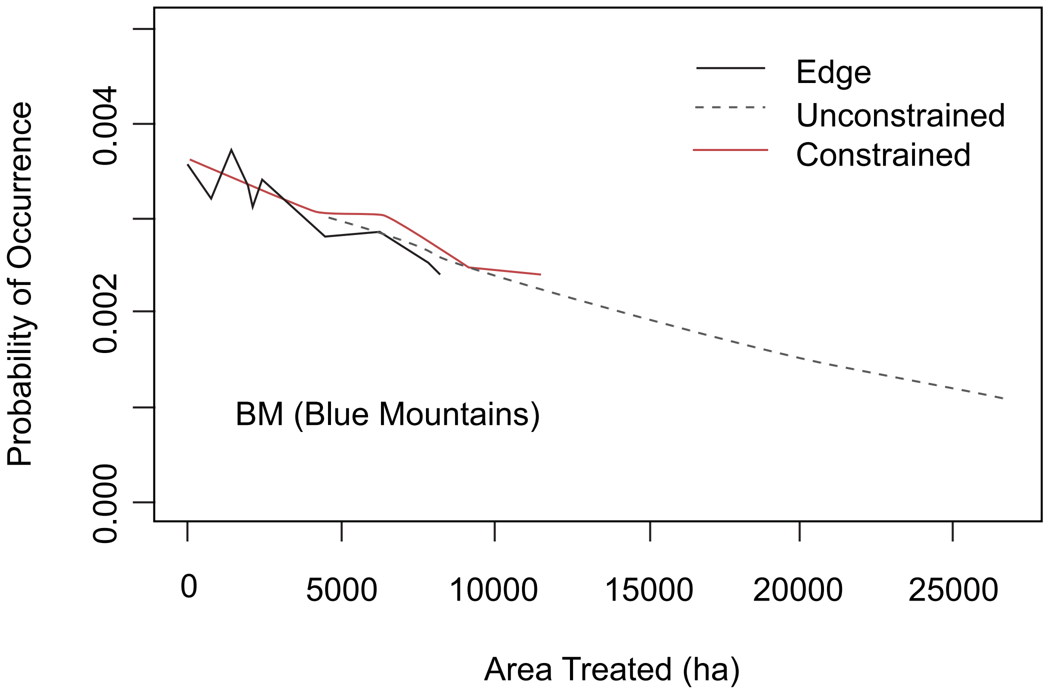



**Fig. 6.21. Simulated effects of prescribed fire, spatial strategy and treatment level (ha prescribed burnt per annum) on the landscape fire interval distribution (% area) in the Blue Mountains.** Key: Categories of intervals are thresholds of potential concern (TPCs) for inter-fire interval (IFI) sensitive plant species: adverse short IFI <7 years; adverse long IFI >30 years. Rows of panes represent modelled effects under different climate scenarios. The different lines represent different spatial strategies of prescribed burning (Edge, Unconstrained, Constrained, Linear) as per the legend in Fig. 6.22a. (After Bradstock *et al.* 2008).

a)




b)

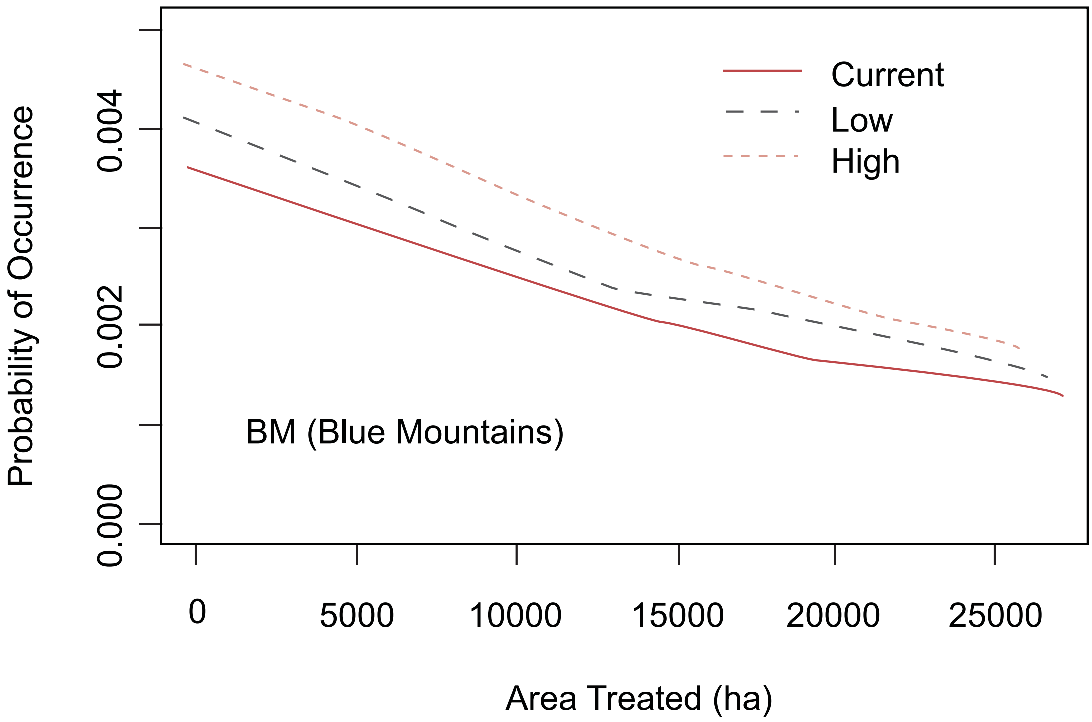
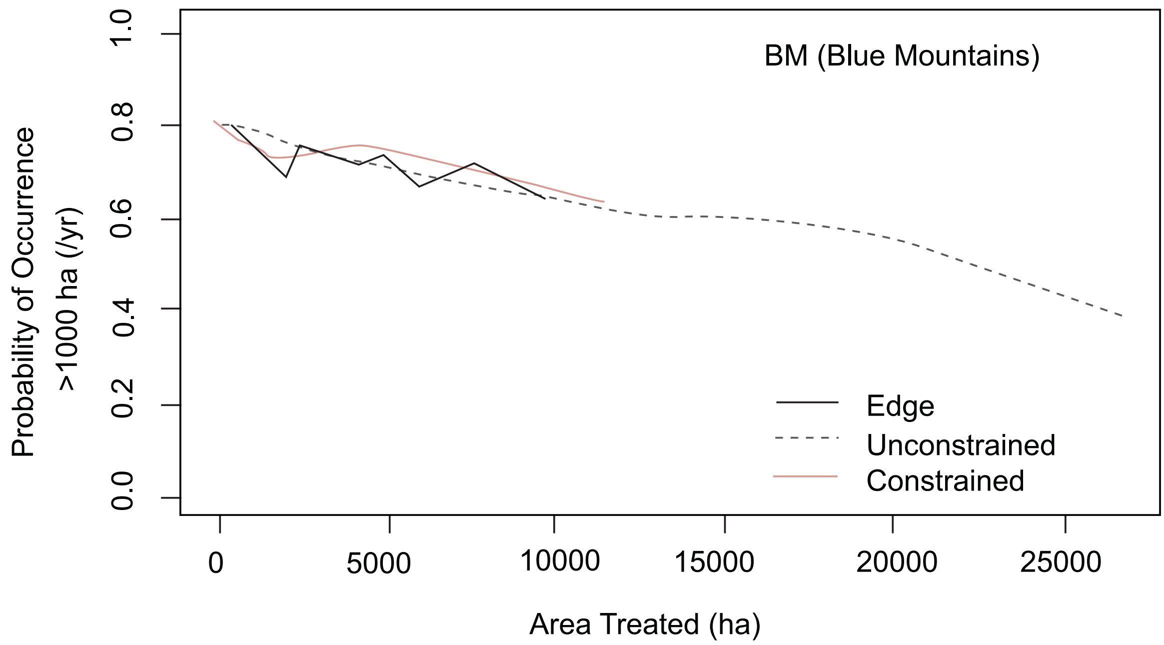



**Fig. 6.22. Simulated mean annual probability of high-intensity crown fires (>10,000 kW/m) fires in the Blue Mountains in relation to prescribed fire, spatial strategy and treatment level (ha prescribed burnt per annum), under contemporary climate.** (a) Under contemporary climate. (b) Effects of climate change on large fire probability; given for the Unconstrained strategy only. (After Bradstock *et al.* 2008).

a)




b)

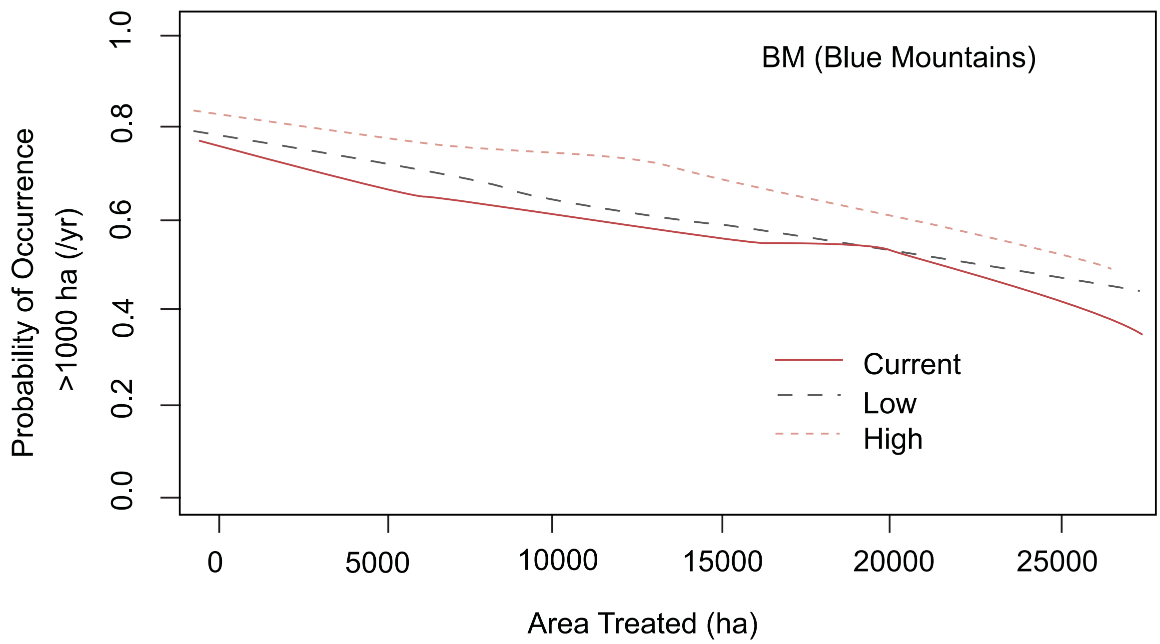
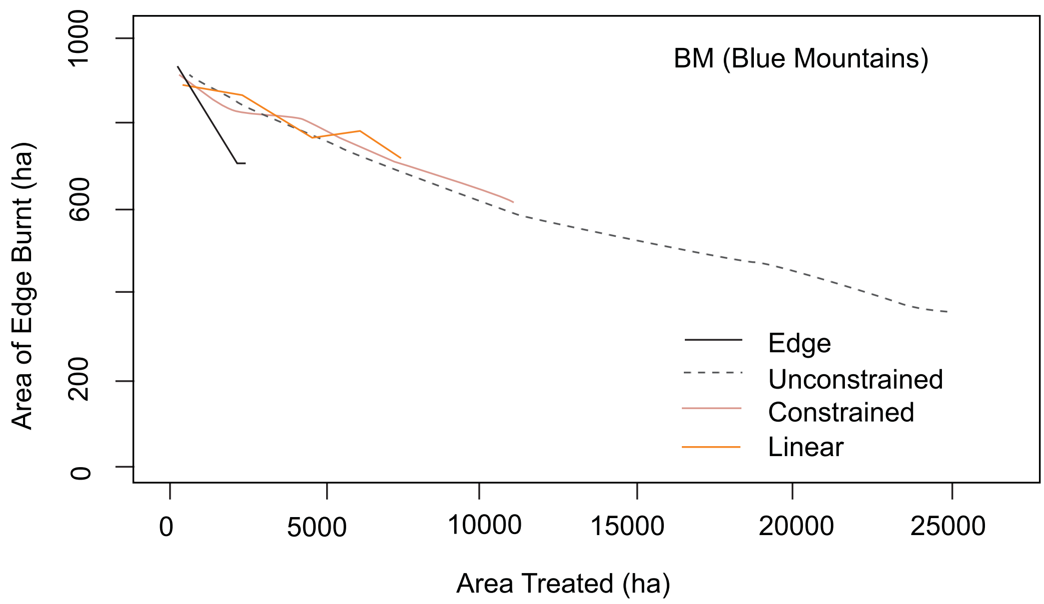



**Fig. 6.23. Simulated mean annual, probability of large fires (>1,000 ha) fires in the Blue Mountains in relation to prescribed fire, spatial strategy and treatment level (ha prescribed burnt per annum).** (a) Under contemporary climate. (b) Effects of climate change on large fire probability; given for the Unconstrained strategy only. (After Bradstock *et al.* 2008).

a)




b)

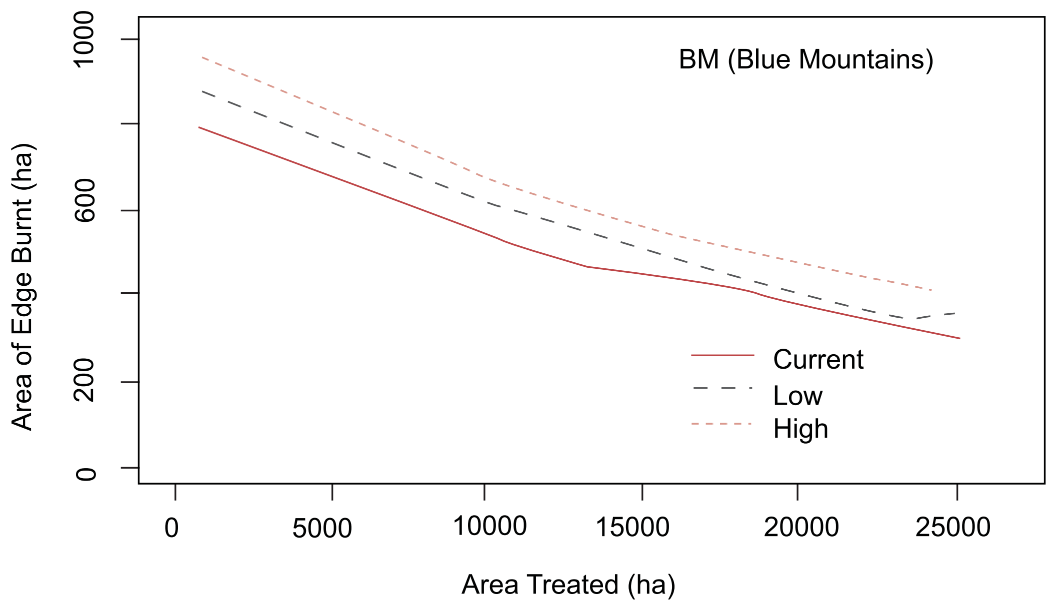



**Fig. 6.24. Simulated mean area adjacent to the urban edge burnt by unplanned fires in the Blue Mountains in relation to prescribed fire, spatial strategy and treatment level (ha prescribed burnt per annum).** (a) Under contemporary climate. (b) Effects of climate change on large fire probability; given for the Unconstrained strategy only. (After Bradstock *et al.* 2008).

Predicted mean annual probability of large fires (i.e. >1000 ha) was negatively affected by amount of prescribed burning (Fig. 6.23). This probability was insensitive to spatial pattern of treatment (Fig. 6.23). Very large amounts of Unconstrained treatment (about 20% of the landscape per annum) are predicted to be needed to halve the probability of large fires (Fig. 6.23). The high climate change scenario increased large fire probability by about 10%, with a smaller change predicted under the low scenario (Fig. 6.23). A greater than fivefold increase in prescribed burning effort would be needed to counteract the effect of the high climate change scenario (Fig. 6.23).

Risks to life and property will be affected by the area (i.e. of urban edge blocks) of urban interface affected by unplanned fire. The area of urban interface affected by unplanned fires was significantly affected by prescribed burning strategy and level of treatment (Fig. 6.24). Affected area declined with increasing level of treatment, though this decline was significantly greater under the Edge treatment compared with the other strategies (amongst which there were no differences). However, a greater predicted reduction in affected area occurred under moderate to high levels of treatment by non-edge strategies, compared with the Edge strategy (Fig. 6.24). Thus treatment (Unconstrained treatment only) of 10,000 ha per annum by non-Edge strategies achieved a predicted reduction in affected area of >30% and a treatment level of 25,000 ha per annum (about 16 % of the available landscape) achieved a reduction of about 65%.

Climate change increased the mean area of urban edge affected by unplanned fires under the Unconstrained prescribed burning strategy (Fig. 6.24). The high climate change scenario had a major effect in this regard (>20% increase). A greater than fivefold increase in treatment would be required to counteract the effect of the high climate change scenario in this regard, while a lesser increment (twofold to fourfold increase) would be need to counteract effects of the low scenario of change.

### *Summary – interactive effects of climate change and fire management*

These model predictions 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 the relative degree of consequential risk posed to these values by the condition of the landscape, as affected by prescribed fire. Is there an optimum strategy (prescribed burning pattern 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?

Current levels of treatment (about 1% treated per annum) resulted in a relatively small degree of predicted risk reduction (about 5% change from level under zero prescribed fire) to urban, catchment and intensity-sensitive biodiversity indicators (i.e. probability of high intensity and large fires, Figs. 6.22–6.24). Such levels of treatment are not, however, predicted to result in high levels of risk to IFI-sensitive plant species (Fig. 6.21) when assessed at a whole-of-case-study scale. Increments such as doubling or tripling the current level of prescribed burning are predicted to result in a further, moderate reduction in risk (about 10–15% below the level under zero prescribed fire) to these indicators of urban, catchment and intensity-sensitive biodiversity values. Treatment of up to 10% of the landscape per annum yielded a 25–45% reduction in risk (cf. zero prescribed fire) to indicators of urban, catchment and SFI biodiversity values without raising levels of risk to IFI-sensitive biodiversity to a high level. 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 high level (i.e. resulted in a high extinction probability).

The results indicate that considerable residual risk (Gill 2005) to urban and intensity-sensitive ecological values will remain, irrespective of choice of strategy. No strategy is predicted to eliminate the component of consequential risk that is posed by the condition of the landscape. Levels of risk reduction to urban values in particular are predicted to be small under current levels of treatment. Feasible increments in treatment (e.g. a doubling or tripling of effort, with commensurate increases in cost) are predicted to result in relatively modest reductions in risk (Fig. 6.24). A halving of risk would require much higher levels of treatment (e.g. an order of magnitude or more over current levels). Such a scenario is probably implausible due to cost and resource constraints. Further economic analyses will assist in developing a better understanding on return on investment (i.e. percentage reduction in risk per level of expenditure) in this regard.

While the Edge strategy offered a higher level of risk reduction for the urban edge (i.e. a greater reduction in area of urban edge affected by unplanned fires per hectare treated; Fig. 6.24) than the other strategies, such an approach may be more expensive to implement (i.e. cost per hectare). Thus more hectares may be treated under the alternative strategies for a given cost, thereby yielding greater benefits in terms of reduction of the chance of high-intensity fires in particular (Fig. 6.22). The specifics of the ratio of cost of treatment of edge blocks versus more remote blocks will be important. The results indicate that the Edge treatment is about four times more effective in reducing the area of the urban edge affected by unplanned fires. The Edge treatment will therefore only be advantageous if cost per hectare does not exceed that of other strategies by about a factor of four. Analysis of the economics of the various treatment options, in terms of benefits and costs, is therefore an important priority.

Climate change was predicted to have the greatest impacts on risks to life and property, catchment values and elements of biodiversity that are sensitive to high-intensity fire. The results indicated that significant increases in prescribed burning (e.g. > fivefold) would be required to counteract the effects of the high climate change scenario. The plausibility of such an increase is open to question, based on financial and resource constraints. A more far-reaching understanding of the sensitivity of risk to differing facets of fire management (including prescribed burning), complemented by economic analyses, is needed to meet the challenge of risk mitigation under contemporary and future climates.

### 6.5.7 Conclusions

Differing methodologies and types of evidence produced insights into consequences of climate change for fire regimes and consequential risks to important values.

- The distribution of plant functional types is partly affected by gradients of temperature and moisture in accordance with theory. Shifts to a warmer and drier climate may alter the balance between obligate seeder and resprouter functional types to favour the former, thus predisposing vegetation to structural and compositional change under any regime of higher fire frequency.
- Fire intensity in the region may be highly sensitive to the effects of weather and terrain but less so to fuel accumulation. This suggests that, under the broad climate change scenario of warming and drying, the effects of warming on fire-weather, and consequently fire intensity and area burned, may outweigh any diminution of surface fuel loads caused by declining moisture. However, there is great uncertainty about whether fuel loads will increase as a consequence of $CO_2$ fertilisation, or decrease as a consequence of declines in available moisture. Further examination of these potentially contrary effects of climate change on fuels, and their consequent effects on fire regimes, will depend on resolution of this major uncertainty.
- Simulation modelling indicates that the effects of fire-weather on area burned and resultant fire regimes in the region are likely to be strong. The results indicate that changes in FFDI could result in larger shifts in area burned.
- The consequences of such changes in fire regimes for biodiversity and other values are, however, mixed. Shifts in IFI were predicted to be insufficient to significantly change landscape-scale extinction probability of IFI-sensitive plant species, assuming prescribed burning is held at current levels. In contrast, the probability of crown fires may increase by up to 20% under the high climate change scenario. Elevated risks to people and property, intensity-sensitive taxa (e.g. arboreal mammals) and soil stability may ensue.
- Prescribed burning can have major effects (positive and negative) on biodiversity, catchments and human protection. Moderate increases over current levels of prescribed burning (e.g. twofold to threefold increase) may be achieved without a concomitant increase in risk to the integrity of plant diversity in widespread, species-rich, dry sclerophyll vegetation. Commensurate risk reduction to urban and other ecological values sensitive to fire intensity is likely to be small. Larger increases in prescribed burning (i.e. > fivefold) would be needed to counteract effects of the high climate change scenario. Such an increase may not be feasible on the basis of cost and resources.
- Benefit-cost analysis of various prescribed burning options (treatment strategy; area burnt) as a function of risk to various landscape values (life, property, biodiversity, water) is an urgent research and development need.
- In their review of climate change impacts on the National Reserve System Dunlop and Brown (2008) indicated that biodiversity in south-eastern Australian sclerophyllous systems was potentially more susceptible to the effects of climate change than that of other regions in Australia. Thus, in this region, there is the potential of synergistic negative effects on biodiversity due to interactions between climate change and changing fire regimes.

There is much scope for further exploration of these insights. Simulation modelling offers the possibility of varying fuel loads to represent either a potential decline due to drying or an increase due to elevated $CO_2$. This could be done in interaction with the effects of changes in fire-weather (Bradstock *et al.* 2008). This would provide a formal exploration of the effect of weather and fuels on fire regimes and risks to important values. Similarly, empirical studies of effects of contemporary fire regime variations, using a stratified space-for-time across gradients of temperature and rainfall, may provide additional insights into the balance of plant functional types and resilience of vegetation composition (Bradstock and Hammill 2008, and unpublished data). These studies emphasise dependency of plant cover and herbaceous responses on temperature and moisture: warmer and drier conditions that are favourable for herbaceous resprouters with possible feedbacks into the fire regime. The ability of native and exotic species to exploit such changes also warrants investigation. Other effects – such as increased juvenile periods of woody obligate seeders under drier conditions, suggested by theory and empirical evidence (e.g. section 6.3) – have not been accounted for here. Such effects could alter the critical IFI windows demarcating the TPCs used above.

## 6.6 Conclusions from the four case studies

### 6.6.1 Insights into climate change, fire and biodiversity

We have analysed the projections for climate change in terms of potential impacts on fire regimes, and the impact of climate change–fire regime change on biodiversity. We have used four case studies from different parts of Australia to illustrate the potential approaches that may be taken to analyse the potential impacts of climate change on fire regimes, and how the potential interactive effects of climate change and fire on biodiversity may vary across the country.

Our analyses are, by virtue of the sheer scale of the problem, incomplete, and subject to considerable uncertainty; the 'cascading complexities' outlined initially in Fig. 1.1 remain as such. Nevertheless, key themes and principles are apparent from these case studies:

1. Projected increase in mean temperatures, as a consequence of climate change, is likely to lead to an increase in overall fire danger. This is a clear signal for all of the broad biomes analysed in these four case studies, and from sections 4 and 5.
2. Modelling the various effects of climate change on the drivers of future fire regimes is at a very early stage of development.
3. Notwithstanding point 2 above, the approach presented in the Sydney Basin case study is a generic approach that is likely to be fruitful in other regions of Australia. However, further examination of the impacts of climate change and changing fire regimes on biodiversity at regional and local scales will depend on the development of landscape fire models that are linked to dynamic landscape models of biodiversity change.
4. The impacts of climate change on fire regimes will depend on the extent to which projected increases in fire danger/weather act synergistically or antagonistically with climate change impacts on fuels (as a consequence of changes to moisture and $CO_2$). If the elements of climate change act synergistically, then the major uncertainty will be about the magnitude of change. If the elements act antagonistically, then both direction and magnitude will be uncertain.
5. The potential impacts of current land use practices or trends in land cover types may prove to be of equal or even greater significance to biodiversity than the projected impacts of climate change and changed fire regimes.

### 6.6.2 Key knowledge gaps and future research priorities

There are six broad critical gaps in knowledge that the case studies have highlighted:

1. The potential impact of climate change on fire-weather in other regions of Australia, as per the analyses undertaken for the south-east (Hennessy *et al.* 2005), as discussed at the end of section 4.2.3
2. The potential impact of elevated $CO_2$ on ecosystem production, and hence fuels, and on patterns of post-fire vegetation recovery. Elevated levels of $CO_2$ have the potential to alter fire regime components and ecosystem responses to fire, but the magnitude and direction of such impacts are highly uncertain
3. The effect of climate change on the frequency and intensity of: (i) droughts; and (ii) lightning. The temporal distribution of each may change with climate change, but current assessments of the impact of climate change on fire regimes have not incorporated such potential changes into projections
4. Evaluation of the relative importance of elevated fire danger, elevated atmospheric $CO_2$ and changing moisture availability as determinants of future fire regimes. This requires much more research and analysis, using regional climate change scenarios, regional fuel change scenarios and spatially explicit fire models
5. Regional case studies that explore climate change, fire regimes and biodiversity responses using linked, spatially explicit fire regime–biodiversity models, as we have presented for the Sydney Basin. There is also a need for additional and more detailed regional case studies. Our case studies have been limited to four regions so far, because of peculiarities of expertise and geographic location of consortium researchers, and by the limited development of fire models that can be parameterised for climate change scenarios that are linked to spatially explicit biodiversity models. In this regard, there is an ongoing and critical need for the development and refinement of a national data base on fire response
6. The impact of climate change and changing fire regimes on fauna. This has been, and remains, a pressing problem across all biomes and jurisdictions.

# 7. MANAGEMENT IMPLICATIONS: A RESPONSE FRAMEWORK FOR THE MANAGEMENT OF FIRE AND BIODIVERSITY UNDER CLIMATE CHANGE

The primary implication of our analyses is that fire management in areas managed for biodiversity will become more complex in the coming decades. This will be due to the:

- uncertainties associated with directions and magnitude of the effects of climate change on fire regimes
- uncertainties associated with the interactive effects of climate change and changing fire regimes on biodiversity
- potential trade-offs that will be required to manage biodiversity values in the face of (either perceived or actual) more frequent and/or intense fires
- potential need to modify fire regimes to account for multiple landscape values, such as biodiversity conservation, greenhouse gas abatement, carbon sequestration, smoke management and water yield.

Our approach has been one of scenario setting and evaluation, and this approach is likely to remain the most constructive for the foreseeable future as more regional analyses are undertaken and national principles and patterns emerge. Nevertheless, we present a selection of suggestions for ecologists, reserve managers and others to consider.

Our key message for managers – whether protected area managers or other land managers – is to prepare for change. Some change is inevitable as a consequence of climate change, as stressed by Dunlop and Brown (2008). Hence there is a pressing need to incorporate climate change and its interactions as a context for adaptive management, the most robust framework for integrating fire management and biodiversity conservation. Adaptive management is designed to operate in the midst of complexity, uncertainty and competing demands on resources. A key task, then, is to review, evaluate and enhance fire and biodiversity management systems, in the light of the potential changes that climate change could bring to fire regimes and their management. Heller and Zavaleta (2009) reviewed climate change–biodiversity management recommendations from the past 22 years, and illustrated the complexity of the problem. Adaptation requires greater regional, institutional coordination; incorporation of climate change into all planning and action; and greater efforts to address multiple threats. They discussed several generic, practical actions that could be undertaken (such as regional planning, more protected areas, enhancing landscape connectivity and building resilience) but there was no consideration of how to manage changing fire regimes.

In this context, then, we discuss six key topics of relevance to fire and biodiversity management: (1) uncertainty, and its implications: (2) the elements of an adaptive framework that deals with potential climate change-induced changes to fire regimes; (3) an example of a domain and threshold of potential concern approach that is currently used in the management of fire in a protected area in New South Wales; (4) the role of prescribed burning in mitigating potential risks posed by climate change; (5) the applicability of fire as a disturbance to other disturbance types; and (6) the next steps, and how this area of research and development can be developed.

## 7.1 The implications of complexity and uncertainty

A consistent theme from our analyses of the potential impact of climate change on fire regimes across Australia is that the incidence of fire-weather conducive to fire spread across the landscape is likely to become more frequent. This could be expected to manifest itself in an increased annual sum of FFDI over a year, and/or and increase in the number of days per year where FFDI is >25 (very high and extreme). It would seem likely that this is a potential outcome for the majority of the Australian landscape. The projections for climate change and fuel, and climate change and ignitions are, however, less certain.

Thus, a key uncertainty for fire regimes and their management is whether these key drivers of fire regimes will act synergistically, with a complementary effect on fire regimes, or whether they will act antagonistically, with opposite effects on fire regimes. For example, the 'warming and drying' scenario – the most likely for the whole country – could act in an antagonistic manner, as discussed in section 4.3.3: warming may increase FFDI, and therefore the probability of ignition and spread; whereas long-term drying could reduce fuel loads, thereby mitigating the effects of increased FFDI. The effect of $CO_2$ fertilisation will also act in complex ways with changing patterns of fire-weather.

There is also uncertainty with respect to the impacts of changing fire regimes on biodiversity. While recognising plant species as taxa or as functional types with respect to fires, habitat elements or fuels, there are many interactions and other variables at work. Functional types at present do not capture all the effects of season or fire type (Gill 1975) on response nor take into account possible interactions between components of the regime. Some success is to be had by monitoring species that are most vulnerable to current fire regimes, in conjunction with mapping fire interval and severity components of fire regime, but such monitoring across functional types is very demanding of resources.

The global network of protected areas (and its management) has not been established as an insurance policy against global climate change (Dunlop and Brown 2008). Rather, it has been established as an insurance policy against landuse intensification. Moreover, the concept of biodiversity conservation, especially within protected areas, has evolved from a view of environmental stasis and protection of what is there (implying some form of certainty). It has not, generally, embraced a paradigm of environmental flux (Bond and Archibald 2003). The latter world view recognises the inherent variability in ecosystems, and hence uncertainty with respect to directions and rates of change. At present, there are also changing societal imperatives in biodiversity management, which include reducing the risks posed to biodiversity by global change, and reducing risks to neighbouring landholders posed by protected areas.

Because of the inherent uncertainty and complexity of climate change–fire regime–biodiversity interactions, the preliminary nature of the research that we report on here, and the limited number of regions within Australia for which we have been able to evaluate potential climate change–fire–biodiversity interactions, we are therefore not in a position to offer management prescriptions by which managers may mitigate risks to biodiversity posed by climate change. However, managers are called upon to act and decide in the face of uncertainty, and there is impetus to act to mitigate the risks to fire regimes posed by climate change. For example, the Victorian Parliamentary Committee on Environment and Natural Resources has recommended landscape-scale prescribed burning on public land be increased as a strategy to mitigate fire risk posed by climate change (Environment and Natural Resources Committee 2008). We discuss an adaptive framework, with elements tailored to the needs of acting upon climate change signals, in the next section.

## 7.2 Managing risk in an adaptive framework

### 7.2.1 Adaptive management

Despite considerable uncertainty, climate change poses a risk to Australia's biodiversity, and that risk may be elevated by the impact of climate change on fire regimes. Adaptive management (Holling 1978) is one system of management that is designed explicitly to incorporate uncertainty and risk into its operation.

There is a rich literature on the application of adaptive management in situations where knowledge is incomplete and/or contexts are continually changing, but where risk must be managed (see section 6.5.2 in the National Inquiry on Bushfire Mitigation and Management – Ellis *et al.* 2005). All major inquiries following the 2003 fires (e.g. the Victorian Inquiry – Esplin *et al.* 2003; the COAG Inquiry – Ellis *et al.* 2005) highlight the need for adaptive frameworks. Adaptive management is likely to remain a strategically important component of sustainable resource management in an increasing complex world (Lee 1999).

In practice management systems may be divided into two groups: (i) action based on theory and a calendar; and (ii) action based on what is happening in the landscape, with many possibilities between these two extremes. With respect to fire management, a strong observational program is wise because fire interval is not the only fire regime variable of interest, not all plant species responses are known, animal species may not be accounted for simply by considering plants, and the environment is changing.

The COAG National Bushfire Inquiry provided a concise overview of the scientific issues that need to be addressed in devising effective management for biodiversity conservation (see section 6.5.2 in Ellis *et al.* 2005). This inquiry made the following finding in relation to dealing with incomplete knowledge about setting appropriate fire regimes for biodiversity:

**Finding 6.7**

The Inquiry supports the adoption of an adaptive management approach to setting fire regimes appropriate for biodiversity conservation. Such an approach should:

- explicitly state the biodiversity objectives
- recognise lack of knowledge and clarify questions that need to be answered
- design burning prescriptions such that can answer these questions
- devise and fund monitoring and other data-collection
- review and communicate results
- use the new-found knowledge to modify the management prescription.

This finding highlights the key features of an adaptive management framework are outlined in Figure 7.1.

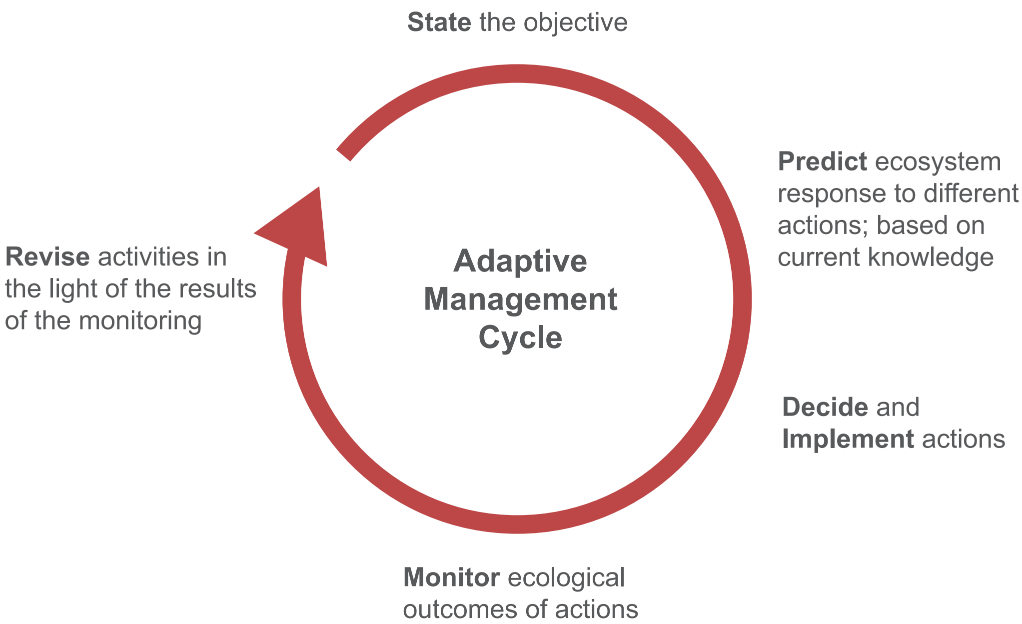



**Figure 7.1** The Adaptive Management Cycle

The need for a robust adaptive management framework is all the more pressing when attempting to manage the response of the biota in landscape subject to changing forces, such as climate change, changing fire regimes, and the complex interactions between landscapes, fire regimes and climate.

This is not to say that adaptive management is not practised at present by fire and biodiversity managers; adaptive management is obviously a key component of fire management in areas managed for biodiversity conservation in Australia (Gill 2008). However, as Burrows (2008) pointed out, 'there is a significant gap between … knowledge and its on-ground application to deliver fire management outcomes'. That is, the key questions are: how prepared is the current adaptive capacity of fire and biodiversity management, and what may need to be modified in the light of new knowledge – in this case, the potential impacts of climate change? It is beyond the scope of this report to review and evaluate the adaptive capacity of fire and biodiversity management in relation to climate change. However, this is an obvious and high-priority area for investment in the future, if the management of fire in Australia's conservation estate is likely to become more complex as a consequence of continental-scale warming and climate change. We discuss some of the issues involved in the following sections.

### 7.2.2 Fire and biodiversity management objectives

The management objective may be to maintain species assemblages as they are. This seems appropriate unless mass die-offs occur, or replenishments of populations fail. Knowing what to monitor as an indicator of ecosystem state is the most difficult part of the process outlined above. One way to do this is to choose indicator species on the basis of their perceived vulnerability to fire regimes (e.g. Gill and Nicholls 1989; see also section 6.2.3).

Assessment of objectives may bring its own challenges, given the near certainty of change. Dunlop and Brown (2008) argued that, with respect to the National Reserve System, managers should prepare for change and manage for minimising species losses by managing to maximise habitat complexity. This is consistent with the 'golden rule' of sustainable natural resource management proposed by Holling and Meffe (1996), that management should aim to retain critical types and ranges of natural variation in resource systems, in order to maintain resilience. Such a change in emphasis could, however, cause a fundamental shift in, or at least reappraisal of, management objectives, because protected areas are often established with the specific objective of conserving taxa, or minimising the risk of extinction of taxa, rather than with the maintenance of habitat variables *per se*.

In the traditional, static context (i.e. without directional climate change), a local species loss or declining population might be attributed to a changing landscape context, altered fire regime or other ecological factor as potential 'drivers' of the change. The management response may be to maintain species assemblages as they are. This seems appropriate unless mass die-offs occur, replenishment of populations fail, etc. In addition, the landscape surrounding protected areas, and other lands managed for biodiversity conservation, needs to be considered, as some species will migrate in response to changing climate.

In the context of climate change and associated changes in fire regimes, the question of additions and losses to biodiversity would need to be treated differently than in a relatively static context. As far as we know, this has not been formally addressed before. Consideration of possible causes of losses or additions of species will need to include migrations of species to or from neighbouring areas. The questions of where species have gone – or could possibly go – and where they might come from may become highly relevant in devising, monitoring, assessing and modifying management responses.

### 7.2.3 Monitoring ecosystem state

Remote sensing has become a key tool for monitoring fuels, fires and fire history (Justice *et al.* 2003; Xiao-rui *et al.* 2005). For biodiversity management, monitoring programs are generally ground-based (although see WS Turner *et al.* 2003). What ever technique is used, knowing what to monitor for is as, if not more, important than knowing how to monitor.

For fire and biodiversity management, one approach is to choose indicator species on the basis of their perceived vulnerability to fire regimes (e.g. Gill and Nicholls 1989). Vulnerability of plant species may be discerned from their functional groups including immediate response to fire (resprout or not) and time of regeneration (inter-fire, post-fire) (e.g. Noble and Slatyer 1980), and from life history markers (e.g. time to seed production from seedlings – primary juvenile period; or resprouts – secondary juvenile period; Gill 1975) and longevity.

Management systems in a number of Australian states and territories are rapidly developing; they use functional groups of plants as a guide to management, together with estimates of juvenile period and longevity to set a desired mean fire interval (e.g. Wouters *et al.* 2002). Thresholds can be set for desired minimum and maximum intervals, and some form of variation in these intervals established on the basis of some quantitative measure (e.g. statistical, expert opinion). In general, the knowledge of thresholds of concern in relation to fire regimes and fauna is poor, compared to that of fire regime thresholds and plants (Bradstock *et al.* 2005; Clarke 2008).

Important ingredients for climate change include:

- monitoring of long-term trends in fire-weather and fuels (see further on)
- developing comprehensive fire mapping programs to track fire activity across the landscape

- identifying the sensitivity of critical elements of regional biodiversity to variation in fire regime (see further on)
- evaluating the sensitivity of risk to climate change and management options
- evaluating the benefits and costs of all risk mitigation decisions.

In this process, identifying thresholds of potential concern will be as important as prescriptions and targets (Keith *et al.* 2002b).

Monitoring of ecosystem state must be undertaken in the context of vulnerability of components of ecosystems (species, functional groups, communities; both plant and animal) to variation in components of fire regime. While recognising plant species as taxa – or as functional types with respect to fires, habitat elements or fuels – there are many interactions and other variables at work. However, some success can be achieved by choosing the most vulnerable species to present fire regimes and monitoring it (or them, across localities) along with maps of fire occurrence and possibly intensity/severity. Where intensity/severity is important to species response, knowledge of functional type of the species, alone, is inadequate for predicting the future course of the species; what individual fires do to the populations present is significant. Functional types at present do not capture all the effects of season or fire type (Gill 1975) on response, nor take into account possible interactions between components of the regime.

Monitoring of species thought to be most vulnerable requires knowledge of functional types, at least; systematic recording of these in relation to fire is a feature of the management programs of most Australian states. Whatever the uncertainties associated with climate change and fire regimes, directing resources towards monitoring such taxa is a 'no regrets' policy. For example, given the finding from the south-west Western Australian case study (section 6.3) that serotinous seeders may be more vulnerable to the interactive effects of climate change and fire regime change than seeders with soilborne seeds, a general management objective should therefore be to determine the relative proportions of these functional types in all areas where such seeders are a significant proportion of the flora. Because of uncertainty, monitoring 'wildcards' (species with unknown responses or responses at variance to chosen indicators) is also recommended as a subsidiary activity to the main program. Watching the effects of fire regimes on fungi and potoroos, or beetles (Doran *et al.* 2003), for example, could be revealing.

An integrated framework for assessing vulnerability of species to climate change has been proposed recently by S.E. Williams *et al.* (2008). A formal program of monitoring along an altitudinal gradient has been established in Tasmania (Doran *et al.* 2003). High mountain systems of monitoring for climate change impacts on biodiversity have been developed by the GLORIA research consortium (Pauli *et al.* 2004). Such a program could be extended by its adoption in other regions and for the same or other taxa. It could also be run in association with a simple experimental program, so that at each monitoring station experimental treatments (e.g. burning or grazing) could be implemented.

### 7.2.4 Monitoring drivers of change: using predictions to guide monitoring

Our analyses of the potential impacts of climate change on fire regimes and biodiversity provide four key predictions that can be tested within an adaptive monitoring framework. These are: (i) fire-weather will become more severe; (ii) fuel mass and type will change due to changes in moisture and $CO_2$; (iii) ignitions will change as a function of changing climate and patterns of human settlement; and (iv) the distribution of fire regime components (e.g. interval or severity classes) across the landscapes (upon which inferences about certain elements of biodiversity can be made) will change as a consequence of changing fire regimes. All four can be monitored using standard, modern techniques or data sources, either from meteorological records, field survey and records, or using remote sensing and geographic information systems. Commencing to monitor these variables now will provide baseline information against which to judge change.

Trend analyses of regional fire-weather using cumulative FFDI or the number of VHE days could be undertaken, based on local meteorological records and using the approach outlined in section 4.2.2. Trends could be updated and evaluated over the coming decades. Lucas *et al.* (2007) proposed that the trend towards increased FFDI over the past decades has a climate change component to it, so more evaluation to test this hypothesis, as part of a formal monitoring program, is a practical indicator of likely trends in regional fire activity.

Trends in fuels can be monitored using standard fuel monitoring techniques. Climate change – temperature, rainfall and increasing $CO_2$ – may alter fuel loads, but the magnitude and direction of change in likely to be uncertain for any given region. To manage fire regimes for biodiversity or any other purpose, knowing the state of fuels across the landscape is, of course, important. Targeted monitoring of fuel loads will reduce uncertainty, and will allow formal testing of the hypotheses that, for example, drying may lead to less fuel, or indeed may have already led to declining fuel loads. A prospective region to undertake such trend analyses is south-west Western Australia, where there has already been significant drying over the past 30 years.

Space-for-time studies along moisture gradients can be used to evaluate potential impacts of climate change on fuels. The approach illustrated for south-west Western Australia (section 4.3.3), where the potential effects of a 20% reduction in moisture on fuel loads and rates of accumulation were evaluated, is a simple technique that can be applied elsewhere in Australia. It provides explicit fuel variables that can be measured. This approach could be applied at other locations in southern Australia to seek to identify whether and at what rate fuels may respond to changes in moisture.

Changes to lightning incidence can be assessed from meteorological records, field records and archives of remotely-sensed imagery.

With respect to monitoring biodiversity, knowledge of the current distribution of interval and severity classes across protected areas, as indicators of habitat suitability, are increasingly being used in biodiversity conservation management (Bradstock and Kenny 2003; Bradstock *et al.* 2005; Clarke 2008). There is a need to know what domains give acceptably low risks of species extinction and how sensitive that domain is to climate change and changed fire regimes. We illustrate this below, using as an example a scheme based on data from a conservation reserve (Brisbane Water National Park – 12,000 ha) in the Sydney Basin of NSW (Bradstock and Kenny 2003).

## 7.3 Monitoring domains and thresholds of concern for biodiversity conservation: an applied example

The approach of Bradstock and Kenny (2003), as discussed in section 6.5.6, is based on the concept of 'thresholds of potential concern' (Bond and Archibald 2003) in relation to critical life history attributes of key species; in this particular case, sclerophyllous woodlands and forests with a significant proportion of obligate seeder shrub species that are sensitive to fire interval to either frequent fire or infrequent fire. Using critical life history markers (primary juvenile period, longevity of adults) a domain of 'acceptable' fire intervals was derived – in this case, between seven and 30 years. The fire history of the conservation reserve was then analysed, and the distribution of intervals determined. This, in turn, allowed determination of the proportion of the landscape with intervals between the 7-30 year thresholds of concern. The conclusion from associated spatial modelling was that a significant decline in species within the reserve may be expected if more than half the fire intervals in a landscape lie outside the acceptable fire interval domain. This sets a measurable fire interval objective for conservation managers of the vegetation concerned but, as with the application of all such models, continual assessment in the field is recommended.

The framework outlined by Bradstock and Kenny (2003) is a potential generic framework that has flexibility and predictive power, within which the key components can be monitored. It should be possible to modify this approach to include species with populations that are vulnerable to particular fire intensities – such as alpine ash, as discussed in case study 1.

We believe that exploration of the approach for other ecosystems would be a very fruitful area of fire ecology research and development. This would involve determining the appropriate interval variation around a mean constrained within upper and lower bounds for the persistence of species. The interval-TPC concept would also need to be expanded to determining the vulnerability of species to variations in other fire regime components, particularly intensity variation, season and type of fire (e.g. peat versus above-ground). The formulation of domains and thresholds of concern for fauna is poorly developed compared with plants

(Bradstock *et al.* 2005; Clarke 2008) and requires further research. We see this as a high priority for future research in landscape fire management.

The approach also has implications for reserve size – the approach predicts that it is easier to maintain the appropriate proportion of landscapes within the critical domains in larger reserves than in a smaller ones. Thus, for the Sydney Basin, an analysis of the susceptibility of Royal National Park (a relatively small park) versus, for example, Morton National Park (a large one) would be instructive.

The Bradstock and Kenny example implies that local extinction of canopy-type seeders, at least, will occur as a critical percentage (i.e. the threshold; in that case, 50%) of the landscape has an interval too short for the persistence of the species. If the regime is randomly allocated then this process of local extinction will continue until all the landscape is affected. Dispersal of the species could mitigate this effect.

An alternative model comes from Gill (2008) in reference to heathland dominated by the seeder *Banksia ornata*. The minimum interval in that community can be decreed as being equal to the juvenile period or, preferably, the time to first seed maturity. The mean interval can be set at two times the juvenile period while the upper end of the domain for species persistence can be set by the longevity of the species, which in this case is believed to be about 50 years. In this model, no extinction occurs and variation is set by a statistical frequency distribution. The method allows for species persistence while maintaining fuel levels at their minimum. The simple statistical function used in this example may be seen by managers as giving too great a chance of fires burning near the minimum limit, so the minimum may be artificially raised, or a more complex function like the Olson function used (McCarthy *et al.* 2001).

## 7.4 Prescribed burning, climate change, fire regimes and biodiversity

Prescribed burning is an important tool in fire management in Australia, in both conservation and non-conservation lands, and will continue to be an important component of the management of fire regimes and biodiversity within protected areas. Following the 2003 fires in southern Australia, both the COAG and Esplin reports (Esplin *et al.* 2003; Ellis *et al.* 2005) recommended an increase in the level of prescribed burning in landscapes as part of a general approach to improved fire management. One recent report on fire management on public land in Victoria (Environment and Resources Committee 2008) has recommended (Finding 7.2; p. 244) that the level of prescribed burning in the Victorian landscape be increased significantly, partly as a way to mitigate the risk climate change poses to the fire regimes; the recommendation was for a tripling of the extent of prescribed burning, from about 130,000 ha to 385,000 ha. Thus, there is impetus to increase the level of prescribed burning in the landscape as a means of mitigating additional risks posed by climate change.

A key issue is the effectiveness of prescribed fire in achieving stated objectives in a risk management framework. This includes mitigating the extent, intensity and impacts of unplanned fire, and achieving appropriate spatial and temporal distribution of regime components to minimise risk to biodiversity. Quantitative assessments of the efficacy of prescribed burning in achieving such multiple goals are, however, rare.

In northern Australia, ongoing active management of landscapes through the use of prescribed burning is, and will remain, a key component of most natural resource management (NRM) objectives, because remoteness is of itself no barrier to the evolution of inappropriate fire regimes (Woinarski *et al.* 2007). There are three main, interrelated landscape issues that underpin the application of prescribed burning for biodiversity conservation in savanna landscapes – the use of early dry-season prescribed fires to: (i) mitigate the impact of more intense and extensive late dry-season fires (Dyer *et al.* 2001; Williams *et al.* 2009) thereby; (ii) decreasing fire frequency (Andersen *et al.* 2005); and (iii) maintaining landscape heterogeneity (Russell-Smith *et al.* 2003a; Whitehead *et al.* 2005; Woinarski *et al.* 2007).

In northern Australia, quantifying the influence of fuel and season of ignition on fire behaviour has been a feature of medium- to long-term fire experiments such as those at Annaburroo, Kapalga and Munmarlary (Williams *et al.* 2003b). In a study of the effectiveness of prescribed burring in the early dry season to mitigate the extent of late dry season fires, Price *et al.* (2007) analysed the effect of firebreaks created by aerial prescribed burning in mitigating unplanned fires in western Arnhem Land. They found that such prescribed burning achieved its aim in parts of the landscape, but cautioned that such firebreaks do not guarantee fire mitigation because of, for example, gaps through which late dry-season fires can spread.

In southern Australia, there has also been considerable research effort directed at determining the impact of prescribed burning on fire behaviour, and at the production of guidelines for the deployment of prescribed fire. Generally, these studies have been at the experimental plot scale (hectares). McCaw *et al.* (2008b; their Table 1) presented data from Project Vesta in Western Australia (Gould *et al.* 2007; McCaw *et al.* 2008a) on the effect of fuel age on the behaviour of experimental fires. In three- to four-year-old fuels, rates of spread and headfire intensity were reduced by factors of 3–30 compared with fires burning under comparable metrological conditions in six- to 22-year-old fuels. However, these (and other similar) plot-scale experiments were conducted under moderate to high fire-weather conditions (i.e FFDI <25) as opposed very high to extreme fire-weather conditions (FFDI >25).

Experimental fire in 1999 in eucalypt forest as part of the Project Vesta investigations into fire behaviour; south-west Western Australia
Source: Geoff Cary

At the landscape scale in southern, temperate ecosystems of Australia, there have been few published, quantitative studies of the effectiveness of prescribed burning in mitigating the impacts of unplanned fire at landscape scales of hundreds to thousands of square kilometres (Burrows 2008). This is not surprising, given that such analyses would require evaluation of multiple factors that affect fire behaviour in generally complex landscapes. Moreover, at landscape scale, any fuel reduction treatment is usually partial in extent. A recent example from the USA, using satellite imagery and prescribed fire records, is that of Finney *et al.* (2005). They explored the impact of fuel treatments on the behaviour of the Rodeo-Chediskski wildfires in south-western USA in 2002, and concluded that fire severity was reduced in fuel-treated areas, but that the effect diminished over time. Within Australia, King *et al.* (2006) explored the issue using simulation modelling for landscapes in south-western Tasmania. They concluded that treatment of 5–10% of the landscape could reduce the extent of unplanned fire, although higher levels were needed to fulfil the potential for maximum reduction of unplanned fire size. The effect of prescribed fire on area burned by unplanned fire also depended on complex interactions between treatment unit size, treatment level and spatial configuration (King *et al.* 2008).

In their recent global review of the effectiveness of prescribed burning and its role in mitigating unplanned fire, Fernandes and Bothello (2003, p117) concluded that 'The best results of prescribed fire application are likely to be attained in heterogeneous landscapes and in climates where the likelihood of extreme weather conditions is low. Conclusive statements concerning the hazard-reduction potential of prescribed fire are not easily generalised, and will ultimately depend on the overall efficiency of the entire fire management process.' Cary *et al.* (2009), in a multi-model, multi-continent comparison of the determinants of area burned, and in a range of landscapes across the world, found that the extent of young fuels (i.e. those that result from prescribed burning) was less important in determining area burned than were both inter-annual variation in weather and variation in ignition management effort.

A critical national, fire–biodiversity conservation issue is the degree to which prescribed burning affects the landscape-scale distribution of intervals between fire (the 'invisible mosaic' – Bradstock *et al.* 2005; Parr and Andersen 2006). In northern Australia, the chief aim of prescribed burning is to mitigate area burnt by unplanned fire, so that intervals between fires are lengthened, for select parts of the landscape (Andersen *et al.* 2005). For southern Australia, the substantial issue is minimising the increase in short inter-fire intervals that will ensue due to prescribed burning interacting with unplanned fire, and the consequent risk that may be posed to certain plant and animal taxa (Bradstock *et al.* 2005; Gill 2008). Consequently, the outcomes of prescribed burning are divergent in these contrasting ecosystems, as a result of the fundamentally different biota and fire regimes that currently prevail.

In both northern and southern Australia, prescribed burning will also lower the intensity of subsequent unplanned fires that occur with treated areas (Underwood *et al.* 1985; Dyer *et al.* 2001; Gould *et al.* 2007; Burrows 2008; McCaw *et al.* 2008; Sneeuwjagt 2008; Williams *et al.* 2009). Benefits to taxa such as higher vertebrates may ensue. Thus effects of prescribed burning on biodiversity involve trade-offs between resultant fire intensity and frequency (i.e. length of inter-fire interval). These trade-offs need to be 'tailor-made' to suit the biotic and fire regime context of individual ecosystems. Manifestly, differing prescribed burning solutions are required in different parts of Australia to achieve this. Much more needs to be learned in this regard.

On the basis of the preliminary analyses summarised in this report, and the wider literature, we conclude that risk reduction is affected by a variety of factors. Of course, one factor is the level of prescribed burning in the landscape. However, as indicated by the results of modelling for the Sydney Basin case study, and detailed analysis of fire history in these and other ecosystems (RA Bradstock *et al.* unpublished data), relatively large amounts of prescribed burning have to be implemented in Australian forested landscapes to achieve modest levels of risk mitigation for urban and other assets (see section 6). Therefore, risk is unlikely to be totally eliminated by any feasible level of prescribed burning, i.e. there is always residual risk (Gill 2005).

The relative benefits and costs of prescribed burning, and its effectiveness in achieving fire management goals – whether climate change-related or otherwise – require further research and evaluation. This is particularly important if proposals to double or triple the amount of prescribed burning are to be feasibly implemented and sustained in the long term.

Prescribed burning will continue to be a fundamental component of fire management in most public and private lands in Australia – whether lands managed for biodiversity conservation or other land uses – in tropical, arid and temperate regions. Climate change is likely to affect prescribed burning operations, via effects on the number and seasonal distribution of days that are suitable for prescribed burning. The current prescriptions and guidelines may change under climate change scenarios, and the impact of various climate change scenarios on the distribution of days that are suitable for prescribed burning requires further research.

The science underpinning the application of prescribed burning to the achievement of multiple objectives and minimising risk to multiple values at landscape scales is complex, and is still in its infancy. The development of optimal strategies for prescribed burning in relation to climate change, fire regime and biodiversity conservation will be contingent on interactions with other facets of fire management, a better quantitative understanding of the effectiveness of prescribed burning in relation to stated objectives, and an economic evaluation of the costs and benefits of varied combinations of these facets. Current levels of understanding in this regard are not well developed, and require much more research. Potential changes to fire regimes as a consequence of climate change provide an excellent opportunity to develop the multi-disciplinary teams that are required to further this cross-disciplinary research.

**Box 7.1 Climate change and 'mega-fires'**

'Mega-fire' (sometimes 'megafire') is a term gaining some currency in the fire ecology and management literature. It has been used in relation to recent large fires in both the USA and Australia, and potential impacts of climate change on fire size.

The term 'mega-fire' was coined by Jerry Williams, and the Mega-fire Scoping Group was convened in 2003. In a 2004 Issue of 'Fire Management Today' Williams used the term to describe one of four kinds of fire: the small initial-attack fire, the transition or extended-attack fire, the large fire, and the mega-fire. He also applied the term to two large, high profile fires that occurred in 2002 in the western USA – the Biscuit Fire in Oregon and the Rodeo–Chediski Fire in Arizona. In a subsequent paper from the Brookings Institution in Washington DC (September 2005; 'The Megafire Phenomenon: Toward a More Effective Management Model. A Concept Paper' http://www.wildlandfire.com/docs/2007/megafire-concept05.doc; p. 4), Williams described, rather than defined, 'megafires' as being 'a situation as much as they are an incident'. He comments: "Emotions run high when they occur. They are not defined in absolute terms, using physical measures (e.g. acres burned). Instead these are 'headline' wildfires where operational limitations, public anxieties, media scrutiny, and political pressures collide." Further – "While generally very large in size, complexity is their best descriptor. They overwhelm local capabilities and capacity". Williams was also a keynote contributor to a recent publication from the Bushfire Cooperative Research Centre, 'Are Big Fires Inevitable?' (http://www.bushfirecrc.com/events/events/forum_feb07.html), which examined the issue of fire size and the concept of the mega-fire from the perspective of land management and climate change.

Thus, we can say that mega-fires:

**(i)** are not always defined as million hectare/acre fires;

**(ii)** demand attention by society in general and engage the emotions of the public at large;

**(iii)** overwhelm suppression capacity.

The term has been applied the recent fires in south-eastern Australia and the western United States. The term is also being used to describe a potential outcome for climate change-induced alteration of fire regimes; a common theme is that climate change will lead to more/bigger/hotter mega-fires. Recent examples of the use of the term include works on Australian perspectives on the phenomenon (Bartlett *et al.* 2007), bushfire losses in the built environment (McAneney *et al.* 2007), community bushfire safety (Handmer and Haynes 2008; Downing *et al.* 2008); and fire-atmosphere interactions (Guerova and Jones 2003).

While the term 'mega-fire' is a relatively new, it is actually describing an old phenomenon – that of large fires. Large fires occur periodically in all fire-prone parts of the world (Williams and Bradstock 2008), with the recurrence rate dependent on a number of factors. In southern Australia, the recurrence rate is multi-decadal (Bradstock 2008) whereas in northern Australia large fires can occur in any year (Yates *et al.* 2008).

During the course of this study, we were not able to undertake any numerical/quantitative/scenario modelling of the impacts of climate change on large fires. However, on the basis of the published literature on Australian fire regimes, we can say that, spatially, large fires are a part of Australian fire regimes. Moreover, and just as importantly, large fires are a part of the historical range of variability for virtually all biomes in Australia (Bradstock *et al.* 2002; Bradstock 2008; see also Section 5). That is, large fires have occurred across much of Australia from time to time in the past.

We would argue that, with respect to climate change, fire regimes and biodiversity, it is not the occurrence of individual large fires per se that poses an additional risk to biodiversity assets, but the recurrence rate, as we have argued earlier in Section 7. Managing for recurrence rate and the associated inter-fire interval were stressed as vital components for the management of biodiversity in both tropical and temperate ecosystems (the case studies; Section 6; section 7.3).

Some immediate research and management questions then flow from this consideration of mega-fires. One would be to examine the biodiversity consequences in those areas in Victoria and NSW burnt by both the 2003 and 2006-07 fires. More broadly, there is a critical question concerning the ways in which planned and unplanned fire (including large unplanned fires) will interact to determine inter-fire interval under climate change scenarios. These are the same issues addressed above in the Section on prescribed burning, and, while challenging, represent an exciting opportunity for fire-biodiversity research in the coming decades.

Regenerating dry sclerophyll forest on steep slopes, burnt by 2003 fires, Mitta Mitta Valley, Victoria
Source: David Dunkerley, Monash University

## 7.5 What can the analysis of climate change and fire regimes tell us about other disturbances?

One major question is ‘How applicable is the framework we have presented – for examining climate change impacts on fire regimes – to other disturbances?’ What can this regime analysis tell us about other disturbances? Fire regimes interact with other disturbances as well as climate, and the concept of the regime applies to all event-based phenomena that affect ecosystem states in one way or another. The basic answer is that any event-based phenomenon can be placed into a regime framework. Four examples for fire, precipitation, grazing and cyclones are outlined in Table 7.1. The effects on the system in the longer term depend on the regime as, by definition, the system state is affected by each event.

**Table 7.1. Examples of event-based phenomena that can be understood as regimes.** Application of the regime concept in hydrology (and stream biodiversity) may invoke other variables like 'fire-affected area' or 'amount of precipitation'; in fire regimes, the importance of intensity may be downplayed in grassland examples but be of major importance in forests.

| | Fire | Precipitation | Grazing | Cyclones |
|---|---|---|---|---|
| **Type** | Above/below ground | Snow, rain, hail, fog drip, drought | Kangaroo, phasmatid, deer, domestic stock | Storms; cyclones |
| Intensity | kW/m edge | mm/hr<br>1-in x years, etc. | Animal mass/unit area<br>utilisation rates | Category 1–5; central pressure, wind |
| Season | month or season | season | season | Season |
| interval | years | days | days | Years to centuries |

Events as phenomena are discrete (digital) and may be contrasted with pervasive circumstances, which are continuous (analogue). The latter include, for example, soil moisture, plant growth and litter decomposition. Some continuous phenomena are broken into discrete units and can then be considered to be events, e.g. drought.

Any event-based disturbance – such as grazing, floods, frost and cyclones – can be scrutinised in terms of regime, and key regimes components (intensity, interval, etc.) quantified. The impact of variation in regime component on functional types and traits can also be analysed according to the logic and methods employed in this report. Examples include Lunt *et al.* (2007) for grazing, and Cook and Goyens (2008) for cyclones.

A key feature of fire and other disturbances is that they interact. Fire x grazing, and fire x drought are common examples. Fensham *et al.* (2003) highlighted the significance of rare events (drought; improbably long fire-free period) for influencing savanna species composition and structure. Cook and Goyens (2008) emphasised the interacting effects of fire and cyclones on the long-term carbon balance of savannas.

Extensive damage to trees in tropical savanna in Arnhem Land, NT, following Cyclone Monica (Category 5), April 2006. Approximately 700 000 ha of savanna was severly impacted by this cyclone.

Source: Gary Cook

Prescribed burning in Mornington National Park, Kimberley region, Western Australia.
Source: Nick Rains

## 7.6 Exploring climate change, fire and biodiversity through research and management partnerships: the next steps

The review of our approach to the problem (sections 2 and 3) and our technical analyses (sections 4 to 6) constitute the major component of this report. We see these analyses as being Phase 1 of an ongoing project. We have devoted less effort to the management implications of our analyses. The next phase of the research into climate change, fire and biodiversity should therefore concentrate on exploring the key management questions: the objectives of fire management for biodiversity conservation; the adequacy of the current management frameworks to deal with the uncertainty of climate change; identification of vulnerable biodiversity assets in the landscape and how altered fire regimes may impact upon them; and the role of fire management, prescribed burning in particular, in achieving conservation aims and mitigating the effects of unwanted fire regime changes.

### 7.6.1 Further research needs

There is a spectrum of future research needs and questions. These include more detailed analyses of the impacts of climate change projections on fire-weather and fuels, specific modelling of climate change–fire regime–biodiversity interactions, and evaluating the potential for rethinking fire and biodiversity management strategies. Our analyses have focused on: (i) predicting particular effects of climate change on fire regimes and biodiversity, and the associated uncertainties; and (ii) proposing the elements of plausible and potentially effective management pathways to manage fire for biodiversity conservation, in the face of changing climate and competing demands. Research to address these issues is necessarily multi-disciplinary and collaborative, and will involve research and management agencies. Australia is well placed internationally to develop such research partnerships.

Management responses are being developed in response to the current uncertainty in this area; this is a critical area in need of further research, especially at a landscape scale. However, we need to refine our understanding of the alterations to fire regimes that might be expected from changes in climate, at a scale that is relevant to land management.

A key conclusion from our research is that there are as yet no prescriptive, generic 'solutions' to the problem of mitigating risk to multiple values and assets – including biodiversity – posed by climate change-induced changes to fire regimes. However, in general, the lessons learned from the management of fire for biodiversity conservation over the recent decades, and further consideration of the nature of change to fire regimes that may accompany climate change, will provide a robust basis upon which to plan future management strategies. A key requirement will be to develop and enhance monitoring strategies that track the impact of climate change on biodiversity, the evolution of altered fire regimes, and biodiversity responses to interacting drivers of change at landscape scales.

Key features of future research and management programs that allow risks to biodiversity posed by altered climate and altered fire regimes to be evaluated, and minimised by appropriate management, include:

- monitoring the spatial and temporal components of fire regimes across the landscape. This includes measures of habitat complexity. Concomitant with this, there is the need to further develop a strong observational base
- determining the impact of increased atmospheric $CO_2$ concentration on rates of primary production, and the rate at which fuel levels recover after fire
- identifying key taxa and communities that are sensitive to different components of fire regime (in particular, to fire intensity and interval)
- determining the impact of variation in fire regimes on fauna
- evaluating thresholds and domains of concern for components of fire regimes and biodiversity; management thresholds for fire interval are currently better developed than are thresholds for fire intensity or fire season
- evaluating the risk posed by different scenarios of change to biodiversity values
- identifying the trade-offs that will be necessary to minimise risk to multiple values
- developing and refining the means to evaluate the effectiveness of fire management actions, including benefit-cost analyses of management options.

### 7.6.2 Seven priority areas for action

The outstanding research and development issues, and the suggestions for future research outlined at the end of various sections, translate into seven priority actions. One relates to national fire regimes, three to ecosystem dynamics research, and three to ecosystem management. All could commence immediately and, most importantly, all are 'no regrets' actions, in that further research and development in these areas of fire management is justified whether or not climate change poses an additional set of risks to biodiversity values that have to be managed.

#### *Ecosystem dynamics*

1. **Determination of Australia's fire regimes.** While there are nationwide studies on fire occurrences and fire intervals for short periods – and detailed studies on fire intervals for a few small study sites – there are no sites where intervals, intensities and seasons, let alone types of fires, are known. Without baseline data, any knowledge of the effects of climate change on fire regimes is necessarily limited.
2. **Determining the potential impact of climate change on fire-weather in other regions of Australia.** This is as per the analyses undertaken for south-eastern Australia (Hennessy *et al.* 2005), as discussed at the end of section 4.2.3. Such analyses will indicate the potential effects of climate change on key weather variables that affect fire regimes, such as the incidence of Very High and Extreme fire danger days, and the annual sum of FFDI.
3. **Evaluation of the relative importance of elevated fire danger, elevated atmospheric $CO_2$, and changing moisture availability as determinants of future fire regimes.** This requires much more research and analysis, using regional climate change scenarios, regional fuel change scenarios (including the invasion of exotic grasses), and spatially explicit fire and biodiversity models. A key area of uncertainty is the effects of elevated $CO_2$ on both vegetation productivity (and hence fuels) and the capacity of vegetation to recover after fire. Elevated $CO_2$, and its interactions with moisture and temperature regimes, has the potential to change fire regimes, but we do not know in what ways.

4. **Expanding our understanding of the effects of variation in fire regime components on fauna.**
   The assumptions regarding fire regime and plants do not correspond to fauna (Clarke 2008), and require further research and validation in the field. Even within individual reserves, the assumptions regarding individual faunal groups do not necessarily hold for other groups, and there are no obvious surrogates or shortcuts for faunal biodiversity monitoring in relation to variation in fire regimes (Lindenmayer *et al*. 2008). The exploration of particular species of fauna as 'focal' or 'umbrella' species for devising fire management strategies offers potential (Burrows 2008).

*Ecosystem management*

5. **Review and assess current adaptive management capacity to accommodate change.** A key finding of Dunlop and Brown (2008) is that there will need to be a re-evaluation and possible recasting of the objectives for conservation management under climate change scenarios. The climate change and fire regime scenarios we have outlined obviously raise issue of how to manage fire. We have deliberately avoided providing prescriptive actions, as we believe that, given the uncertainties, the potential options require concerted, coordinated discussion between researchers and managers at a national and/or state fora. We see this as the priority action for Phase II of the process that has commenced with this report.
6. **Explore approaches to domain and thresholds of concern.** This could begin immediately, in a range of conservation reserves. In addition to thresholds for interval, thresholds for intensity, fire season and fire type need to be explored. This probably needs to be done on a park-by-park basis, but with the added complexity that the condition of the regional landscape surrounding a conservation area, and its capacity to accommodate potential species range shifts, all need to be taken into account.
7. **Undertake benefit-cost analyses of potential management responses.** This is particularly acute for the issue of prescribed burning, because increased levels of prescribed burning are being proposed currently as a tool to mitigate the effects of climate change on fire regimes. Prescribed burning is very resource-demanding, yet the level of prescribed needed to counteract potential effects of climate change on fire regimes is unknown. Moreover, socioeconomic as well as biophysical variables influence wildfire risk (Mercer and Prestemon 2005). Evaluation of resource requirements and commitment, in relation to measures of effectiveness of prescribed burning in reducing area burnt under future climate scenarios, requires a major research effort.

# 8. RESOURCES

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# 9. APPENDICES

## 9.1 Appendix A. Workshop report

### Department of Climate Change Workshop:

*The effects of Climate Change on fire regimes in areas managed for Biodiversity*

**University House, ANU, Canberra: 26-28 March 2008**

Wednesday, 26th March 2008 Location: The Hall

**Welcome – DCC** – Dr Anne-Marie Wilson (PowerPoint available)

Key points:

- Adapting to unavoidable climate change – 1 of 3 climate change (cc) pillars
- Australia particularly vulnerable to cc
- Key elements of Australia's strategy – building understanding and reducing sectoral vulnerability – aim to mainstream into policies and programmes.
- Project is one of a number commissioned by the NRMMC, including the national biodiversity (bd) vulnerability assessment.
- Can this project give generic guidance about other disturbance regimes?

**Welcome – CSIRO** – *Dr Dick Williams* – Aims of workshop

- To wrestle with problem – contestability
- Evaluate the work of the consortium
- Discuss research ways forward and comms and data needs
- Start thinking about the final report

### Collaborator and participant introductions

See comment notes under participant names at end

**Project overview** – *Dr Dick Williams (PowerPoint available)*

Key points:

- Already have very powerful current framework for understanding fire and biodiversity interactions
- Principle that regimes rule remains important
- Different biodiversity components will be sensitive to different components of regime
- Interactions between climate change and fire regime will vary eg sensitivity to interval/intensity
- Using case studies to explore issues – impact will vary north to south
- Is biodiversity more sensitive to climate change regime change or fire management change?
- Persistence in the landscape – factors affecting that persistence – longevity, migration, genetic adaptation
- Importance of recognising complexity in the task *'it's not rocket science – it's much harder than that'*
- Drivers and interactions – tools – needs. $CO_2$ fert, temp & moisture, fire weather, 2 7 3-way interactions
- Interactions with other disturbances such as weeds and cyclones and land use changes and changes to their interaction as a result of cc. Landscape heterogeneity
- Climate change is an operational reality. Operational principles – key to dealing with uncertainty is adaptability; technical innovations, new stock, new climate envelopes; trick is to know the asset portfolio – identify benefits, identify and manage risks and vulnerabilities. (lessons from the agric sector)

## Current climate change scenarios –

*Setting the fire–climate change–biodiversity scene*

**Presentation: Current climate change and fire scenarios** *(Dr Chris Lucas) (PowerPoint available)*

Key points:

- Effects of climate change on fire danger – modelling with Kevin Hennessy – used two climate models CCAM2 and CCAM3 – apply changes to observed – recompute FFDI for SE aust only (Sept 2007 Climate Institute report)
- By 2020 ~ 10–20% increase in very high to extreme days (VHE)
- Cumulative FFDI results – mostly large increases internal NSW Qld (40%) – to minor changes in Tas and half on coast – roughly 50 – 300% increase in VHE days by 2050 – worse in interior. Earlier start to fire season (e.g. spring rather than summer in Vic) and slightly longer end. Longer fire seasons – less time for prescribed burning – bigger chance for escaped burn.
- **Conclusion/hypothesis** – current fire behaviour in FFDI is combination of inter-decadal variability and climate change – but how much of each factor? CLIMATE CHANGE models may be too conservative, or results may return to projections if decadal variability cycle continues. Time will tell.
- **Discussion** – (Bowman) Which part of record should we use for pre climate change conditions? Serious implications for fire ecology. A – Not clear.
- (Roxburgh) First extreme day data – shift into late spring of fire season in Melbourne – is that due to increased Summer r/f? A – more likely to be lower winter r/f.
- (McCaw) How climate projections deal with humidity? Models don't predict strong humidity changes – has effect but pretty small. Most change explained by temp and precip – wind and RH have minor impact.

***Presentation:*** **Fire regimes and Biodiversity conservation in a climate of change: Alpine ash in south-eastern Australia.** *(Dr Malcolm Gill) (PowerPoint available)*

Key points:

- Resilient static forests? – lots of factors affecting their persistence.
- Ash population survival – boundaries between fire persistence and being killed at different times of life cycle if interval or intensity wrong – fires in relation to life history markers – after first flowers and seed produced, and before seed declined last seed and death. Species persistence if intensity too high and between fire interval to short. *Domain for survival* of spp. – no intensity too high or too low and between fire interval not too long.
- Spp on the move – does alpine ash move? Projected to move up the altitude and down in latitude. Realised niche – compare with fundamental niche if not limited by competition. Realised niches – will they expand or contract? Depends on what happens to other spp. Issue of barriers and bridges to relocation.
- Uncertainties in the forecast – changes to fire regimes (fuel accumulation, fire weather change, effects ignition rate, effects of improved suppression methods), naturalisation of other spp., etc. (missed a couple)

Our attitude – Be alert and alarmed

Be aware and be beware?

Discussion –
Persistent seedbank? – Only in canopy. Roulette as to whether fire comes at right time.
Long distance dispersion – opportunity to disperse in short time?
Regeneration responses?
Highlighted complexity with this example. How would re-sprouter add to complexity? General decline with time since fire. How many cycles go through between big fires may govern change in communities.
Consider time lag before veg responds to change in regime.

## Plenary: State and territory representatives' presentations: Biodiversity conservation in a changing world

**Setting the fire**–climate change–biodiversity management concerns state by state: concerns for the future and expectations of the project. Brief presentations from each state representative.

**NT – Steve Sutton** – Few conservation areas (but large aboriginal areas). Fire season middle of year. Number of fire ban declarations increased heaps in past couple of years. However trend in area of area burnt declining in early burns in particular. Trend towards subdivision of pastoral land; however still much larger blocks than in Nth Qld. Trend towards more intensive land use. Consistent popln increase – mostly remote and rural increase. Increased stocking rates on pastoral properties – increased nutrition – plus increased investment in mango and horticulture. Increased infrastructure – challenges to manage all these plus biodiversity – new element for NT. Need to make it palatable to the people of the region esp indigenous people. Lack of interest of the people in biodiversity management as a relative priority. Need to include in framework of other pressures that exist.

**SA – Mike Williams** – Issues important in SA – trying to manage 22% of state in fragmented reserves. Linkages and roles in fire. Land for biodiversity conservation – a lot in private ownership – implications? Fire interval issue in Mallee and coastal landscapes? Becoming more frequent. Do they intervene? What does it mean? Increased efforts and costs to suppress – more money being put in the back end than the front end of the equation. How to make good decisions on the ground. Tools to make choices between options with limited amount of money.

**Vic – Gordon Friend** – One of the most fire prone areas of Aust – only 3% area but 50% impacts. 2 million ha burnt recently – much of it intense – left with huge conundrum of single age cohort. Options? Problems of asset management – infrastructure damage, people safety issues, settlement patterns – seachange/treechange implications – how to introduce landscape level management regimes – what appropriate in changing environment, what strategies best? Vic bushfire strategy – 6 themes – one about introducing ecol dimension into fire management plus 5 others. Need more info – support scenario analysis, which spp. most vulnerable to climate change and fire change, state shifts in veg and fauna, indicator spp, beyond state level – critical regions – what ecological responses to expect. Social and economic issues of fire.

**Tas – Louise Gilfedder** – similar trends – fragmentation, people movement in landscape. Models aren't working for Tassie – only a couple of data points in most analyses – working better in east where warmer and drier but not in west as that is getting drier and hotter too. Key increased rate of lightning strikes. Now much more fire as a result of strikes – 50% source 60% area burnt now whereas used to be small percentage. Decreased fire intervals being observed. King Island 2 fires in quick succession – lost 2 metres of peat. Lost paperbark swamp forest entirely. Prioritising efforts. National perspective alpine areas and rainforests with high levels of endemism very important – half of Australia's alps and high proportion of endemism. Conifer forests, very old, don't regenerate after fire – increasing occurrence of fire. Cost of fire management increasing – increasing pressure for prescribed fire and sacrificial zones. Lack of information and science still at biome level. Some really basic.

**WA – Lachie McCaw** – WA spans huge range of environments – fire season lasts 12 months – moves geographically. Manages huge areas of land including uncommitted crown land. Planned burning and context of burning important. Significant conservation values on non-reserve system lands. Clear trend in declining rainfall in SWWA – by 20%. IOCI useful work in better defining changes and attributions. IOCI 3 will shift focus to NW – hot spot over Pilbara –

infrastructure developments there. Also will look at tropical cyclones. SWWA green fringe around desert. Fragmentation issues. Loss of past refugia in hills and semi-arid zone. Specific issues – declining groundwater levels, changes in fire regimes, peats increasingly vulnerable, interactions with other events – e.g. frost events – impact on jarrah and coastal forests. Changes of boundaries for planned burning seasons. Future patterns of lightning ignition.

**NSW – Liz Tasker** – Two categ of concern – water quality and air quality. Biodiversity – what can managers do in terms of control burning. Fire suppression, where concentrate resources, when should they respond to fires, more specific regional predictions of what is going to happen. NSW diverse – need more specific predictions to formulate responses. Very flammable landscape – problems with urban/bush interface – tradeoff between biodiversity and people safety. Other key issues as to what biodiversity managers should be doing.

**ACT – Margaret Kitchin** – Concerns with increasing min temps at night affecting suppression capacity and backburning. Loss of communities in big 2003 fires, water quality issues. Fire management plan developed. Identified key threatened communities – sphagnum bogs – important for water and also for Corroborree frog, Reviewing strategic fire management plans to include climate change at present. Working to get more RAFT crews and improving skills to respond most appropriately to diff fire types. Location in the landscape – important to work collaboratively with adjacent states and monitor together and develop indicators for climate change.

**Qld – Rhonda Melzer** – protected area management perspective.

*Goal setting and planning*
There is a lack of reliable information on likely climate change impacts at the regional and local level. We need to be able to advise staff on how to respond in face of this uncertainty. There is a risk of 'paralysis' in on-ground management if we are not able to provide some guidelines and direction to steer field staff through the complexities and unknowns. So on the one hand we need to raise awareness that climate change impacts are not simple cause and effect relationships (e.g. interacting competitive relationships) but at the same time provide the means for on-ground management to go forward and achieve the best possible outcomes.

Considerable effort goes into reinforcing the need for clear and measurable objectives to be set in fire plans and planned burn programs. What are we trying to achieve? Fire management is then tailored to achieve those goals.

Current goals may become less feasible and achievable with ongoing climate change. Recognising which goals will continue to be achievable and which we have to 'let go' will be important. Acknowledging that we may no longer be managing to maintain the values for which a reserve was gazetted won't be easy – if it comes to that – and will require engagement with all levels of management and the public.

What will be realistic outcomes to aim for in a reserve and how will they be measured? The goals of managing to minimise loss, facilitate adaptation and maintain resilience are appealing … but how do agencies and the public determine what is feasible and what is acceptable 'loss'.

Can we transfer/modify our current approaches to maintaining the natural role of fire in ecosystems and maintaining healthy communities to fire management in the face of climate change? e.g. burning under conditions that: promote patchiness; promote native species over exotics; minimise the risk of losing critical habitat 'features' such as fallen logs and hollow-bearing trees? Perhaps the way to go is to develop a suite of 'habitat health goals/indicators', that are appropriate to different ecosystems, and associated fire management guidelines that can be used by field staff to guide on-ground management.

*Implementation*
The windows of opportunity to burn will become narrower and more critical.

Need new and novel approaches to management – cultural hurdle.

*Monitoring, learning and knowledge transfer*

We do need structured, scientific research and monitoring but we also need to foster field staff that know their reserves, recognise species and have 'an eye' for changes occurring around them. Rangers have an incredible range of responsibilities and there is high turnover in some areas. How to promote continuity in the gathering and transfer of knowledge – will become more critical in face of climate change.

## Plenary discussion: have we captured the issues?

**Some general points made:**

- Does any state feel they are (not) equipped to meet the challenge?
- We don't know what we need what tool box is required to respond to cc.
- Settlement patterns and landscape fragmentation – will underlie any fire response. A2A will focus attention on the N-S linkages and on how to manage for functional connectivity between reserves.
- Anything extra or that will exacerbate cc? What is it about climate change that is new?
- Can be optimistic compared with where we were 20 years ago. Knowledge being translated in all sorts of ways.
- Value of adaptive management approaches developed in the face of natural variability will presumably stand us in good stead.
- Mixed value of connectivity – doesn't benefit all spp.
- West Arnhem Land experiment – starting to meet its objectives – 50,000km being managed by aboriginal people using gas company money – good model.

Thursday, 27th March 2008 Location: The Common Room

**Fire and the National Biodiversity Vulnerability Assessment** *(Prof Will Steffen, Chair of the Expert Advisory Panel)*

Biodiversity Vulnerability Assessment and what to do to help biodiversity adapt. Outline of report and role of input.

5 parts to report – 1– set scene – groundwork chapter why biodiversity is what it is before people & characteristics;

2 – setting climate change context – climate change over time and rel/ship with people changes & ecological principles to how they respond to disturbance including fire – fire is important factor in this report, both natural and human affected);

3 – climate change and what is and might do to biodiversity in future; climate change operating clearly since mid 20$^{th}$ century, responses of biota showing; striking in marine systems because mobile – moving down n-s flowing currents; assessment of risk vulnerability to future climate change – not focussing on particular scenario – and indirect impacts from other disturbances such as fire and invasive spp; which will be winners and losers? extreme events, moisture regimes – not good handle on this yet – most models don't agree for Aust except SW WA and far south coast of VIC and SA;

4 – current management and policy tools & match between management principles and ecological principles; which ones of these particularly good in climate change and which less so – e.g. NRS still useful but for different reason than originally conceived (to conserve in situ);

5 – new things to do – new concepts, tools, governance structures. Taking more regional approach and including socio-economic change eg tree change – opportunities in land going out of agriculture. Rapid growth of population in SE Qld – implications for biodiversity. Don't know how biota will respond – stand back and give best opportunity to self-adapt – arrange environment to best allow for self adaptation. Need to manage adaptively – learn from how things develop.Discussion - Opportunity to raise level of management. Changed management approach from do nothing to high tech analysis ability – however transfer of

that knowledge to the on the ground is not good – this could be an opportunity to increase level of management access to better information.

Thursday, 27th March 2008 Location: The Common Room

## Technical presentations

**National Overview:** Climate change, fire regimes and biodiversity responses– *(Prof. Ross Bradstock)*

- Continental patterns of changes to fire regimes and their drivers – see power point; Why variation in the fire regimes across Australia? e.g. schema Russell-smith *et al.* 2007
- 4-switch model to drive fires – Biomass (S), Fuel (m), ambient weather(S),Ignition (I) – all need to be on for fire to go.
- Egs Top end Central Aust NSW Tas differences
- Can these examples be generalised? No. What else is missing? Semi-arid woodlands (temp, tropical) chenopod shrublands (thr by mediterr grasses)? What major gaps can we fill?

Discussion: NT – dry boundary against wet tropics would be useful to examine. Catch is that biomass – steps over exceptional systems that are particularly sensitive to fire. Systems under threat – issue of spatial weighting to "archipelago of rf remnants". Sensitive inclusions within larger systems.

Indirect impacts of $CO_2$ – additional biomass – woody biomass increase by 20% globally – feedback effects either positive or negative. Needs to be considered. Some projections for shrubs to do better in grasslands – would change equation considerably – changes fuel – changes fire interval etc. Important. Needs to be considered against moisture impacts on grasses too. Intensification of cattle grazing also could suppress grasses more which would also increase shrubs. No accident that semiarid possibly most sensitive to climate change – connectedness more likely to knock switch off more quickly. Avail moisture switch delicate and vulnerable to other changes. Treat fuel as a sensitivity analysis – build uncertainty for fuels into the analysis so cover options. Ignition switch needs more input. Trends in lightning may be important if combined with increased fire weather. Can we learn lessons from alps? Trade off between endemism and potential for change. Models based on functional groups – but groups may change. Species mixing. Tas alpine characterised by high degree of scleromorphy and movement of sclerophyll trees above treeline. Changes in seasonality – important in mallee – resprouters may not be able to resprout. [Underlying process perspective]

**Modelling fire regimes under climate change** *(Dr Geoff Cary)*

- Modelling under climate change – changes in fire weather and fire regimes or do we need to model intermediate steps
- Firescape model – example of process based modelling. Models elements of fire weather from ignition occurrence and location, fuel dynamics, fire spread, drought index etc. Climate change scenarios can be incorporated.
- In Flammable Aust 2002 – dramatic shift in fire freq with rising CO2 – increase to 0–10 yrs in north central ACT. Consistent trends coming from the models.
- Reduction of extinguishment of spread under climate change – partic important at lower intensity. Persist in landscape longer and so eventually run into extreme fire weather.
- Comparative fire model study Cary *et al.* 2006 landscape ecology – used 3 standardised climate change scenarios to examine how sensitive to any sort of cc. observed – warmer wetter – warmer dryer. General level of agreement – some sort of consensus btn models.

Discussion – potential changes in the vegetation modelling? Tas e.g. of changes from button grass to woodland. Debate about how to deal with it. Effects mostly fuel load dynamics. Becomes a problem for alpine ash communities. Linkages with NCAS. Sensitivity analysis of impact of cc. Button grass eg – unplanned area burned rel to planned burns increased with cc. depends on rates of fuel accumulation. Lowland Kakadu tradeoff between early and late burns – hillier country no tradeoff – more you put in more it burned. Different pattern in SE Aust (Gill). Different spatial arrangements matter. Important role about being strategic. Night – increased min Ts – interaction with ability to backburn, extinguishment. Spatial arrangement – patch burning creating heterogeneity in landscape. Effects of spatial arrangement less important than total area burnt in some circumstances. Patchiness.

**Regional case studies**

**Case study # 1 Sydney Basin** (Prof Bradstock)

**Discussion** – why downwood trend for woody resprouters? Fewer respouters at higher drier temps now. Any effect of fire suppression? Initial attack effect observed. Suggestion – template for model runs in different environments? Need to take a stepwise approach – sensitivity of most interest. Not worth running models everywhere – explore gradients of drivers – macrocontrast. State of knowledge problem for many areas. E.g. not much out there on extinguishment thresholds. CLIMATE CHANGE should directly impact on biomass etc. which should lead to response. Why not seen in Sydney modelling? Can be response to landscape structure – heterogeneity of landscape is important. Process based firefuel models could take away from Mcarthur problems. Social response to CLIMATE CHANGE – pressure for increased prescribed burning – plus impact of smoke.

**Case study # 2 Savannas** (Dr Adam Liedloff)

**Discussion**

Is there a spread model for gamba grass? – no, put in as grass into savannahs and looked at impact. Non-eucs in system – are they in model? – yes. Any trials on removal of gamba grass? Manipulation of systems may be necessary? – not very persistent seedbank, but good pasture spp so political issue. Fast moving problem in NT – no of treatments being tried. Can burn twice. Need to burn twice for prescribed burning. Intensive buffalo grazing works. If can eradicate a crop before seeds, have reasonable chance of success. Gamba making northern fires much more dangerous for people. Very different fire management scenario – novel fire regime from global change. Buffel grass is providing same pressure in central Australia. Demo model shows little change in tree canopy for 70 years but then huge decline to new steady state after 200 years.

Model use for hint of future changes – this one using basal areas – if were looking at life histories would get a hint before 70 years of the problems in life history. Management of woody thickening – lots of seedings resulting from floods now turned into big trees that are a problem. Conversely drought kills others. Capacity to manage consequences of drought in different locations – tree/grass ratios key question in northern Australia.

**Models** – will these be OK if all linked together? – care needed with models – easy to misuse in policy context if limitations not well understood and not well validated. Buyer beware. Look for comparative analysis as in IPPC. NSW TPC approach heartening to NT. Precautionary principles. Models should be used as conceptual hypotheses to establish monitoring frameworks. Flames model has lot of stochasticity in it – many runs to establish variety of scenarios under variability of situations. Adaptive management needing to be part of our response – needs to be well understood – management idea and anticipated response and measurement to see what happens – use model to make prediction of what happens and then monitor what happens and another loop to do with biodiv conservation – is the regime actually giving outcome wanted – need to work out how to build model into management process. Ecologists get hung up about modelling – economists power ahead regardless – we are perhaps being unduly cautious. In the north the concerns have been about the animals rather than the savannah itself. Why not possible to build fate of fauna spp. into the models? Managers want rule of thumb. Link conceptual models to give guidance on impacts on bandicoot trends. David Lindenmayer's work – applies to possums and algae. Flames can offer bandicoot question habitat info needed e.g. change in dead timber availability. Opportunities to look backward e.g. dendrochronology, historic context. Instrumental record length short. How did we get here using all the tools of history (Dave B) – tension between historic and prediction – prediction is based on historic understanding. Loss of central aust mammals is a warning that we don't understand the drivers. To approach complexity need large teams with ability to work together to … Fire may be less important than some of the other threats – how do we go about narrowing the focus to identifying priority areas for attention.

**Case study # 3: Response of plant spp and veg structure to 2003 fires in Kosciuszko National Park** (Michael Doherty)

Large area current but range of intensities – some areas no fire since 1939 – fire breakouts from driest areas on 18th. Strong environmental gradients and variety of spp. Year 1 – 80% vegetative, 20% seed only. 12 months post fire – 30% seedlings 45% flowered 15% flowered and fruited. <10 tree mortality except alpine ash. Fire killed spp more susceptible to frequency. Weedy opportunists often temporary – ongoing weeds similar to before fire. 55% fire killed spp on rocky outcrops – refugia "safe sites" very important for fire-killed spp. Hide in rocks and scree. Some didn't respond till yr 2. These species less likely to occur more broadly so not good indicator for TPCs. Structural effects of wildfire, most spp resprouters three types of responses seen - – epicormic resprouters; in higher severity areas – basal resprouting – need longer lag time before reintroducing prescribed burns, fire-killed response. Multiple recovery mechanisms in same spp – epicormic, basal and seedlings. Some areas massive die-off of seedlings and in moist sites survival of all seedlings. Even in same climate, microhabitat diffs led to diff responses. Even wet communities responded well. Changes post fire – composition, structure and function. Latter – carbon and water dynamics story questions. Floristic composition and structure respond to frequency, tree canopy response to intensity. False resilience where no recruitment. Diagram of vegetation response types – some that flip flop and those that are more gradual in impact. Dick: Story here is one of resilience. What is going to be use of this for management response? Mgrs taking broad points only. Real test will be in next fire management plan how this data will be used to drive management response – climate change highlights importance of protecting the rocky outcrops. Mgrs commitment to capitalising on long-term monitoring will be a question.

Thursday, 27th March 2008 Location: The Common Room

## Breakout small group discussion: research needs for national and regional analyses

*What are state and Commonwealth agencies doing now? How can methods discussed in this session be applied in other regions? What further analysis will enable adoption of such projects? What technical inputs are needed?*

9 big ticket items arise from what have heard in the presentations – break into 5–6 groups of 5-6 people; choose 3 of the 9 and Discuss for 45 minutes:

- 1. Regional FFDI projections – utility?
- 2. Regional demand for information and approaches shown?
- 3. Case studies – utility, priority?
- 4. Climate change induced changes to fire regimes as priority?
- 5. Governance adaptability and capacity to deal with climate change and fire
- 6. Monitoring effectiveness of actions – thresholds, property/biodiv assets
- 7. Examples of best practice
- 8. Opportunities – trading/offsets; partnerships – private/public
- 9. Gaps – fauna, reserve design

**Report back to group: (butcher's paper presentations)**

**Group 1** – Lalage – scenarios and gaps – info useful but better to be able to pull into decision-making hierarchy or framework to recognise scale and relevance to specific managers. Useful to ID which elements are important for the area they are in. Changes of seasonality in fire – coupled with observed phenology effects.

**Group 2** – FFDI – useful – link between climate change and regime change – use of different components such as increase in high and low danger ends. Wary of some of the last 10 years data because may be climatic variability rather than climate change. Ignition sources need more information – pattern changing through time, other measures such as fuel moisture content. Implications for achieving prescribed burn targets and implications of seasonal shift of burning programmes

Governance – strong regional variation even within states – all agencies policy level climate change priority but at operational level meeting current fire management targets more important. Lack of clear understanding of what impacts may be limits agency response. Issues of nonfire climate change related impacts of significance such as invasive spp.

Monitoring – lack of commitment to monitoring as worthwhile activity – value of information for adaptive management – expensive and time consuming so often set aside for higher priority activities. Take advantage of key events e.g. 2003 fires or major frost events – document events and responses. SMART monitoring, well designed, fauna, flora and weather. Also need field experiments in recently burnt areas, especially in areas to be burnt at frequent intervals.

**Group 3** – Louise

Monitoring – fire and fire management patterns at large scale – use remote sensing and tools. Fire management objective – biodiversity usually not reason for it – need to anticipate monitoring in areas managed for asset mgt.

Fire history maps important. Cross-tenure issues and inconsistencies. Opportunities re carbon trading to link to fuel loads etc. useful to identify key areas on maps and share information across agencies.

Institutional governance – yearly forecast meetings – have biodiv person there. Partnerships across agencies responsible for fire mgt. good visualisation tools important.

Information gaps – weeds, implications for reserve design, smart ways to supplement reserve design using carbon trading opportunities. Let go of the generic biodiversity management aim and greater focus on maintenance of processes. What can we learn from the paleorecord?

**Group 4** – Steve

FFDI – real opportunities for more nuanced understanding of FFDI as management tool. Need to generate better explanations of it and its use. Opportunities to project fire management tools into future if use back-casting and be able to provide better information

Better prediction of freq of lightning – opportunity for more refined analysis of that.

FFDI flows into regulation of fire bans – impacts on ability to prescribed burns. Defence lands – pressure not to have fire bans as they impact on their ability to light incendiary devices.

Monitoring – what are we monitoring? Do we have mechanisms in place to work out questions to ask? How to develop sound monitoring program. Under-resourcing of monitoring – argument could be crafted that money spent on monitoring could reduce need for money spent on suppression. Don't call monitoring research in agencies – wont be funded.

Best practice examples monitoring, mgt, etc. and good information transfer and mechanisms for managers to make things work.

**Group 5** – Rhonda Melzer

Governance – Vic – need to get fire right in the landscape – fire plan in Vic – public involvement – fire ecology strategy being incorporated into operational strategies. Qld – Climate change has recently been included in the portfolio of the EPA; climate change information is just beginning to filter down to management unit level. QPW has a comprehensive state-wide fire planning and reporting system which is currently being updated – the revised version will promote an evaluation of climate change issues in planning and implementation processes.

Qld cont. Burning planning uses a zoning scheme that includes burning for conservation outcomes. Information is lacking in relation to appropriate fire regimes for some regional ecosystems/vegetation communities and threatened species.

Qld cont. Each planned burn has measurable objectives. Simple surrogates are often used to help judge outcomes and to help field staff assess their planned burns. For example, a simple surrogate (for getting burn conditions 'right' to maintain habitat and structure) might be to retain >95% of hollow-bearing trees and large fallen logs (Dick I can go into this further if you want but at the time I probably didn't say much more).

Current objectives, measures and surrogates for ecosystem health may not be adequate for longer term trends.

Dick this paragraph is true of Qld but it may have been a general point for the whole group. Fire-related research/monitoring is currently largely undertaken by individuals who see a need for it or have a specific question to address rather than their being specific responsibilities assigned to positions. Continuity is therefore a problem if staff leave.

The requirements of fauna are largely not well known and therefore probably not adequately addressed.

Climate change not driving fire management – community concerns about hazard/expectations much more important.

Monitoring – Vic – see butcher's paper

Fire monitoring or research driven by personalities rather than agency led. Need longer term perspectives. Qld lacks agency fire ecologists. Fire biodiversity consortium in SE Qld is a good model – linked with unis, agencies and provides input to landholders. Fire North work is invaluable and landholders and NRM catchment groups are using the fire mapping work to improve on-ground fire mgt. There is a shift in culture from output to outcome management – positive. Vic – water concern driving some responses. Monitoring of effectiveness of asset protection rather than biodiversity. Easier to burn for conservation in remote localities. Adaptive management concept understood but not well operationalised.

Knowledge gaps – biodiversity response data needed, especially for fauna. Impacts of $CO_2$ fertilisation in terms of fuel accumulation rates etc. Vic – Management response (to large fires?) is to put more fire into the landscape – what is actually going to happen as a result?

**Will Steffen** – summary thoughts on the day (recording available).

- Management crucial and could outweigh climate impacts and will in turn will respond to climate change impacts on people.
- Connectivity– spatial arrangements role in relation to spread of fire – being able to connect remnants is important but has different role in terms of carrying fire – spread of invasives and fire – tradeoffs
- Fire sensitive refugia embedded in fire-prone matrix – pockets – what happens to them in face of climate change e.g. Wollemi.
- Impact of change in seasonality in rainfall and in fires – impact on phenology – vulnerability may change – changing rf regimes, earlier spring times etc. – disjoint btn fires and phonological stages.
- Importance of management – fire managed disturbance regime to social pressures – will increase and change with climate change and may be greater impact in the shorter term. To do with perception of cc, hard to understand and predict how they will respond.
- Socio economic trends at regional scale – treechange, changing structure of the ecosystems that will interact with cc. How triggers differ in different parts of country – overlap with physical framework – Ross's 4 switch model
- Surprises – multiple chain effects that couldn't predict, Monitoring is only way to pick these up. Learn from them. A lot of directly modellable ones may not have the greatest impact. Interpretation using basic principles.
- Palaeo-fire – any lessons? Climate change may not be biggest impact in short term but useful to look at what's happened in the past – palaeofire and palaeoclimate. What have they meant for the continent in the past. 3 degrees is very big shift and would anticipate big changes in fire regimes. Overall change from 5 million BP to now is 2.5 degrees and major shifts.

- Also important to look at last 500,000 years – warmed about 5 degrees since last glacial maximum – upper level of current IPCC projections is that sort of change, over only 100 years rather than 1000 years. Currently tracking at worst possible scenarios – tracking on 6 degree change by 2100. Things could change radically on decadal timeframe.

## Plenary summary, with additional points raised in discussion

6. Monitoring (4 groups)

- put aside too readily
- take advantage of major events
- exploit all sources and communicate
- biodiversity not usually primary objectives
- can anticipate where needed and set up monitoring now
- mapping
- mostly driven by individuals not agencies
- go for REAL adaptive management

9. Gaps in knowledge (3 groups)

- ecological processes vs. biodiversity
- season of fire
- weeds
- reserve/landscape design
- $CO_2$ fertilisation + fuel accumulation
- biodiversity response data (esp. fauna)

5. Governance/Institutional arrangements (3 groups)

- yearly forecast meeting … including biodiversity
- communication pathways and develop partnerships across agencies…working across tenures/zoning structures the planning
- structures … risk management approach (high risk because personality-driven not institution-driven

1. FFDI and its projections (2 groups)

- VHE fire days – easier or harder to do prescribed burning
- ignition/lightning

3. Usefulness of Information (1 group)

- but ….need decision framework re which
- which elements important for my system?

4. Where does climate change sit in 'pecking order' of concerns (1 group)

- regional variation
- high at policy level/low at operational level
- not really driving current fire management

7. Best practice examples of fire management (1 group)

- Kruger NP, South Africa
- including governance/systems (5)
- VIC – monitoring system
- QLD – benchmark sites

## Plenary summary and discussion

**Summary thoughts from Malcolm Gill (PowerPoint available)**

Current trends and fire implications
more people, more ignitions but…

- smaller parcels of land, agric intensification, more fire suppression
- more urban edge and more urban-edge fires
- carbon trading – increase in agric and hobby farming & forestry especially on abandoned cropping land, more fire suppression, increased likelihood of fewer higher intensity fires
- move from grassy to woody landscapes and more arson

Conservation of biodiversity

$CO_2$

Correspondence between national rainfall patterns/trends and fire. What would map of national fire management look like ideally esp if interested in carbon emissions? Perfect map for C would have no fire but clearly need balance between competing needs. NT – raw amount of precipitation important to determining extent of fire in a year. Low fire in intensively grazed areas with biodiversity consequences – seen also in SW Qld. Have to be careful with this fire map – this is map of event not fire regime. Still sharp line visible on other fire maps – why. Map if of areas more than 400 ha burnt in period 2002-03. Need map of Landsat image in 1788 – what did it look like?

**Plenary discussion of other issues (general notes and ideas; unedited)**

What managers can use – from dirty hands to higher policy needs. Ross's 4 switch model – is that useful concept? If that could be turned into map of what is likely to happen based on expert opinion that could be useful – but how would that help managers? Timing frequency and intensity for different areas might help to identify different sensitivities – structural elements – match up to identify vulnerabilities – match up sensitive parts of different systems to identify what is most at risk. Manager with ability to throw switches – how do you help them understand which one to throw when. Different levels of manager – ones that can translate science into mgt. Who are the target of this work? Mike's reality is constrained group of managers with competing demands – can't take home abstract complex message – need clear messages and tools. Maybe even principles or parables. Simple messages can be risky – eg message that early burning is good in Kakadu went too far when became entrenched in mgt. Guiding principles and also things to think about – things to watch for, issues to balance, competing implications. Biodiversity fire roadshow? Educating staff about issues to consider. Development of guidelines for best practice in fire management – thresholds, monitoring, patchiness in low intensity burns etc. More fundamental task for this group – need to understand management approaches occurring nationally now. Summation of approaches used to manage fire for biodiversity now – opportunity to compare and synthesise approaches before we develop any advice about cc. How to achieve that? Have a list of dot points that cover core functions but how …need agency people to give distillation of agency practices and underlying principles of approach. Lot is seat of the pants. How to overlay climate change on that? What is actual stated objective for biodiversity conservation? E.g. what was the background to decision to drop water on specific place at specific time – organisational drivers and capacity elements eg funding sources, pressures to take action, motivations in terms of economics vs non-monetary assets in different situations, eg urban edge, near forests, remote areas. Suppression is sexy esp near built up areas – disconnect with biodiversity needs. Is this captured in management framework/zoning across country. How much are plans used in the reality of fire suppression? Priorities in suppression are life, property/assets – biodiversity a distant third. Real value of plans is in the preparation before the extreme events. Vic developing system of big fire breaks across Vic – done in matter of days at Premier level in reactive way bc not enough patchy mosaic burning done beforehand so kneejerk reaction – strategic buffer strip approach prior didn't work. No assessment of the efficacy of the break approach or of the costs involved both economic and biodiversity – trying to do that now but with no discernible way to assess outcome. Fire behaviour guides not even used in planning fire responses in most cases – how to try to communicate principles that may apply to people with very different priorities and perspectives. Who should roadshow be to? Need to have it enshrined in policy and practice as well. Need template for state/territory outline of their current approach to fire mgt. Need this information as baseline for considering impacts of climate change and how they vary across the continent and match… Current institutional arrangements may need to change to reflect changed challenges of cc. AFAC may have done something relevant? State/territory reps not necessarily representative of their states – need wider engagement to prepare the response. Need little constellation diagram about fire and biodiversity – id agencies with roles, budgets, etc policy and planning instruments and decision support systems to help agencies make decisions about fire. Snapshot of immediate instruments that help people not all the detail of every possible line of legislation. Disconnect between policy and on the ground. May also highlight conflicts eg in s44 situations. Relates to fire management plan incorporating event management and sticking to it even if change it afterwards. 5% issue. Do need to influence policy support – simple question about fire management decision making support – do you have one, is it any good? Need to understand them to influence them. Will need to change how we do things so State/Terr review not really going to make much diff to need to do it. Worth quick review though. What is planning coverage and scope that we'd be aiming to influence. York – comparison of fire management plans between states – Vic all standard – other states vary between grass and forest – underlying principles in each state applied generically rather than responses tailored to the needs of that landscape. Some tiny little reserves had excellent plans because of local interest cf some plans for big places very ordinary. Generic templates very common. Could create new templates that are more tailored to different biomes and locations. Little planning at landscape scale that is actually implemented. Things improving but still disconnect… doesn't matter what's in the management plan – I draw the maps. Depends on how plans developed – if those people involved and have ownership makes more likely that plan implemented. Primary audience is decision makers in agencies as to what to follow up on in the face of cc.

Paul Williams Qld - Other management issues – climate change direct impact on fires and overriding influence on other threats – fragmentation and weeds – that may get worse. Aim to build resilience of systems. Underlying principles helpful. Guidance on implementation strategies for how much to burn and where – practical and tangible – where and how much to burn and why. Egs – key research questions – good experimental designed projects – transects over range of areas – guidance on monitoring and how to go about it – patches of scrub springs riparian strips in pastoral landscapes – more vulnerable – implications for them and what might protect them. Wetter Euc against rainforest boundary – fires may carry further – sandstone communities usually require less fire than surrounding landscapes – key fire management issues – exotic grasses in tropics – not declared weeds but worst threat to biodiv – fuel load dynamics – needs national structural mechanism to give guidance and information – centralised monitoring information holding facility or mechanism – CRC like. Internal agency review of data before goes to the central facility. Could be reports observations or data sets. Ability to generate fire histories for specific places but can't give principles for how to manage into the future. Suggestion that central planning top down model not as good as distributed network – virtue of diversity but opportunity for network of relevant people – opportunity to get together and interact. More modest achievable and effective response. Gill and Bradstock book. Anywhere that would be a good place to demonstrate climate change impacts would be useful for inclusion in report consideration of gaps and opportunities. Monitoring key need for the future. Will need help from this group in developing the report. Want it to be a two way street – want to see the other state details circulated as well as synthesised in the report. Take care not to achieve unrealistic expectations of this project – can point to directions. Scope of scenarios to deliver in this report. Certainty is impossible dream due to complexity. Fire and biodiversity and climate change is happening now – organisations need help as to how to respond. Understanding of switches and how they might change under diff circs in diff parts of Australia, how does that range of climate change affect the switches. What does this mean for biodiversity. Spp focus only useful if telling how healthy the functional processes are – how will these changes affect the regimes that affect the processes. And what are responses given that. Usefulness of the diff models – strengths of models and situations for their use. And other lines of evidence. And other needs in the face of uncertainty. We aren't going to get 100% solution – need to state that up front in the adaptive management context and to emphasise the need for mgt. does there need to be tweaking. Ignition switch. Switch model – do at different scales – local, regional, national. Ross's model.

**Final report structure:** working from a straw man draft for efficient filling throughout project to meet the needs of end users

**Key messages**

- climate change (warming, rainfall, $CO_2$, ignitions, fuel) influences FFDI shifts regimes – $CO_2$ increases fuel loads
- national framework – 4 switch model and ways of bringing climate change thinking into how switches will be turned on or off
- case studies – SE and North
  - regime and management interactions
  - specific uncertainty therefore need multiple lines of evidence
  - where to do the next ones in detail – SW WA heathlands;  woodlands, mallee
- state and territory needs – need for statement of principles/ postulates
  - snapshot of state/territory management now
- monitoring and adaptive management
- Discussion monitoring groundswell of interest and documents – Woinarski, Smythe etc. (Malcolm Gill) – lot to consider. How might monitoring differ around climate change questions. Extra imperative - especially important given climate change is likely to taking us into previously unchartered areas. Gill – can suggest who and how to do it – e.g. phenological data sets
- gaps and opportunities – partnerships/networks, how to share lessons learned – mechanism to distil. Skills of people on the ground to do the work? – variable – national perspective on where going nationally – identify what gaps exist in info in relation to switch and link to potential monitoring activities. (Neal).
- Add discussion of Feedbacks and interactions and nasty surprises to ecol consequences chapter.

Could use the proposed structure of the report as the starting point for the comments from this meeting – also an aid memoire for identifying points missing. Also clear dates and timelines – including 'bureaucratic onion rings'– plus who doing what part of report.

**Management needs:**

Audience – state agencies, C'wealth, NRMMC – R&D and how to spend taxpayer $$ - adaptation R&D

Content – Preliminary outline – work so far, gaps, pathways to 2020
See report outline
Key outcomes of this meeting and relationship to report.

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In addition to Consortium members

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