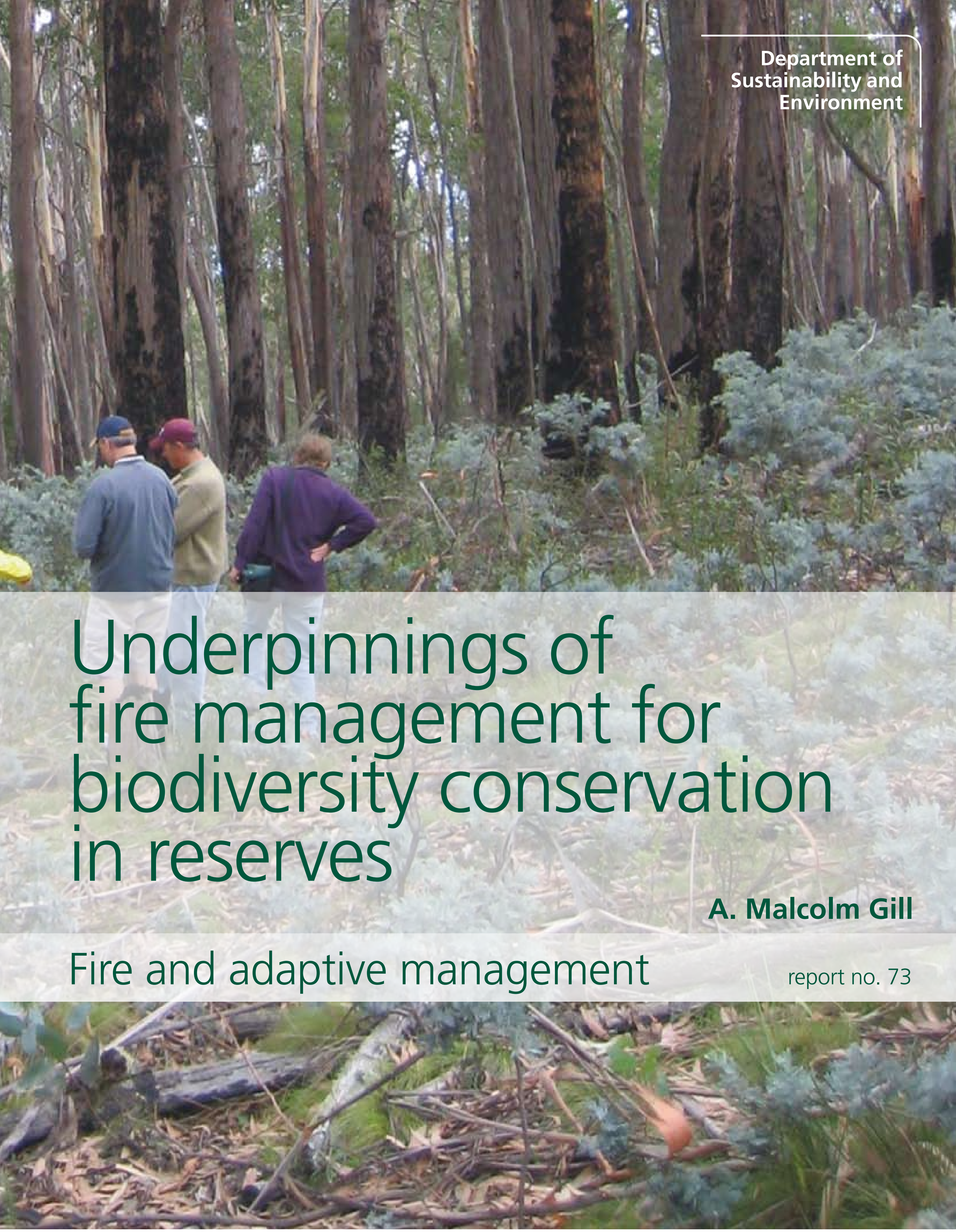


Department of
Sustainability and
Environment

Underpinnings of fire management for biodiversity conservation in reserves

A. Malcolm Gill

Fire and adaptive management

report no. 73

A Victorian Government initiative

Victoria
The Place To Be

Underpinnings of fire management for biodiversity conservation in reserves

Fire and adaptive management

report no. 73

A. Malcolm Gill

CSIRO Plant Industry, GPO Box 1600, Canberra, ACT 2601

Fenner School of Environment and Society, The Australian National University, Canberra, ACT 0200

Bushfire Cooperative Research Centre, Albert St, East Melbourne, Vic 3002

Published by the Victorian Government Department of Sustainability and Environment

Melbourne, November 2008

Authorised by the Victorian Government, 8 Nicholson Street, East Melbourne.

Printed by Stream Solutions

Printed on 100% Recycled paper

ISBN: 978-1-74208-868-6 (print);
ISBN: 978-1-74208-869-3 (online)

For more information contact the DSE Customer Service Centre 136 186

Cover photograph: Malcolm Gill

Acknowledgements

This contribution was initially inspired by senior officers of the Victorian Department of Sustainability and Environment (DSE) – especially Mike Leonard. His encouragement, and that of Gary Morgan and Dr Kevin Tolhurst (University of Melbourne, Victoria), is greatly appreciated. Facilitation of this project by Francis Hines and Liam Fogarty is gratefully acknowledged. Liam Fogarty, Gordon Friend and Andrew Wilson, all of DSE, provided many useful suggestions after scrutinising the draft manuscript. Angie Kelly kindly assisted with editorial matters.

For generously assisting in providing various details for Chapter 2, I would especially like to thank Ian Dicker, New South Wales National Parks and Wildlife Service, and also Nick Lhuede (Rural Fire Service, ACT), Simon Hemer and Liam Fogarty (formerly of the Department of Urban Services, ACT), Leigh Miller (Country Fire Service, South Australia), Dave Broderick (ACT Bush Fire Council, ACT Volunteer Brigades Association and formerly a volunteer with the Country Fire Authority of Victoria), Mike Leonard (formerly DSE) and Nick Ryan (DSE). Ian Dicker generously read and commented on this chapter.

Dr Ian Lunt of Charles Sturt University made constructive and detailed comments on Chapter 4, which were much appreciated. Discussions with Dr Joe Walker and David McKenzie on the grazing issue were also valuable. For reading the entire manuscript and offering helpful suggestions, I would like to thank Dr Simon Heemstra, New South Wales Rural Fire Service and Dr Mick Brown. The assistance of Dr Jon Marsden-Smedley, Dr Jeremy Russel-Smith, Dr Peter F Moore and Mike Wouters for reviewing and commenting on a number of chapters is also greatly appreciated.

In a general sense, I would like to thank Professors Peter Kanowski and Brendan Mackey for my appointment as a Visiting Fellow at the School of Resources, Environment and Society, The Australian National University (ANU) – now the Fenner School of Environment and Society. Similarly, I am grateful to CSIRO Plant Industry through the Chief, Dr Jeremy Burdon, and Program Leader, Dr Andrew Young, for supporting me as an Honorary Research Fellow.

For great encouragement and constructive debate over years or decades, I would like to acknowledge Professor Ross Bradstock (University of Wollongong), Dr Geoff Cary (ANU), Dr Karen King (ANU) and Dr Rod Weber (University of New South Wales) – my colleagues in the Landscape Fire Ecology Group at ANU and Project B1.2 of the Bushfire CRC.

As always, Beverly has had a strong and helpful supporting role in my work.

Australian Government
Attorney-General's Department
Emergency Management Australia

This publication has been produced with the support of the Commonwealth Natural Disaster Mitigation Program as part of the DSE's Learning by Doing Project.

Contents

Preface

Nowhere in Australia are bushfire matters more contentious than in south-eastern Australia, where the twin tragedies of the destruction of economic assets and the loss of human life are graphically presented by the media, along with dramatic pictures of high intensity fires. What should be done about such fires?

Media reports implicitly highlight the question, 'What are our assets?' Human life and property are major societal assets. Pastures and stock are rural assets. Native plants and animals, catchments and infrastructure are also assets. Some assets are affected by one event – houses, human lives, trees and individual animals – others by a sequence of fires, especially species of plants and animals. Thus fires affect a diverse array of natural and societal assets, individually and collectively.

Topics within the discipline of bushfire science and management are numerous and complex. If the knowledge needed for making informed decisions had already been acquired, and if the conceptual understanding necessary for informed application of knowledge had already been gained, then matters of contention would appear to be readily resolved. However, there is still much to learn and the debate involves wider matters, such as: government priorities and budgets; what different groups in our society consider to be *assets* (a reflection of their values); the methods and speed of fire suppression; land-use planning; the nature and speed of recovery after major events; and how landscapes and their components respond to fires.

Responses by authorities and landowners to these issues usually involve intervention in the landscape, through fire suppression and fuel manipulation. These are inextricably linked with the land use being considered. A farmer's approach to fuel management is likely to be quite different to that of a conservation manager, or that of an urban-interface dweller, because of the differences in assets being considered and the resources available to them.

Given that the range of bushfire issues is so large, no short publication such as this can address all of them. Here, the purpose is to present and explain some of the ideas underpinning fire management for the conservation of the indigenous organisms (biodiversity) present in public conservation reserves. This may involve continual attempts to eliminate exotic, naturalised species. Thus the emphasis here is the effect of fire and fuel management on native plant and animal species in reserves; such management takes place within the general context of the protection of human life and property.

Fire (and fuel) management is a difficult and contentious practice. Fires can escape reserve boundaries and create problems in adjacent landholdings. The reverse can also be true. Outside, but adjacent to, a reserve there may be different landscape objectives and different impacts of fires on those objectives. Land just outside the reserve boundary may be owned by another public agency or by private landholders – the extreme being an urban interface. Neighbouring land uses affect reserve management.

Fire management involves, among other things, preparation for unplanned (wild) fires. Networks of tracks and trails, prescribed burning and grazing are common themes of preparedness, but are also items for heated discussion. These topics are covered in the following chapters.

There are no easy answers to the problems of fire management for biodiversity conservation in public reserves. Knowledge of biodiversity is usually incomplete; knowledge of the interactions between the elements of particular biodiversity arrays is inadequate; techniques for the fire management of biodiversity are continually being updated; and unplanned fire occurrence is probabilistic. However, alternatives are presented for consideration and reasons supporting or not supporting them given against the background of a rapidly expanding literature.

This publication is intended to encourage the exploration of ideas rather than be a manual of what to do and when; it is not intended to cover all the issues of fire management. Hopefully the reader will find awareness, stimulation, ideas and knowledge here that inform fire management of biodiversity conservation in public reserves. It is designed as a support – an underpinning. Providing clear-cut answers for all the circumstances of all reserves, let alone one, is impossible because of unique suites of indigenous and exotic species, land use, fire histories, fuel types, probabilistic events, extent of knowledge and the neighbourhood circumstances that prevail. Emphasis here, therefore, is placed on information, ideas and explanations.

Managers of public reserves are employed by governments. They are focussed primarily on the environment but may also have a role in firefighting where the protection of human life and property is paramount. In Australia, emergency services and rural fire services are led by professional officers but are staffed by numerous volunteers. The firefighting approach has been response-oriented in many landscapes, and featured as heroic in the media. Biodiversity conservation is much less dramatic, although issues surrounding charismatic fauna, like Koalas (*Phascolarctos cinereus*), can evoke strong responses.

How to encompass both life and property protection objectives and species-conservation objectives has been the traditional dilemma in the management of conservation reserves. In approaching this topic, one can focus on biodiversity conservation and then look at fire management to enhance the chances of species, genes or community survival. Alternatively, one can look at fire management from the point of view of the protection of life and property alone and consider how to modify fire behaviour and increase the chances of unplanned fire control. A third alternative is to attempt both. The choice adopted here has been to approach the topic from a protection point of view initially, then emphasise the effects of fire management on biodiversity and the landscape generally. In this way, the consequences to the environment of the more interventionist approach often desired by firefighting groups can be examined in the light of the aims of biodiversity conservation. This is the path less trodden.

This publication has a south-east Australian emphasis, but draws upon the wider literature. It is hoped that it will have widespread geographic appeal. It is intended to be a starting point for students of the environment in the broad sense: university students, lecturers, environmental scientists, managers of biodiversity reserves, park rangers, naturalists and fire-agency officers. However, it is to the thoughtful manager of biodiversity reserves that this discourse is particularly directed.

Foreword

South-eastern Australia has many areas of very high conservation value. It also has people and assets and is highly prone to fire. In recent years the area has experienced extended droughts, mega-fires and floods and faces the uncertainty and effects of climate change. Managing this landscape is complex and challenging.

A major source of difficulty is the connection between issues, place and time. For example, decisions about fire and water catchments on one side of the mountains may impact on livelihoods and biodiversity more than 100 km away and 50 years hence.

Knowledge is essential but incomplete. Therefore, a manager such as DSE must use the best knowledge available and predict the likely consequences of action – and just as importantly, inaction. We must make decisions and move forward, while continuously learning and adapting.

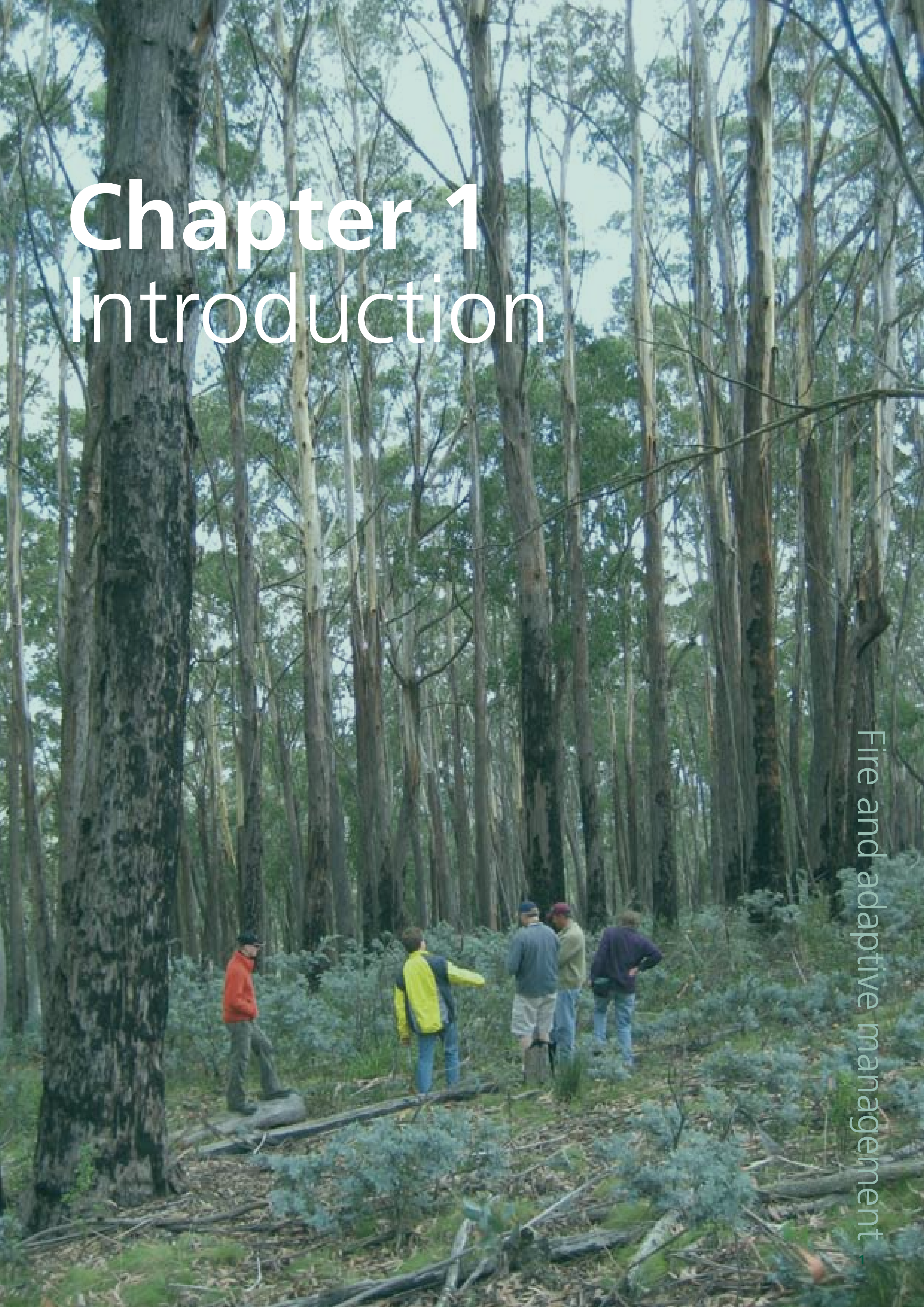
The author of this report, Dr A Malcolm Gill OAM, is very aware of these challenges and is well qualified to offer insight. He is a pre-eminent fire ecologist who has authored or co-authored over 200 scientific works. Dr Gill was also one of two independent experts who assisted in the conduct of the 2002–2003 Victorian Bushfires Inquiry. Most of all, however, Dr Gill has long recognised the need for managers to be adaptive and to account for multiple needs, not just those that fall within his main interest area of fire ecology.

This report summarises existing knowledge about the management of fire in relation to the conservation of biodiversity reserves. It has the worthy aim of exploring ideas and promoting discussion, not of dictating particular actions. It will provide significant support for decision-making about fire in Victoria and elsewhere.

Ewan Waller

Ewan Waller
Chief Officer Fire and Emergency Management
Department of Sustainability and Environment

Chapter 1
Introduction



Chapter 1 Introduction

Management in brief

Conservation reserves range from large wilderness reserves to small, isolated remnants in modified landscapes. Reserves can be embedded within regions dominated by agriculture, pastoralism, cities or commercial forestry.

The native flora and fauna of reserves – *biodiversity* in its common meaning – has changed over time. Some species have become extinct within particular reserves, while some have disappeared from the continent altogether. There are species that have invaded reserves as exotics from foreign lands or as natives out of place. Populations of indigenous species of plants and animals have changed too, as have the occurrences of certain landscape events. Because of all these changes, management of conservation reserves can be seen primarily as some form of *restoration management* (Gill 2003).

Of special significance is that fire regimes – the sequence of fires occurring at a range of intervals with different properties at various times of year in different fuel types (Gill 1975, 1981) – have changed over time. The role of Aboriginal people in ignition has changed; the various influences of non-indigenous settlers has changed; and contemporary human communities are affecting fire-induced change through planned ignitions, carelessness and arson. Furthermore, fire suppression has changed dramatically since European settlement, and markedly since the Second World War, and it continues to change, with increasing emphasis on technology.

Do we need to try to domesticate fire regimes in conservation reserves? In one sense, the answer is easy – we have no option. This is because responsible management and law dictate that fires originating within reserves should not be imposed on those outside reserves. Similarly, fire regimes suited to lands adjacent to reserves may not be suited to the reserve so should be kept out of the reserve. Furthermore, protection of human life and economic assets, both inside and outside the reserve, is an ever-present imperative that calls for intervention of some sort. If fires are eliminated from areas external to the reserve, the flux of fires across the reserve boundary necessarily change. Given this, fire regimes change to various extents from the boundaries inward and require intervention if they are to be restored.

While the above observations do not answer questions about the relationships between fire and biodiversity directly, they do indicate that intervention is usually inevitable for biodiversity conservation in fire-prone reserves. Assuming intervention is necessary, there are three ways in which this can occur, within the context of protecting life and property[1]: (i) establishment of fuel breaks and buffers that provide resistance to fire spread; (ii) direct fire suppression; and (iii) fuel modification or reduction. While each of these has its merits in relation to domesticating fire regimes, they also have environmental effects that may or may not enhance conservation value or the visitor's appreciation of biodiversity. The premise of this publication is that it is important to understand biodiversity conservation management issues from both a biodiversity point of view and firefighting and fire management point of view.

In Australia, most of the land set aside for conservation of biodiversity is managed by government agencies. Their attitudes, knowledge, skills and resources can have a strong affect on management processes and outcomes on the ground. The attitudes they exhibit may be quite varied but show particular emphases, such as leave nature to get on with its work or be strongly interventionist. The agency may see itself as an acquirer of land rather than the manager of land already acquired. It may see itself as a *learning organisation* (knowledge or evidence based; Senge 1990), or as just another government department. Agencies may see themselves as strategists, politically wise, businesses, politically sensitive, doers, experts, battlers against the odds, good neighbours, team players or processors of forms. They may be up-to-date or way behind scientifically. Regardless, what is important is the outcomes on the ground in relation to the democratically (i.e. government) determined aims and objectives for the management of landscape.

1 There may be off-reserve programs, like community education, also.

This publication does not give a description of how to do things in the field. It is not a prescriptive blueprint for managers. As such, it does not provide simple answers to complex problems. Rather, it is a document that aims to provide a thoughtful basis for serious discussion and debate about fire-management issues, with particular emphasis on practices suited to conservation reserves. It examines a series of topics of major interest, rather than try to be uniformly informative about all topics of relevance.

The title of this publication refers to underpinnings – a background to ideas and ways of looking at various issues, and management – any deliberate process undertaken by staff of the management agency that affects the successful attainment of the aims and objectives of management. *Fire management* consists of actions that directly or indirectly affect the presence of fire or its behaviour. *Biodiversity* can be taken as the variety of indigenous[2] organisms, great and small. Genetic diversity is not explicitly considered here, although it may enter the definition of biodiversity. Similarly, the variation among communities or ecosystems may be considered to be a part of biodiversity, but again, this variation is not explicitly considered here.

The management process

Landscape context: assets

Declaring a reserve, then fencing it to mark the boundaries, does not ensure conservation. Across borders of reserves there may be exchanges of plants (exotic and native), accessions of fertilisers and pesticides from the rural or urban side, flow of animals (including introduced and native species), and the spread of fire, smoke or water. The flow of people can be significant too. The significance of exchanges across boundaries may be a function of the size of a reserve: the smaller the reserve, the greater its perimeter-to-area ratio and the more likely that exchanges will have a significant impact.

The wheel of management

Management processes are linked together in a cyclical system, which can be represented as a wheel (Figure 1.1). This system is sometimes called *adaptive management* (Holling 1978). The wheel is used to imply that all the sectors of the rim need to be in place for a smooth passage of the management vehicle. The sectors chosen will vary among observers, but the general idea will remain the same. A hub called quality control could be linked to the rim by spokes designated as performance criteria, such as clarity of objectives, comprehensiveness of inventory or adequacy of planning.

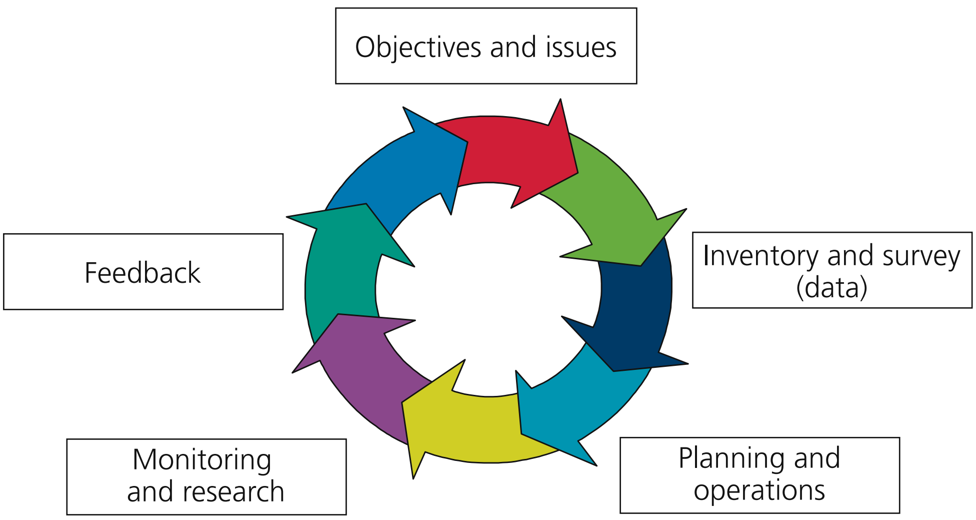


Figure 1.1 The wheel of management (Gill 2003). A simplified illustration of the continual learning process that is ideal for effective management.

2 *Indigenous* is, strictly speaking, more appropriate than *native*, as some species of native plants and animals are now found in reserves outside their traditional distributions.

Objectives and issues

What processes are needed for the conservation of biodiversity (indigenous plants and animals) in reserves?

The management aims and objectives[3] of various agencies for biodiversity conservation are backed by a range of legal documents in different jurisdictions and include, for example, the declaration of reserves under certain Acts and the declaration of rare or threatened species. There may only be a few aims for biodiversity conservation among the many aims for a reserve. The aims for the same reserve may concern biodiversity conservation, protection of human life and property, recreation, water supply and cultural heritage. Roads through a reserve may be a conduit for travellers. Increasingly, reserves may be seen to have a part in the dynamics of greenhouse gases.

Inventory and survey (data)

Survey implies a systematic process of assessment, whereas inventory implies collation of a list of all items of interest. Both are of value in a conservation context. The items chosen for the process reflect the aims, interests and priorities of the observer. They may also reflect budgets, level of knowledge and the ease by which it is attained. For example, birds may be easier to inventory than vascular plants, and bacteria are harder to inventory than small mammals. *Fire area* may be easier to record as a number in a report than as a map in a Geographic Information System (GIS). Fire properties, such as fire intensity, vary within a fire and are often ignored in reports and maps, yet they are important to document if there is to be an understanding of the effects of the fire itself, as well as the fire regime.

There are numerous conservation reserves without the benefit of a list of all vertebrate animals and vascular plants, let alone a full inventory of all organisms great and small. Atlases of fire maps are growing, but long periods of collection of accurate maps are needed if we are to understand what our local fire regimes are.

Planning and operations

Reserves usually have Management Plans and Fire Management Plans. Some have explicit Work Plans for a variety of objectives. To accommodate variety, Management Zones may be created, such as asset-protection zones (in this case, assets mainly being houses and people) and ecological zones. In New South Wales (Bush Fire Coordinating Committee 2005) and Victoria (Victorian Department of Sustainability and Environment 2006, pp. 14–16) there are four fire-management zones. In the Australian Capital Territory (ACT) (Australian Capital Territory 2004, p. 43), there are three fuel-management zones. Declared zones allow managers to operate more effectively in terms of the treatments they may apply for the priorities at hand. In the asset protection zone, burning to minimise fuel levels may take place regularly and frequently, but in an ecological zone may be less frequent and less regular and more prone to unplanned (wild) fire. Different jurisdictions have different zoning systems, although there are common elements between them, like asset protection zones near houses.

Monitoring and research

An important management principle is that of the continual update of concepts, knowledge and practice to achieve desired results. Experimental management may be invoked where the effects of differences in treatments are monitored formally or informally, and the results recorded, analysed and evaluated to see whether or not changes in operational practices are needed (Hopkins 1987). Such adaptive procedures can be applied at all levels in an organisation. In the field, there's usually enough doubt or lack of knowledge to suggest that monitoring in some form is always necessary. Monitoring methods depend on the knowledge context, the skill base and the importance of the issue. For local application, it is best carried out by managers whose practices are the most likely to be affected by the results.

3 Aim implies a direction, whereas objective attains a goal. However, in practice, aim and objective are often used as synonyms.

Monitoring of a stricter kind – with hypothesis testing using replicate treatments – is important too. This is best carried out by researchers or field workers using scientific advice, and is really research. Any *recipe* found to optimise outcomes in one area does not necessarily apply in another[4]. Therefore, in the adoption of research results from another region, identifying the underlying principles through an examination of the research assumptions and details of the ecological context of the research is worthwhile. Local applicability of a recipe could be tested in a simple and carefully monitored trial.

If conserving biodiversity is the key objective of management, then it should continually be assessed in relation to known processes that affect it (e.g. fire regimes), while simultaneously searching for change that may be due to other ecosystem *drivers* (e.g. global warming). Applying a system of monitoring is often difficult due to the shortage of resources (or sometimes organisational imperatives). Thus strategic planning with an emphasis on minimal sets critically formulated is essential (one such scheme for plant species in fire-prone reserves was described by Gill and Nicholls 1989).

A key challenge for managers is to know which of the many ecosystem and infrastructure components and processes present in a reserve need to be monitored, and decide when, where and how to implement an effective monitoring system. What is appropriate to any one situation will depend on the nature of the local conservation resource, issues arising from local operational procedures, availability of staff and the environmental, historical and cultural context of the reserve. *Components* to be monitored may include populations of rare or threatened species (their performance), key plant and animal indicators, weeds, feral animals and vegetation structure. *Processes* to be monitored may include prescribed burning, grazing, road and track formation and fire-suppression actions, such as the use of chemicals, burning out of fuels ahead of a fire, driving across areas without roads and tree clearing.

Feedback

Feedback is highlighted here, although it is often considered within the context of monitoring. It is separate as the results of monitoring and research could be considered to be of local reserve interest only. However, in the wider sense, results may need to be examined by policy makers for their consideration and action. An explicit, routine practice involving all the people concerned seems appropriate.

In conservation management, the principle of continual learning by continual re-examination of the success of operations, followed by refinement or alteration of processes in order to achieve explicit objectives is still in its infancy. The need for continual learning also applies to the local and general knowledge of the conservation resource, where there are continual changes in the appreciation of species and landscapes from historical, scientific and cultural points of view.

Feedback may take place at different levels – there can be wheels within wheels – for example, the many sub-processes that comprise an operation, like prescribed burning, can be monitored and the results fed back into an improved operation. Some aspects of this may only apply locally, such as the way the weather is affected by local features, while others may have organisation-wide value, such as the prescription used for the burning of heathlands. Feedback can be conducted to involve a whole range of interested parties (Hopkins 1987).

4 For example, Gill (1996) discusses the various recommendations made for fire management of the Ground Parrot's (*Pezoporus wallicus*) habitat in eastern Australia.

Fire management – especially fire suppression

Fire management – defined as consisting of actions that directly or indirectly affect the presence of fire or its behaviour[5] – may be seen as the set of actions involved in the preparation for and practice of suppression of unplanned fires, and the preparation for and practice of prescribed burning or other fuel-modification techniques. In other words, there are pre-suppression or indirect operations and direct suppression operations. Prescribed burning, in this classification, would be a pre-suppression measure. This is not to deny that prescribed fires may be ignited for ecological reasons, as well as increasing the chance of quickly extinguishing unplanned fires during a fire event.

In this section, an outline of methods and limitations of direct fire suppression is given. Prescribed burning is considered in detail in chapter three. If a prescribed fire gets out of control the same limits apply to it as they do for any unplanned fire.

Fire suppression

Direct fire suppression is effected by spraying water or chemicals on a fire or just in front of it, creating mineral-earth fuel breaks in the fire's path and burning out the fuel between the fire front and the fuel break surrounding the fire. A short classification of techniques for fire suppression follows:

1. **Use of water or retardant chemical** – Placement of water, with or without foam suppressant, or flame retardant on or adjacent to fuels using light units (small 4WD tray-back utilities with pump and relatively small water tank e.g. 400 litres), 4WD tankers and fixed-wing or rotary winged aircraft.
2. **Use of narrow fuel breaks** – Narrow fuel breaks may be established using rake-hoes or other hand tools and are usually used in forest country with litter fuels.
3. **Use of relatively wide fuel breaks** – These may be established using graders or bulldozers; the former usually in grassland, the latter in forests.
4. **Use of fuel removal by burning out** – This is usually carried out when weather conditions are relatively mild (e.g. during the night), in order to remove all the fuel between the fire perimeter and a fuel break (containment line). The idea is that as weather conditions become more severe the following day, the fuel to support a high intensity fire will be absent or markedly reduced.

Many suppression operations involve a combination of some or all of the above techniques.

Direct suppression is often followed by *mopping up* or *blacking out* close to, or adjacent to, a fuel break. This is the process of putting out all smouldering materials that may allow the later flaring up of the fire and, perhaps, its spread across the fuel break.

Limits to fire suppression

While experts may not take seriously the idea that suppression will be immediate and always effective, the public often treat it seriously and with confident expectation. This section explores the limits to suppression, even when using high technology: helicopters and other aircraft; modern 4WD tankers with chemical enhancers in water held in high capacity tanks; sophisticated communications networks; remote sensing of *hot spots*; computer predictions of fire behaviour; explicit command structures and procedures; automatic tracking systems for appliances; and well-trained and equipped personnel.

5 A wider definition of fire management may include equipment purchases and maintenance; crew recruitment, training and retention; fire detection; forensic determination of fire cause; community liaison and law enforcement.

In general, there are four types of limit to fire suppression (Gill 2005). They are:

1. Fire is too intense

The intensity of a portion of the fire perimeter is defined as its rate of spread multiplied by the fuel load multiplied by the heat yield (Byram 1959) – it is the rate of heat release per metre of fire perimeter. The unit of intensity is kWm-1 when fuel load is expressed in units of kg m^{-2}, rate of spread is in m sec^{-1} and heat yield is in kJ kg^{-1}. Intensity is sometimes expressed as megawatts per metre, MWm^{-1}. One MW is equal to 1000 kW. Note that the unit of Byram's intensity is not kWm^{-2}. When a fire starts, its intensity is close to zero. The intensity then builds up to a quasi-equilibrium value that depends on fuel, weather and terrain. The more fuel there is, the drier and windier the weather at the time, and (in general) the steeper the terrain in the direction of the wind, the higher the fire intensity at the leading edge (head) of the fire.

On arrival at a fire, immediate suppression may fail as the intensity of the fire has already exceeded the threshold for its control. A threshold exists because: a fire in the crowns of trees may be out of reach; it is unsafe to approach because of the heat; or it is starting spot fires down wind (see Text Box 1.1, Plate 1.1) at distances that could cause entrapment or make any local control irrelevant. Under extreme conditions, exceeding the threshold may occur within a short time from ignition. Thus failure to control the fire, even if it starts close to a fire-brigade depot, may occur because the arrival time for appliances was too long in relation to the rapidity of the fire's development. However, there are many reasons why response times might take much longer than the time for the fire to accelerate to uncontrollable intensities, such as failure to detect the fire quickly (e.g. due to night ignition, obfuscation of origin by smoke from other fires or absence of a formal detection system), a long travel period needed to reach the fire or the inaccessibility of the fire. For bush fires in remote areas, the aim of reaching the fire within the acceleration period may be unreasonable due to the cost of establishing sufficient fire-brigade units in areas of low population density.

The aim of land management might be to keep all fuels below a value so that, even during the worst possible weather for firefighting, the fire intensity never exceeds the threshold for control at any particular place, especially in forests (Gill *et al*. 1987*b*).

Funding for fuel reduction works may be limited, and the cost of fuel reduction might outweigh the value of the asset at risk (e.g. prescribed burning fuels in a young pine plantation would kill the crop). In pastures, fuel reduction means asset loss as the sward is the fodder for domestic animals there.

Plate 1.1 Spot fires – new, isolated, spots and elongated patches of flames from older spots near the edge of the main fire – in eucalypt shrub woodland and forest in Canberra, ACT (Gill 1991).

Text Box 1.1. **Spot fires**

Spot fires are spawned from a parent fire, especially when its intensity is high. Two forms of spot-fire phenomena are usually distinguished – *short-distance spotting* and *long-distance spotting*. Long-distance spot fires may occur up to 30 km from the fire front (Luke and McArthur 1978, p. 102), are usually completely isolated from the fire front, and occur singly or in small groups. They arise from burning materials carried aloft in a convection column, but eventually escaping from it and falling onto an ignitable fuel that is then set alight. The materials causing these spot fires can be called *burning brands* or *fire brands* (Gould *et al*. 2007, p. 117) – ignitable materials of sufficient mass to carry the distance without burning themselves out. Fire brands may originate from a variety of species, have different compositions (e.g. bark, capsule), be glowing or flaming and large or small.

Short-distance spot fires are closely associated with the fire front. The proportional density of fire brands causing these, *D*, has a negative exponential relationship with distance, *d*, in metres downwind of the fire front (Tolhurst and Howlett 2003, Tolhurst and MacAuley 2003, Gould *et al*. 2007, p. 125):

$$D = e^{-a.d}$$

where *a* is a constant that is likely to vary according to the wind speed, height of source and fuel ignitibility. There is little information available. For the fire conditions of Tolhurst and Howlett (2003), *a* was equal to 0.007 (see Figure 1.2); for a fire in the experiments of Gould *et al*. (2007, p.125), it was nearly 10 times as much, 0.057. To obtain a density of embers, a multiplier, *D0*, is needed. In the case of Gould *et al*. (2007), it was 27.2 for one of their fires (p.125). *D0* will vary with fuel type and condition.

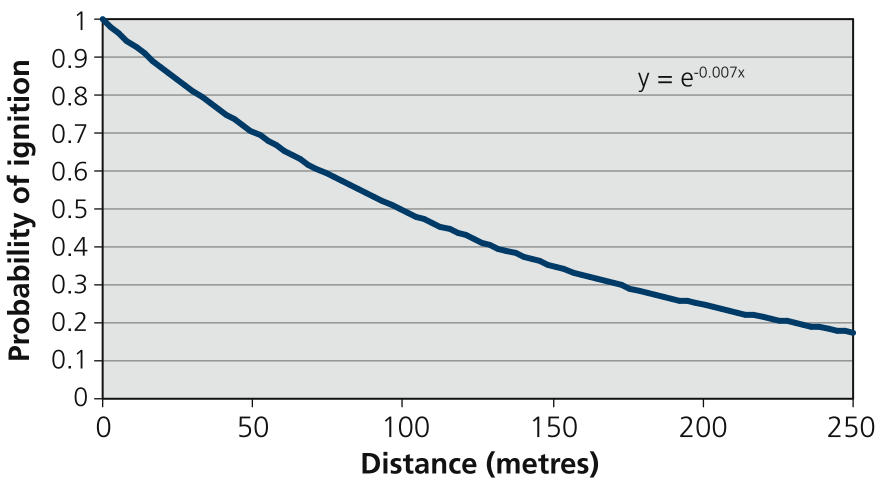


Figure 1.2 A graph of the probability of spot fires using the equation of Tolhurst and Howlett (2003). The value of the exponent (shown as -0.007 on the graph) can be expected to change with circumstance.

For long distance spotting in forests, McArthur (1967) expressed the distance of spotting, *K*, in km, as a function of the fire rate of spread, ROS_k in km hr^{-1}, and fuel load *W* in t ha^{-1}:

$$K = ROS_k(4.17 - 0.033W) - 0.36$$

(see Noble *et al*. 1980). There has been no formal test of this equation. Note that fire intensity is represented through the variables ROS_k and *W*. The equation can be re-expressed as:

$$K = INT(a/W - b) - c$$

Where *INT* is the intensity in kWm^{-1}, and *a*, *b*, and *c* are constants.

Gould *et al.* (2007, p. 118) have an initial model for predicting maximum spotting distance. It is based on flame height and above-canopy wind. These authors canvas the many possibilities relating to spot-fire models.

2. The rate of fire perimeter growth is greater than the rate of extinguishment of fire perimeter

On arrival at the fire, head-fire attack may be possible (i.e. below the intensity threshold), but the rate of perimeter growth may be too great for the suppression force attending the fire to adequately counter it. The perimeter of the fire can be inaccessible to ground crews, so suppression success can be limited. It is possible that the suppression force originally present would have been adequate if there had been no requirement for vehicle maintenance, repair, refuelling, refilling with suppressant, changeover of crews and night-time operations. Even if the force was at the *critical* level – when the rate of extinguishment equals the rate of perimeter establishment – the fire perimeter already established has to be put out if the fire is to be put out.

3. Too many individual assets to protect

The suppression capacity needed to extinguish the fire edge at a rate faster than it grows may be present, but some or all of this capacity may have to be redeployed to protect significant spot or edge assets (e.g. lives and/or property). Equipment and crews may have to stay with each asset (say, houses and other buildings) for a period much longer than that required to extinguish the perimeter of the fire adjacent to the asset. This topic could be considered as part of item two, above, but it is separated here to draw attention to its importance.

4. Too many fires

If too many fires occur or are reported (false alarms), all resources may be mobilised to the fires that were called in first so that no suppression capacity exists to respond to further calls. Standard procedure may be to despatch a certain number of appliances, depending on the predicted weather conditions for the day. If the number of fires times the automatic despatch quantity reaches the total number of appliances available, then capacity is reached and any further demand will be unmet. Capacity may be reached when all units are fully engaged on one or more large fires, or many small ones. It is also possible, during the mop-up phase, that rekindling events occur at a rate faster than outbreaks can be put out. Outside sources may be fully utilised and unavailable to assist. The greater the number of fires, the greater the length of the perimeter for the same area burnt. Too many fires may be due to too many spot fires on occasion.

Pre-suppression measures

Pre-suppression measures include the creation of tracks, the modification of fuel throughout a reserve and the establishment and operation of fire-detection systems. In chapter two, the following hypotheses (or assertions) are addressed in a discussion on the environmental effects of tracks, and their efficacy for fire management:

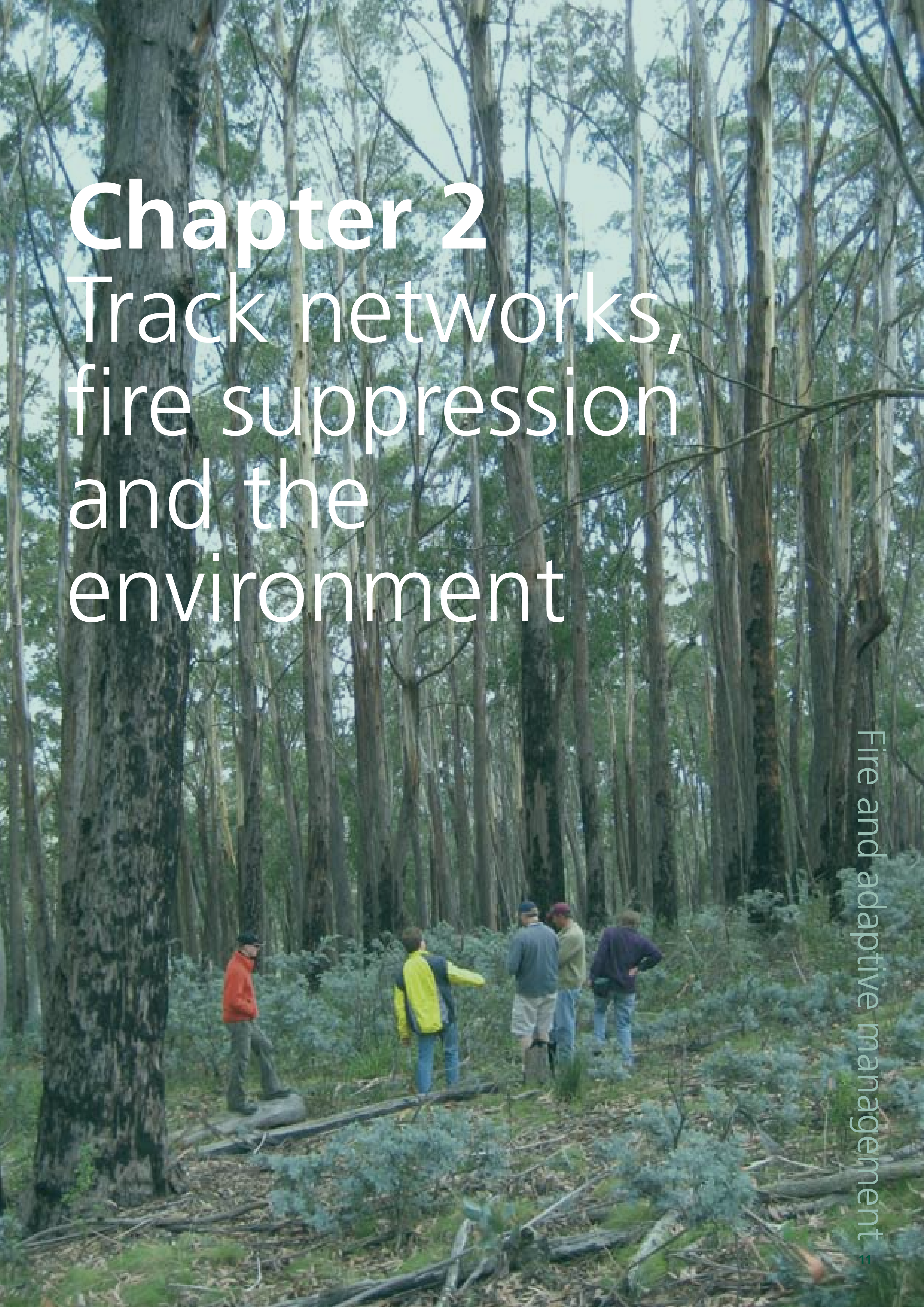
1. Firefighter access to all parts of a reserve, using tracks, is essential
2. A track around the perimeter of a fire is all that is needed to contain it
3. No track is necessary if aerial firefighting is adopted.

In chapters three and four, the issues arising from fuel modification by burning or grazing are considered.

Summary

Fire management may be seen as merely the conduct of fire-suppression operations, or it may be seen in a wider context that also includes fuel modification. There are limits to the success of fire suppression, but these limits may be reduced through fuel modification, including the creation of track networks. Fire management is part of a wider context of management for particular objectives in particular areas. The context here is that of public reserves for biodiversity conservation, particularly in south-eastern Australia.

Management may be seen as a *wheel* that needs all its components intact if it is to operate smoothly. Appropriate aims and objectives depend on the issue at hand; inventory and survey support the setting of objectives and the planning of suitable interventions (operations) in the landscape; and the assessment of the biological assets of the reserve while monitoring significant drivers, such as fire regimes, is essential for the evolutionary development of effective management systems.



Chapter 2
Track networks, fire suppression and the environment

Chapter 2 Track networks, fire suppression and the environment

Introduction

In chapter one, reasons why active fire management is needed were outlined. One of the reasons for intervention is the need for fire suppression. In this chapter, fire suppression from the ground, using tracks for access, is considered. The role of aerial attack is considered later in the publication. An examination of the theory of tracks, from a fire-management perspective, is conducted and a consideration of the effects of tracks on the environment generally is explored.

Tracks can resist the spread of some fires even in the absence of suppression crews and equipment, but their effectiveness can be improved by modifying adjacent fuel arrays so that any fire that occurs has minimal intensity. Tracks can be connected to natural fuel breaks, such as rivers, thereby expanding the network. Tracks may be in place at the time of a fire, but the firefighting operation itself may create new tracks using bulldozers, graders and hand tools (such as rake-hoes).

The length and width of tracks necessary for suppression to be effective is often debated (Andrews 1990), especially after major fires. This is reflected in the reports of official inquiries into major fire events (e.g. Esplin *et al*. 2003, p. 49; McLeod 2003, pp. 94–100; Nairn 2003, pp. 31–40, and in Organ's accompanying dissenting report, *Countering the case for more fire trails*). Ellis *et al.* (2004, p. 255) noted that track access has been a consistent theme of inquiries since 1939. Usually, communities affected by fires call for more and wider tracks, but these calls come with no quantitative suggestions. Nairn (2003, p. 27) recommended that the subject be investigated.

The potential environmental impact of track networks is discussed in a later section of this chapter, as a prelude to the discussion of the theory of track networks. Impact can vary widely according to ecological context. For example, some soils are more erodible than others. Modifications to the environment, caused by track networks, can affect biodiversity directly or through the modification of habitat. Some organisms may be favoured by the presence of the track network; others may be adversely affected.

Track establishment and maintenance usually involves a wider swathe than that of the final track surface as verges are created (Goosem 2004). Grading, slashing and other actions, such as cutting, mowing or spraying of herbicides may occur (Andrews 1990), thereby tending to maintain disturbed habitat. Informal rest areas and temporary dumps of road-maintenance materials may occur on verges. In forests, removal of trees on verges may be favourable to some weeds because of the disturbance that accompanies the process and because of the enhanced growing conditions. Tree removal along roads and major tracks, up to about one-tree height from the forest edge, was a feature of works carried out after severe fires in parts of the ACT following the January 2003 fires. Large trees, leaning trees, hollowed trees and dead trees are likely to be targeted in such programs due to the chance of their falling on cars or people, or blocking roads.

Establishment of tracks and modifying the adjacent fuels can affect biodiversity directly or indirectly, as reported later in the chapter. Firstly, however, a note of track types is presented.

Track types

Tracks usually take the form of paved roads, formed earth roads and rough tracks suitable for vehicular access during fires. At various times they may be called fire trails, containment lines, fuel breaks, control lines or access routes, according to the circumstances. In this discussion, tracks (*cf*. rivers etc) are formed directly or indirectly to support wheeled fire-suppression vehicles. Note that buffers (fuel-reduced strips rather than bare-earth breaks) are not considered to be tracks for the purpose of this discussion. These will be considered later. Areas under powerlines may generally be thought to be buffers and free of tracks, but 'all utility corridors have access roads', according to Andrews (1990), although they may be very primitive in places.

To define tracks in the present context, the following classification is presented:

1. Permanent[1]
    - Float road – for a low loader carrying a large bulldozer, maximum grade 12˚, >5 m wide with passing bays >7 m wide and 20 m long every 250 m, or where topography allows.
    - Tanker road – accessible to a heavy tanker and tipper truck carrying a small bulldozer, maximum grade 15˚, >4 m wide.
    - Light-unit track – for a 4WD tray-back utility carrying a small tank of water and a pump, >3.5 m wide.
    - Mineral earth fuel break (usually incorporating a track) – in the semi-arid zone of New South Wales this may be prescribed to be 15 m wide (New South Wales National Parks and Wildlife Service 2003).
2. Temporary
    - Temporary earth tracks or dormant tracks – closed to traffic except during fires, can be reopened and reused very quickly if prepared to a high standard (may need a grader to smooth the surface and remove litter and small plants, or heavy equipment to remove any earth barriers or large rock emplacements at entrances and exits).
    - Temporary earth tracks or fuel breaks – created during fire events and not maintained afterwards, may vary from narrow rake-hoe lines (1–2 m wide) to containment lines (*syn*. fire lines, control lines or tactical fire trails, Nairn 2003, p. 34) put in by bulldozers or graders (metres wide); may be converted to dormant tracks, as above.
    - Temporary cross-country vehicular tracks – not a formed track, created during mop-up (the operations involved in extinguishing smouldering stumps, fence posts etc) or vehicular access to the fire edge; may or may not rehabilitate and return to pre-disturbance state. These tracks are not considered further here.
3. Pseudo-tracks – bare-earth graded gutters to capture water and feed it into dams (earthen tanks) for domestic, stock or firefighting use.
4. Ephemeral wet lines – created by a tanker or from aerial appliances, volatile tracks about 6 m (medium helicopter) to 10 m (fixed-wing bomber) effective width (N Ryan [Victorian Department of Sustainability and Environment] pers. comm., 2004). These will be considered below, in the context of track effects and aerial firefighting.

Thus tracks are either permanent, temporary (can be found and re-cleared) or ephemeral (like wet lines). They have a range of characteristics, including various widths of bare earth and verge, surface qualities and drainage works. Tracks may change their classification over time, through rehabilitation processes and restructuring.

Track characteristics depend, to some extent, on the means of construction. A D4 (small) bulldozer will cut a swathe about 2.5 m wide with one pass, while a D6 (large) bulldozer will cut a swathe about 3 m wide (D Broderick, contractor, pers. comm., 2004). A grader will scrape a path about 3 m wide with each pass also.

Rates of track construction vary widely according to terrain and fuel. Dozers in forests can achieve 100–900 m hr^{-1}; the larger dozers moving faster and being capable of moving larger debris (McCarthy *et al*. 2003). For comparison, people with hand tools, such as a rake-hoe, will construct a trail about 0.5–1 m wide (Gould 2004) at a rate of the order of 10–20 m $person^{-1}$ hr^{-1} (McCarthy *et al*. 2003). As crew numbers are usually between two and seven, an approximate crew rate of construction is of the order of 100 m hr^{-1}. Rates of wet-line construction are considered later.

1 Mr Simon Hemer's assistance with this section is gratefully acknowledged.

For completion, the concept of the buffer (Gill 1986) is introduced. A buffer is a fuel-reduced strip designed to modify fire behaviour to enhance the chance of direct suppression success. This is not a track in the sense that it is not scraped, bare earth is not created and vegetative regrowth may be expected to begin immediately after its creation. Thus buffers have temporary value. Their usefulness for fire suppression is likely to be greatest when they encompass tracks or fill the area between parallel tracks (Plate 2.1). Buffers may be created by burning, slashing or other methods (see the section on Track Widths, below).

Plate 2.1 Road and track networks: parallel tracks placed to enable between-track burning for the creation of a buffer strip on the coastal plain north of Perth, Western Australia (Gill 1985).

Effects of tracks on the environment

Stating that tracks have no effect on the environment is a tautology because the act of their creation is in itself an effect. The question, rather, is the extent to which their wider effects – compared with merely causing a strip of bare soil – occur in different landscapes.

Tracks and roads have effects on the physical and biological environment and this attains greatest significance in conservation areas. Coyne (2001) concluded, from a review of threats to animal species in the Australian Alps' National Parks, that '10 species were threatened by roads or road construction'. Trombulak and Frissell (2000) detail a number of examples of road effects on populations of native animals in the Northern Hemisphere. While this review is a useful general reference it has a strong emphasis on the Northern Hemisphere, where effects due to road-salting practices, for example, may be expected to have greater significance than in south-eastern Australia.

The effects of tracks include the following.

1. **Tracks affect drainage, erosion and sedimentation**. As Tromulak and Frissell (2000) state: 'roads directly change the hydrology of slopes and stream channels'. These effects occur in three main ways. The first is associated with keeping the tracks dry by mounding and compressing them while creating spoon drains and mitre drains alongside. Consequently, these actions increase local run-off and the channelling of water to areas not previously so endowed, thereby increasing scouring capacity. The second is by the creation of blockages to the normal flow of cross-track drainages, sometimes creating swamps and thus sinks for sediment. From time to time this may divert drainage into new channels where scouring can occur. Finally, general erosion of unsealed

surfaces can cause sedimentation beyond mitre drains onto previously unaffected land. Thus the effects of tracks can extend into catchments generally. The following is an example, 'Roadside erosion has occurred in all tracts of the Monaro [New South Wales], and at alpine and subalpine elevations many fens and bogs with their associated peats have undergone desiccation' (Costin 1954). Eroding tracks can lead to water turbidity in streams, thereby decreasing water quality. Cornish (2001), in a commercial forest area, found that the 'construction and use of permanent roads resulted in increased turbidity levels, but these increases only persisted in the catchment containing stream crossings'. Contour ditches and culverts can support the temporary ponding of water. These mini-storages can supply native and feral animals with added water, and thereby help augment their populations artificially. Changed drainage and sedimentation patterns can affect the habitats of plants and animals.

2. **Tracks change environments and thereby create opportunities for invasion and persistence of weeds** (e.g. Tromblak and Frissell 2000; Goosem 2004). Compaction (Caling and Adams 1999), soil disturbance and sedimentation create new habitat that is temporarily free of competition, and this condition is more favourable to the establishment of horticultural and other weeds. The level of manifestation of weeds will vary according to history and circumstances, and be affected by the size and composition of the local soil-seed pool (Plate 2.2).
3. **Tracks can directly destroy habitat of native plants and animals.** The significance of this effect for conservation in a reserve depends on the extent of the habitat concerned. Small rare plants and animals are likely to be the most affected. Andrews (1990) noted the case of a power-line easement just missing the habitat of Australia's rarest butterfly.
4. **Tracks elicit vehicular access and its consequences.** Vehicular access allows the spread of weeds via various types of machinery – from cars (Wace 1977) to slashers to graders. For example, there is an Indian grass naturalised in northern Australia called Grader Grass (*Themeda quadrivalvis*) because of its prevalence along roadsides after grading (Pitt 1998). Roads may prepare the way for the invasion of formerly natural habitat (Gelbard and Belnap 2003). The sowing of grasses and the placement of hay bales to slow or prevent erosion in spoon drains may allow weeds to enter an area through the materials used, or indirectly on the machinery used for the operation. Fungal disease, such as *Phytophthora* spp., may spread along roads and tracks (see Adam 1995; Trombulak and Frissell 2000; Garkalis *et al*. 2004). Tracks may, if open to the public, allow better access to picnickers, arsonists and maintenance workers, all of whom may cause unwanted ignitions by carelessness or design. Rubbish may also be dumped along tracks (Nairn 2003, p. 34). *Wildness* values may be diminished also. Vehicular access leads to mortality of any 'sessile or slow

Plate 2.2 The native grass *Themeda australis* dominates on the right-hand side, while exotic species dominate the left, where a new fuel break had been cut through and allowed to revegetate in this native grassland in the ACT (Gill 2005).

moving organism in the path of the road' during construction, as well as mortality of animals due to collision with vehicles (Trombulak and Frissell 2000). This can be the cause of the demise of significant proportions of a wildlife population (Bennett 1991; Taylor and Goldingay 2004).

5. **Roads, and associated works, such as culverts and drains, may become barriers to the movements of animals** (Bennett 1991; Goosem 2004) *and affect the distribution of animals* (Trombulak and Frissell 2000). One of the examples in Trombulak and Frissell (2000) is from Reh and Seitz (1990) who found that the barriers created by roads led to genetic differentiation in a common German frog.
6. **Tracks can affect levels of predation.** Tracks may show many signs of feral animals, such as foxes (*Vulpes vulpes*) and wild horses (*Equus caballus*), but it is unknown whether tracks always enhance the presence of feral animals. Feral animals can be predators, such as cats (*Felis catus*) and foxes and where dense cover is necessary for the protection and survival of native prey, such as the Potoroo (*Potorus tridactylus)*, cutting tracks through the habitat, even for research trapping, can expose them (Claridge 1998). Bennett (1991) noted that lightly used roads are often frequented by native and introduced predators to the extent that roads are preferred to adjacent non-roaded areas; presumably roads allow relatively easy access to hunting areas.
7. **Tracks can affect ingress or egress of fires.** Tracks may act as barriers to the spread of fires into a reserve, thereby having the potential to indirectly affect the biodiversity of areas much greater than that of the track itself. Thus by creating artificial barriers to the spread of prescribed or unplanned fires, tracks can affect fire regimes. This may sometimes be positive in terms of biodiversity outcomes, but may also be adverse. By changing microclimate along edges, fire patterns may change. In fragmented Amazonian rainforests, for example, edge effects increased exposure to fire (Cochrane and Laurance 2002).
8. **Wet lines (water dropped from aircraft to suppress fires) may contain chemicals that have adverse environmental effects** (Adams and Simmonds 1999, Bell *et al*. 2005). In Australia, the study of the environmental effects of these agents is in its infancy (Adams and Simmonds 1999). However, effects may be apparent on species' composition of both plants (Bell *et al.* 2005) and animals through their chemical content. Nitrogen may affect legumes' establishment and change grazing patterns, while phosphorus can be toxic to some species of plants (Heddle and Specht 1975); weed invasion may be encouraged because of nutritionally richer soils; mortality of aquatic invertebrates can occur; and foliage death in plants has been noted. 'A summary of the data available suggests that there is a significant potential for damage to terrestrial vegetation from fire retardants, and to aquatic ecosystems from firefighting foams' (Adams and Simmonds 1999; also see Brown *et al.* 1998 for certain frog species). Plant and shoot death of *key species* has been recorded in experiments using a P-based retardant in Victorian heathlands (Bell *et al.* 2005).
9. **Roads and road users may alter environmental chemistry.** Road use may affect the chemistry of an area up to 200 m away and further if streams are affected (see Trombulak and Frissell 2000).

The effects of tracks will vary according to many factors, and not all of the changes mentioned above will be realised in all localities. The extent to which the variety of tracks causes the sorts of changes mentioned above is often unknown. However, changes will occur whenever a track is placed in a reserve. Trombulak and Frissell (2000) 'found support for the general conclusion that they [roads] are associated with negative effects on biotic integrity in both terrestrial and aquatic ecosystems'. When contemplating the introduction of a new track, the manager might consider: how long the effects of this decision will impact on the local environment – 1, 10, 100 or 1000 years; and to what extent restoration can occur if the decision is reversed.

Based on the possible effects of tracks and track networks mentioned above, in general, it would enhance the chances of successful biodiversity conservation in reserves if track lengths and widths and associated verge treatments were minimised (Plate 2.3). But what of the need to suppress and control fires? Is a trade-off necessary? How important are tracks for fire management, let alone other operations, such as weed and feral animal control and built-asset maintenance?

Plate 2.3 Track through mallee vegetation associated with water-supply pipeline and power line, and associated verge treatment. The photo was taken eight months after the Tulka Fire near Port Lincoln, South Australia (Gill 2001).

The following discussion is centred on the role of tracks in assisting the control of fires, whether prescribed or unplanned. In this context, it is important to clarify what the tracks are for. From a fire management point of view, tracks mark out blocks for prescribed burning and provide edges against which to burn. Tracks also provide access and a place for firefighting where retreat is easier. Tracks do not always stop a fire, but may increase the chances of stopping one. A fire may be contained within an area surrounded by tracks.

Here, the approach of arguing from various premises is adopted and the implications of these are followed in simple pen-and-paper models. This approach enables an appreciation of the different factors at work in what may appear to be a simple problem – do more tracks plus better suppression equal smaller fires and better results (while avoiding issues associated with particular localities)? Even so, in pursuing the implications of these models an attempt is made to keep an eye on reality. Track properties important in stopping a fire (treated simply as width or effective width) and data on real track networks are outlined, along with some aspects of real suppression operations and their limitations.

Track width: how wide should tracks be?

Introduction

Track width is an important variable in limiting the spread of prescribed or unplanned fires, either alone or with the aid of direct suppression. Track width may enhance the safe burning out of fuel between a track and an unplanned fire. It may also enhance firefighter safety as track width, including a provision for turning or passing bays, affects the safe passage of fire vehicles.

Fires cross fuel breaks, such as tracks, in two main ways (Wilson 1988): by flame contact and through fire brands. Another possibility, flame radiation, is downplayed as a mechanism here, but it cannot be eliminated in all circumstances, particularly during crown fires. If the radiation is sufficient to ignite the fuel across a track, the flame that produces it is probably in near-fuel contact anyway. If this is not so, yet flame radiation is sufficient to ignite the fuel, the flames are likely to be tall and turbulent and spotting (causing fires downwind; Plate 1.1), which is likely to be quite sufficient for fire to cross the track. Thus to stop a fire passively (i.e. without active suppression), a bare-earth break must be wider than the flame stretch, or the reach of fire brands. Fire brands may travel hundreds of metres to tens of kilometres (Text Box 1.1), while flames can reach lengths of tens of metres in some thicket and forest situations[2]. Thus whole landscapes would have to be bare, green, moist or sealed to be sure that no fire brand could cross them – an absurdity from the point of view of cost, conservation and land degradation.

A plan view of the fuel break situation can be illustrated. Figure 2.1 is a simple model of a study area of length L units and a perimeter track of width w units. The total area of the study site, A_T, is L^2 units while the area of the track, A_t, is $4w(L\text{-}2w) + 4w^2$. When w increases to the limit, it occupies the whole area: A_t/A_T equals 1 when w equals $L/2$.

$A_t/A_T = [4w(L\text{-}2w) + 4w^2]/ L^2$.

The situation becomes more complex when there are multiple cells in the study area and shared sides become usual.

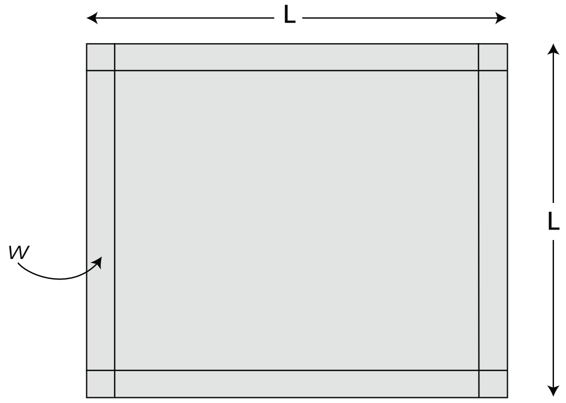


Figure 2.1 Diagram of a simple square study site with a side length of *L* units and a perimeter track of width *w* units. If w increased to its maximum value, it would occupy the entire site.

Tracks, even of the same width, could not be expected to have the same effectiveness in any given fire. For example, flames spreading before the wind (heading fires), will be greater in length than those travelling against the wind (backing fires), under the same conditions. Thus the chance of the flames crossing a barrier, such as a track, will be greater for a heading fire. Also, flames will be greater in length when fires are travelling uphill. Of course, the greater the severity of fire weather the greater the expected length of flames and the greater the chance of spot fires being generated.

If the effective width of tracks can be enhanced using various techniques that fall short of soil disturbance, then we might expect that some of the detrimental effects of track construction and maintenance can be avoided while increasing the value of the track and track edge as an impediment to fire spread.

The effectiveness of tracks during fires can be increased by edge treatments. Here, these edge treatments are termed *buffers* (Gill 1986), as above, and may be created by: (i) changing vegetation type permanently (e.g. by removal of trees or thickets); or (ii) modifying the vegetation or understorey temporarily by chaining (pulling down vegetation with very heavy chains dragged between two tractors; see Esplin *et al*. 2003, p. 90), slashing, mowing, pruning, herbiciding, chopping, grazing or burning. Buffer zones may contain stubs of woody stems – in mallee and other shrub lands, for example – that can cause puncturing of rubber tyres. Various methods of vegetation modification may be practised in combination, for example chaining and burning. Such treatments may be applied to an edge with (at various distances up to several hundred metres) or without a second bare-earth track or other fuel break parallel to it.

2 Flame height in a burning forest is usually measured as the height from the soil surface to the location of the uppermost part of the flame. When the wind is blowing strongly, flames are seen to arise from tree canopies as well as surface fuels, so flame lengths vary in their height of origin. In such cases, flame lengths may be conveniently considered as those from one end of the source (such as a canopy) to the flame tip.

Edge treatments, as well as the tracks themselves, have environmental repercussions (see verges, above). As the environmental effects of edge treatments are often unknown, a manager may decide to monitor and record any change to inform future management. The area affected by intervention can be large, especially when the edge treatment becomes broad-area treatment and extends to the whole reserve.

Track widths for grasslands and forests

How wide should a track be to stop a grassland fire?

Fire crews can create fuel breaks in the path of the fire or use established tracks, the main form of artificial fuel breaks. In forest fires, fuel breaks would be expected to be less effective than in grasslands, because fire brands capable of carrying a fire across breaks are more likely to be produced (Text Box 1.1 and below).

In the absence of lofted material capable of carrying fires across breaks, flame length becomes the most important variable. Flames driven by the wind, if long enough, will be able to bridge the barrier posed by the break and allow the fire to spread downwind. Wilson (1988) examined this proposition for grasslands. Flames travelling against the wind and leaning away from a break as they travel up to it are much less of an issue because their bridging capacity is limited.

Wilson's experiments were conducted in a grassland with fuels up to 1 m tall and fire intensities up to 17 MW m^{-1}, or 17000 kW m^{-1}. He found that the limit to the width of track, *w* metres that prevented a fire crossing it was given by: $w = 1.192\, I^{0.5}$ when fire intensity *I* was expressed in MW m^{-1}. Put another way, the intensity that will give a flame length just capable of crossing the break of a certain width is given by: $I = 0.7042\, w^2$. Wilson did not actually measure flame length, but the relationship used – derived by Nelson (1980) – was consistent with his results and the mechanism of flame bridging. While Wilson's experimental fires reached an intensity of 17 MW m^{-1}, and the maximum intensity of fire that bridged a break was 11 MW m^{-1} (Wilson 1988, Fig. 4), grassfire intensity can reach the order of 30 MW m^{-1} (Luke and McArthur 1978, p. 28). In Figure 2.2, the equations for the experimental data (up to 4 m fuel-break width and up to an intensity of 11 MW m^{-1}) are plotted, along with an extrapolation reaching to the anticipated maximum intensity of grassfires of 30 MW m^{-1}. The latter is, of course, speculative. As the critical width equals flame length in the diagram, the speculative end (above 4 m, Series 1) suggests that flame lengths occur up to seven or so metres in the range of perceived intensities for grassfires. The diagram is given merely to show trends and should not be taken as prescriptive.

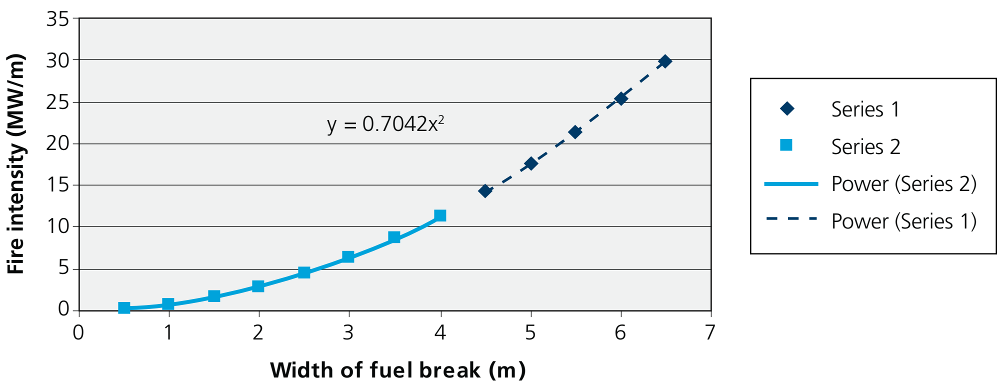


Figure 2.2 Intensity of a grassfire that can be stopped by a fuel break (Wilson 1988). Series 2 is a plot of Wilson's equation for his data domain. Series 1 plots the extension to Wilson's equation up to a possible maximum intensity for grassland fires – as given by Luke and McArthur (1978).

Fire and adaptive management

Wilson's work is particularly interesting in its implications. It suggests that by increasing break width from merely 3 m to 4 m, the intensity of a fire that can just be prevented from crossing the break, rises from 6.3 MW m^{-1} to 11.3 MW m^{-1} – a substantial rise. A fire intensity of 9 MW m^{-1} could arise from a fuel load of 5 t ha^{-1} (0.5 kg m^{-2}) and a rate of forward fire spread of 1 m sec^{-1} (3.6 km hr^{-1}), if the same heat yield as used by Wilson (18,000 kJ kg^{-1}) was adopted. Wilson's model suggests that with a modest increase in the width of a break, there is a large increase in the intensity of fire that can be passively controlled. This only applies in grasslands where no spot fire is likely.

The model discussed above is the simpler of the two that Wilson (1988) produced. The other much more complicated model gives the probability of spanning a break for fires of various intensities burning in grassland fuels with trees at various distances from the break. This model predicts that the approximate probability of bridging a 3 m wide break is 29% for a 2 MW m^{-1} (2,000 kW m^{-1}) fire. Trees near a break can markedly reduce the value of a break in stopping fires because of their propensity to produce lofted fire brands. Thus the effectiveness of a fuel break in forests is markedly reduced, compared with that in grasslands.

Flame length is harder to measure than break width: flames pulsate so length varies all the time, thereby making objective measurement difficult. What is actually taken as flame length is probably the average length of the longer flames, rather than the maximum lengths of flames (e.g. flares). If a photograph is taken, the variation can be observed. The measurement of flame height was discussed in detail by Gill *et al*. (1987*a*). The same principles apply to length. On the CSIRO Grassland Fire Spread Meter (CSIRO 1997) there is the comment that 'flashes of flame may extend to twice [given] values', the maximum height (not length) given there being 4.4 m in *natural pasture* with the very rapid spread rate of 20 km hr^{-1} (5.6 m sec^{-1}).

Although there is no published data on the matter of flame height or length as a function of grass height *and* density, the CSIRO Grassland Fire Spread Meter notes that a shift from natural pasture (generally more than 50 cm tall) to grazed pasture (generally less than 10 cm tall) will halve, or more than halve, flame height for fires travelling up to 20 km hr^{-1} (5.6 m sec^{-1}). Flame height is a curvilinear function of rate of spread of the fire (Cheney and Sullivan 1997, p. 25). Rate of spread drops approximately 20% for fires in *natural* pastures, compared to those under *normal* grazing, but much more in *eaten out* pastures (*ibid*, p. 39). Note that the values on the meter are considered to be a guide only (CSIRO 1997).

A logical consequence of Wilson's research is that attention needs to be given to the details of break construction and maintenance. If the width changes even a little, the consequences can be large. If the break is poorly maintained so that fuel cover is present to some extent, although slight, then the effective width of the break is reduced. A steep but relatively narrow grassy batter on the downhill side of a break can support quite a long flame, which may be well beyond that expected. Debris swept to the side of the break may be a weak link in terms of the effectiveness of the break when it is alight.

While Wilson's work remains valid for the conditions under which he worked, the person applying these findings to other situations may need to adapt them to local conditions. Indeed, Wilson himself pointed out that 'In southern Australia, firebreaks [i.e. fuel breaks] even wider than 10 m can be breached when winds are strong enough to transport surface debris such as smouldering animal manure' (Wilson 1988). The composition of the particular grassland and other fuels of concern can be important. Luke and McArthur (1978, p. 107) note that spot fires can be common up to 100 m ahead of the flame front in grassland fires, due to carriage of burning seed heads of thistles, *Phalaris* (a pasture grass) or wheat, for example. While this suggests that in such circumstances an effective break would need to be 100 m wide, verge treatments, such as slashing, may reduce the problem. Even so, Noble's (1991) case history of an extremely fast and intense grassfire in the Riverina of southern Australia is a cautionary one – the fire jumped a 54 m wide break. In northern Queensland woodlands, where grassy fuels predominate but trees are common, it is recommended that fuel breaks (see Plate 2.4) need to exceed one kilometre in width if they are to be effective in the late dry season when fires are at their most intense (Crowley *et al*. 2003).

Plate 2.4 A fuel break is not necessarily a firebreak. Sutton Road, New South Wales (Cutting 1985).

There are other sorts of grassland than the ones studied by Wilson (1988). Large portions of the vast arid zone of the Australian continent are covered with a hummock grass (Allan and Southgate 2002), each plant typically surrounded by bare ground. Such fuels also occur in north-western Victoria, where annual rainfalls are very low. Fires are spread by wind-blown flames bridging the gaps between hummocks (e.g. Gill *et al*. 1995), so the situation is a microcosm of the track-width and flame-bridging situation. When fuels are discontinuous, fires only spread with the wind, not against it. When there is no wind, the fire dies in such fuels. After an extended period of heavy rain, ephemeral grassy fuels may fill the gaps between hummock grasses, so that when the inter-hummock grasses die they make the fuel array continuous and fire behaviour is akin to that in mesic, temperate, continuous grasslands.

From this discussion, the general principle is that even small changes in track and verge width can change their effectiveness in stopping the spread of a grassfire. However, local factors, such as spotting potential, need to be considered even in grasslands. Consequently, the gathering and assessing of local flame-length and track-width data is important. Then modifications in fire-suppression protocols can be made in the spirit of evolutionary management.

Width of tracks needed as fuel breaks in forests

In the discussion on fires in grasslands, the fire characteristic of primary focus was flame length. When trees or other means of producing spot fires downwind of breaks enter the picture, attention shifts to the ability of the fire to produce lofted burning material as a major fire characteristic. This characteristic may reach an extreme in long-unburnt stringy-bark eucalypt forests where pieces of fibrous bark break away from burning trunks and branches and are carried away in the wind (e.g. Cheney and Bary 1969; Gould *et al*. 2007). Long strips of smooth gum bark may also break away from parent trees and ignite in fires (Cheney and Bary 1969), thereby potentially carrying the fire long distances downwind. I Dicker (pers. comm., 2005), while flying during the 2003 fires in Kosciuszko National Park, observed strips of smouldering ribbon bark to be common at about 150 m above ground, and at about 10 km from the fire front.

Cheney (1994) considered the head-fire intensity at which suppression was likely to fail during a fire in an open forest of stringy-barks that had reached 10 ha in area and in which suppression was being attempted by various means: for a hand-tool crew of seven, the fire intensity was 800 kW m^{-1}; for two D6 (large) bulldozers, it was 2000 kW m^{-1}; for one D6B air tanker, the intensity was 2500 kW m^{-1}; and for ground tankers on a 40 m wide break, the level was 3500 kW m^{-1}. This sequence can be taken to reflect effective break width under various fire intensities. Note that this is an interpretation of the data and does not reflect the original intent of its use. Hand tool tracks may be 0.5–1 m wide (Gould 2004), dozer tracks 3–5 m wide, aircraft tracks 20 m wide and the ground tanker break 40 m wide (as stated). If this is so, there needs to be a *large* increase in break width to attain a small increase in the upper threshold intensity for control (Figure 2.3): a doubling of the assumed width from 20 to 40 m led to only a 1 MW m^{-1} increase in intensity controlled. This is a contrasting situation to that in grasslands, where a *small* increase in break width gave effective protection against a large increase in fire intensity, according to Wilson (1988). The reason for this difference may be that in grasslands, as intensity increases the increase in flame length (i.e. proportional increase) decreases and remains within reasonable bounds. In highly spot-fire prone forests, as intensity increases spotting distance quickly exceeds any reasonable track width. Thus track width as a method of passive fire control is much less effective in forests than in grasslands.

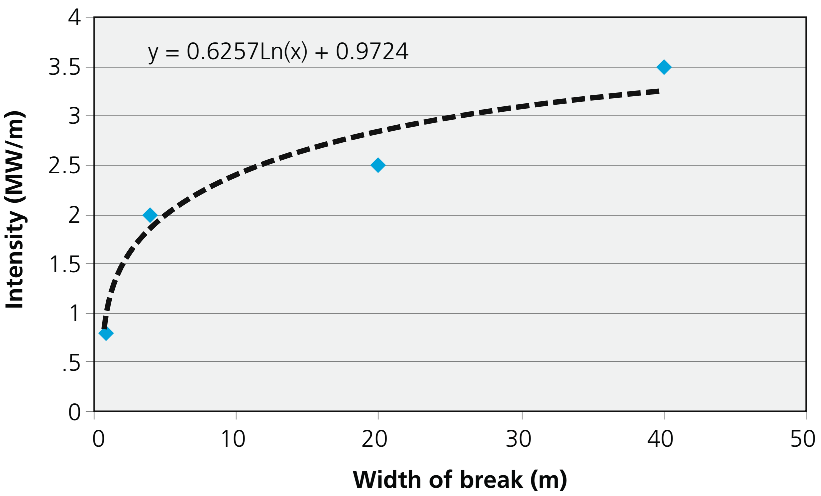


Figure 2.3 Intensity of a forest fire stopped by a widening fuel break with increases in suppression forces in a stringy-bark forest (interpreted from data in Cheney 1994). Notice the few data available in this interpretive diagram. This diagram may best be considered as an hypothesis developed using the best information available.

It is important to note another contrast with Figure 2.2. In the grassland case, the maximum intensity was assumed to be 30 MW m^{-1} (Luke and McArthur 1978). In forests, the maximum intensity is more likely to be 100 MW m^{-1} (Gill and Moore 1990). A 4 m wide break in forest may assist in the control of a 2 MW m^{-1} fire, but in grassland a similar break may be effective in holding an 11 MW m^{-1} fire. It should not be assumed that fires of the same intensity in grassland and forest have the same properties, as this is not the case.

Summary

The previous discussion highlights the factors that need to be taken into account in planning a network of strategic fuel breaks, including tracks.

For grasslands, the following need to be considered:

- Width (a small increase in width may give a large increase in controllable intensity when spot fires can be discounted)
- Maintenance (keeping tracks free of material that will carry a fire)
- Proximity of trees (keep tracks away from trees) and other materials potentially able to cause spot fires, such as thistle heads within 100 m or so of breaks
- Grass height and therefore potential flame length near breaks (also, other fuel factors that might affect flame bridging of breaks).

For forests, the following suggestions need to be considered:

- Increasing track width appears to be much less effective than in grasslands because of the spotting potential of forest fuels
- Maintenance of tracks – keeping tracks free of combustible material is important because burning-out procedures during suppression operations assume a clean break in the fuel
- Fuel treatments that reduce spotting potential are most important in the effectiveness of tracks.

The situation for fires burning with the wind in other plant communities may be considered to fall between the extremes considered here. For fires burning against the wind, a fuel break can be a potent inhibitor of spread.

Track width in grasslands can be a useful variable to consider in the light of possible fire characteristics, while in forests, track width is more likely to be a consideration of vehicular-access requirements, control of backing fires and prescribed burning activities.

What is done in the way of track construction, maintenance and vegetation treatment to enhance break effectiveness should be done in the context of the overall aims of management of the area. Thus action should vary according to whether the area is farmed, cropped or used as a water or biodiversity conservation area, for example.

Track density: how much track is enough?

Suppression operations can benefit from fuel breaks (tracks) in a number of ways: (i) provide passive breaks to fire spread within the limits of fire properties; (ii) provide trafficable access to the fire and water; (iii) provide a place from which fires can be fought and where rapid egress is possible; (iv) define boundaries within which a burnout of fuel between the fire perimeter and the break can be carried out during a suppression operation; and (v) provide a break against which back-burning – lighting an edge so that the fire created burns back to the nearby oncoming fire front – can take place (often a risky operation). Item (iv) is similar to item (v) in removing fuel between a fuel break and the fire, but distinguished by the proximity of the fire front – nearby in (v), but at a distance in (iv).

Before we delve into the issue of track density, it is instructive to review what densities of tracks occur in various places.

Track densities

The densities of a number of track networks in south-eastern Australia are shown in Table 2.1. The table shows that track density varies widely, probably increasing with proximity to built assets and tourist venues, but also affected by land use and terrain. There is a 100-fold increase in density from that of Yathong Nature Reserve in the Western Division of New South Wales, with a figure of 0.1 km km^{-2}, to the suburban Canberra figure of about 10.5 km km^{-2} .

Table 2.1 A sample of track densities. Track includes many forms of bare-earth features or paving suited to the carriage of firefighting vehicles, including roads. There is no guarantee that what one source calls a track or road is the same as another in a different data set.

Location	State or Territory	Land use	Sample area (km^2)	Track length (km)	Track density (km km^{-2})	Area (km^2) that contains 1 km of track	Reference
Albert River	Vic	Hardwood and softwood plantation	82	345	4.2	0.24	Takken and Croke (2004)
Booderie National Park	Jervis Bay Territory	National park	55	143 (38 km public roads, rest management trails)	2.6	0.385	Booderie National Park (pers. comm., 2004)
Canberra	ACT	South-western suburbs	0.25		c. 10.5	c. 0.095	Street directory
Cooleman Ridge	ACT	Conservation reserve, suburban edge	1.92	17	8.8	0.11	Environment ACT (pers. comm., 2004)
Cuttagee Creek	NSW	State forest, native spp.	38	75	2	0.5	Takken and Croke (2004)
General	NSW and Vic	Forestry, native			median 0.5; range 0 to 2.3		Croke (2004)
General	NSW and Vic	Forestry, plantation			Median 4.2; range 1.2 to 5.6		Croke (2004)
Namadgi National Park	ACT	Conservation reserve	1058	551	0.52	1.9	N. Lhuede (pers. comm., 2004)
NSW National Parks and Wildlife Service, Northern Branch Reserves	NSW	National parks and nature reserves	15 303	8393 (omitting dormant tracks)	0.55	1.8	S. Hemer (pers. comm., 2004)
North-eastern Victoria	Vic	Fire area 2003	9773	4435	0.45	2.2	Wareing and Flinn (2003, pp. 195 and 208)
Victoria	Vic	Public land	77 000	25 000	0.32	3.1	Auditor General, Victoria (2003, pp. 3 and 136)
Yathong and associated nature reserves, western NSW	NSW	Nature reserve	2472	250	0.10	9.9	NSW National Parks and Wildlife Service (2003)

Fire lines, containment lines, fire control lines and suppression lines

Few figures are available for the densities of lines established for the suppression of fire. The preferred term used here is suppression lines – to avoid the impression that the established line will guarantee the containment or control of any fire that encounters it. Lines may be bare-earth breaks (fuel breaks), or lines laid down by aircraft as water, with or without added chemicals, and referred to as *wet lines*. In this section, discussion is exclusive to the former type. Data indicates that densities of suppression lines vary widely:

- 1997 – Mount Martha Park, Victoria: 7.5 km on 0.36 km^2 (36 ha) or 1 km line per 0.05 km^2 – 21 km km^{-2} (Caling and Adams 1999)
- 1998 – Caledonia fire, Alpine National Park, Victoria: 358 km fire line, 312 km^2 burned area, thus *ca*. 1 km line per 0.9 km^2 burned area – 1.1 km km^{-2} (Caling and Adams 1999)
- 2003 – North-eastern Victorian fires, Victoria. *ca*. 7000 km of fire control lines requiring rehabilitation[3] on public land and 2000 km on private land (Wareing and Flinn 2003 p. 211) – areas of 9773 km^2 and 900 km^2 respectively (*ibid*. p. 195). These figures give an approximate density of tracks of 1 km per 1.4 km^2 and 1 km per 0.45 km^2 respectively or, alternatively, 0.7 km km^{-2} and 2.2 km km^{-2}.

Densities can be high and may include regular tracks.

A common question is 'Should there be more tracks?' A common assertion is that there indeed should be more tracks; however, what sort of track network is ideal?

Model track networks: access all areas?

In this section a hypothetical, featureless square study area is invented, upon which we will establish a track network. This exploration of networks is used to illustrate principles in the simplest of all possible terrains. It is far from reality, but the principles are better demonstrated in this way, rather than using a set of track networks in real landscapes in which so many other variables intrude. The variables affecting networks in real landscapes will be considered later. Each area marked out by lines in Figure 2.4 is called a cell, while all cells combined form the study area.

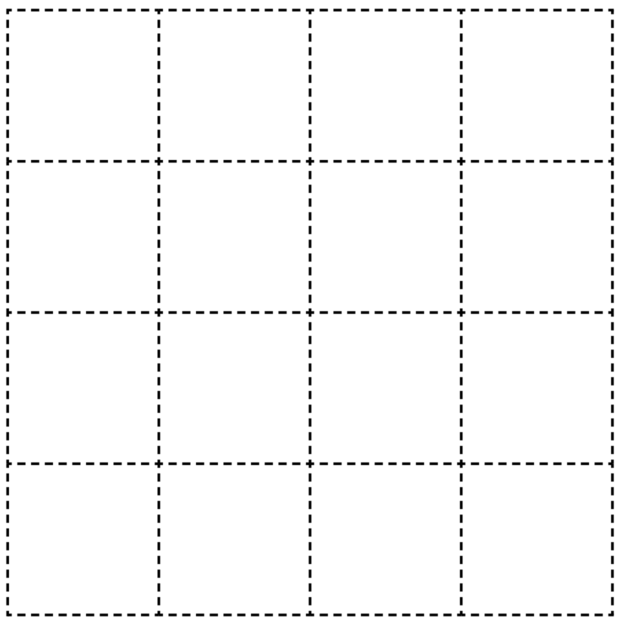


Figure 2.4 The hypothetical study area. The intersections of the grid are considered to be the points to which track access is desired. Each area marked out by lines is called a cell. There is no attempt to define scale.

3 *Rehabilitation* sometimes means closure of the track and application of revegetation and drainage works. At other times it means works are carried out alongside tracks that were widened at the time of the fire, but the tracks themselves remain open.

The first proposition examined considers that access to all points (corners and intersections of lines) in the study area (Figure 2.4) is necessary for effective fire control – a reliance on aggressively and quickly getting on top of the fire and putting it out. Later, suggestions will be made about *aggression* being directed to the fire rather than the land – minimum impact suppression tactics or M.I.S.T. (Mohr 1989) – including the perceived use of aerial instead of ground suppression. In this example, access to all points at minimum cost is desired; the usual expectation and the one to which attention is first directed. The example quickly shows the inadequacy of considering minimum length alone, and the importance of other variables.

In Figure 2.5, a sample square cell from the study area (Figure 2.4) is used to illustrate ways in which access to all points could be reached. Which has the minimum length and, by implication, minimum cost? A unit of track length is set as the length of one side of an individual cell.

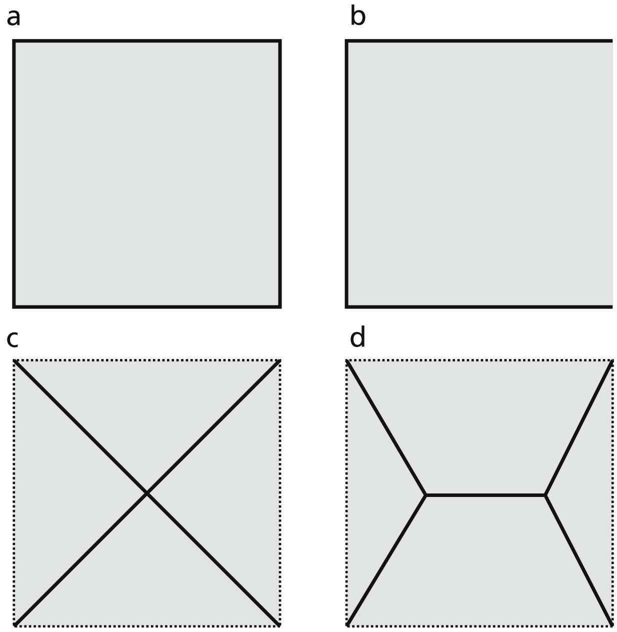


Figure 2.5 Four options for tracks reaching all corners of a sample cell of dimensions *L* by *L* units within a square-grid study site: *(a)* perimeter track has length 4*L*; *(b)* all points are reached with a track length of 3*L*; *(c)* tracks totalling 2.83*L* cut across the cell – 94% the length of the track in *(b)*; *(d)* sketch of the minimum length of tracks reaching all points (Stewart 1997, pp. 287*ff*), 2.73*L* – 91% of the length of track in *(b).*

Notice the savings in length as one goes from Figure 2.5 *(a)* to *(b)* – perimeter tracks – to *(c)* and *(d)* – a *spanning tree* and *Steiner tree*, respectively (Stewart 1997, pp. 287*ff*). The minimum length is *(d)*, the Steiner tree, where the angle made by the intersecting arms of the tracks is 120^{o} (Brazil *et al*. 1997; Stewart 1997, p. 287*ff*). While example *(d)* shows a saving of only 9% over *(b)*, the value of this saving increases as the number of cells increases.

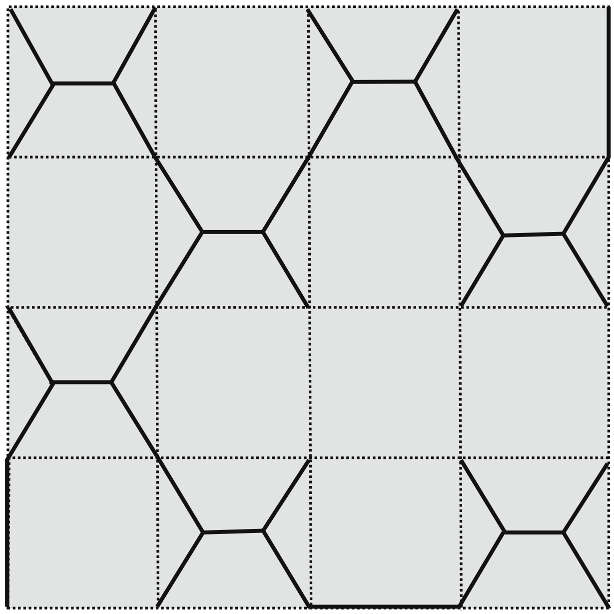


Figure 2.6 Sketch of a track network aimed at reaching all points (i.e. complete access) with minimum length – based on the concept of the minimal Steiner tree for a rectangular array (Brazil *et al.* 1997; Stewart 1997, p. 287ff). While the track length is relatively short, about 55% of the length if all sides of squares were used, there are 16 dead ends from a possible maximum of 25.

Figure 2.6 is a sketch of a connected network that attempts to attain access to all points (corners and intersections of dotted grid) at minimum connected length. There are alternative configurations using the same unit shapes. While achieving a minimum length of about 21 units, the network is completely impractical for at least four reasons:

1. There are 16 dead ends that represent potential traps for firefighters.
2. The number of intersections poses a potential danger for users and adds to cost. Intersections slow traffic down and it could be argued that maximum speed is desirable for maximum access.
3. Transport within the network would be extremely inefficient. Consider travel from the top right-hand corner to the bottom right-hand corner. The shortest route would be the most direct route consisting of the four unit lengths of the right-hand cells. With the illustrated network, one would have to travel almost right across the network and then back again along a track that wanders to and fro. If this was a real network it could represent unsafe conditions, not only because of distance, but because of the possibility of having to double back across the path of an active fire.
4. This network lacks boundaries to the overall site, so it has little benefit for suppression operations involving the burning out of fuels (by burning under mild conditions during the fire event e.g. at night), or of the confinement of fires to a single cell or group of cells, let alone the entire area. Therefore, a perimeter track around the grid is considered necessary to meet our objectives. Another reason for the circumferential track is a common-law reason – managers will not only want to retain any fire within their area, but also keep those fires originating in neighbouring properties from entering their area, the grid in our model.

Having started from the simple consideration of the minimum track length (assumed minimum cost) to attain full access, three new criteria for the ideal network have emerged. They are: (i) safety of firefighters (through avoidance of entrapment and provision of direct routes to safety); (ii) provision of enclosed cells or groups of cells against which to burn, contain or repel a fire; and (iii) trafficability. Thus the minimum-length option of a track network for a model system, although providing access to all points, is seriously insufficient in any consideration of firefighting needs.

Compartmentalised track networks

Figure 2.7 illustrates the most comprehensive compartmentalised model track network possible, in which all points are visited to satisfy access requirements. The relative danger to hypothetical firefighters is set to nil as there are alternate routes away from every point (intersections of two or more lines on the grid). Compartmentalisation is complete. However, the relative cost as indicated from edge-length alone is at a maximum – 40*L* – almost double the minimum of about 21*L* of Figure 2.6; thus all other networks have a cost between these two values.

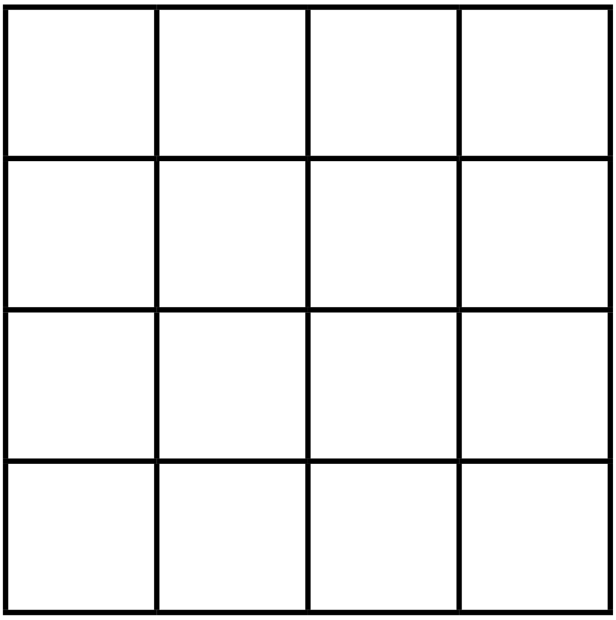


Figure 2.7 The grid of 16 square cells with the number of tracks (solid lines) needed to define every cell. In this case there are 40 sides to the cells (some shared), so the total length is 40*L*. The number of dead ends to tracks among the total of 25 corners and intersections is nil. There are numerous intersections.

Can we reduce the track length and retain or improve safety, while retaining groups of cells (cell clusters), all with perimeter tracks? It can be declared that reaching the perimeter track implies safety in this model system. If this were to be more realistic, the model system would have to expand well beyond the area of jurisdiction and consider the tracks and roads surrounding the study area. There are numerous alternative track designs for grids with a perimeter track but only two more follow.

In Figure 2.8*a*, all the tracks lead directly to the perimeter edge. The whole network is compartmentalised, there is no dead end and the track length is 28*L* – instead of 21*L* in Figure 2.6, or 40 in Figure 2.7. There are six exits to the perimeter. In Figure 2.8*b*, track length has increased by 4*L*, but there are more exits to the perimeter. There are also eight cell clusters rather than 4. This may be regarded as safer for firefighters than Figure 4*a* as there are options for travelling in contrasting directions from the interior to the perimeter, according to the direction from which the fire is approaching. The second diagram implies that fires could be confined to cell clusters, and because there are more of these clusters kept smaller on average. Thus these simple models suggest that there are cost implications with length of track, and trade-offs between access, firefighter safety and potential for confining fires to a group of cells.

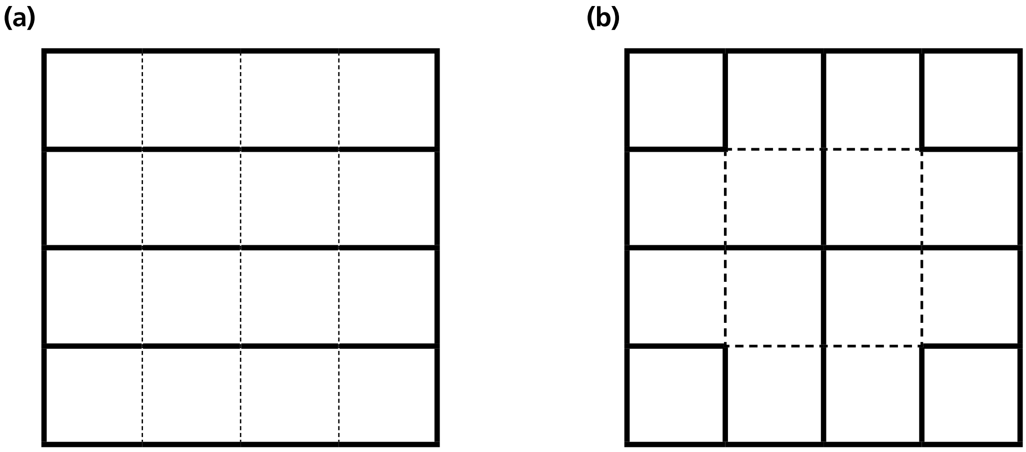


Figure 2.8 In network (a) access is provided to all points, there is no dead end and track length is 28*L*. There are six exits to the perimeter track from the interior and no dead end. There are four cell clusters. In (b) the track length is a relatively high 32*L*, but there are 12 exits to the perimeter and eight compartments.

There are many ways of organising track networks, even in these simple examples. In real life the terrain is almost never flat, there are natural breaks in the fuel (in the form of rivers and other drainages), there are terrain and other limitations to installation of tracks (e.g. cliffs and dune fields), there are tracks and roads present that were inherited from pre-reserve periods and there are reasons other than fires for establishing roads (e.g. to access viewing points and other attractions).

Declaring that there should be more tracks, as is a common response after major fires, begs the question, 'What should the length of the track network be for firefighting purposes in a reserve?' There is no logical answer to this question, because the question is incomplete and the ultimate answer, for perfect fire control sense, is one that creates bare earth over the entire area.

Seeking the most appropriate question to ask is often difficult. Perhaps the question that needs to be addressed is, 'What is the best track network that allows for effective fire management and complete firefighter safety within reasonable budgets and allows the purposes for which the reserve was created to be achieved?'

Reality involves heterogeneous terrain, varying fuel types, an inherited track network, limited budgets, varying distances to safe havens, varying internal travel options and travel conditions, varying access to water and varying effects of tracks on aims of management other than firefighting. Tracks may have different vulnerabilities during fires e.g. wooden bridges spanning them may be burnt or rock falls may occur in steep terrain. The environmental implications of track networks can be serious in terms of biodiversity outcomes, the extent of the problems introduced by them being dependent on the actual landscape of concern, its attributes and designated aims of land use. The art of management is in the implementation of *strategic* plans that take into account, among many other things, the likely variation in fire behaviour and likely fire effects in the designated landscape.

So far we have considered track networks in light of a largely passive control of fires, while recognising that firefighting practices may depend on track networks for the burning out of fuels and for gaining access to the fire. We have not yet considered the actual sizes of cells or cell clusters – our simple models are scale free – but this is done now.

Sizes and shapes of areas delimited by track networks

In the previous section a number of observations were made about track networks and the principles that may be applied to them. In this section the sizes, densities and shapes of areas defined by track networks are considered in relation to fire suppression. Here, areas are surrounded by tracks and are usually quite irregular in shape, compared with those in Figure 2.4.

Consider the areas defined by perimeter tracks. They could be:

1. **Very small so that fires never attain properties that allow them to spread across the encircling tracks, even on a day of extreme fire weather** – Fires start from a zero rate of spread and accelerate to a quasi-steady state for rates of spread, flame lengths etc. The quasi-steady state to stop them without active intervention in the form of burning out of fuels or the application of water and/or retardants may be well above that for perimeter tracks. Could fires be confined to areas in which intensities were never uncontrollable? Using this criterion alone, areas would likely be very small, unless fuel loads were always light and weather never extreme.
2. **Small and grouped where there is a history of a high density of ignitions (by lightning or people), close proximity to economic assets or greater chance of effective suppression** – Ignitions, proximity to economic assets and ease of suppression are not randomly located on landscapes. In areas where ignitions are more common, assets are denser, and ease of suppression is highest, a *group* of smaller areas might be contemplated. This group of areas could represent a zone in which management would be more intensive with more track options if one failed. Some areas that receive high numbers of lightning ignitions, however, may be steep and unsuitable for high-density track networks. They may also be *wild* areas where tracks are to be avoided, if possible.
3. **Large enough to contain fires within the first day** – If areas surrounded by tracks are defined in such a way that the main run of a fire (during a long summer afternoon on the day of ignition) is likely to run its course before reaching a break where it can be contained, then the minimum length of track for the greatest effect might be obtained. There are a number of problems with this idea. Variables to consider include fuel type, ignition location and topography. It may be unknown just how large a fire can grow in a day. If a fire was travelling at an average of 1m sec^{-1} (3.6 km hr^{-1}), for example, then in an eight hour run it would travel nearly 30 km (28.8 km). If the width was about one-third of the length, and the fire elliptical in shape, then the area burnt on this day would be about 22 000 ha or 220 km^2. Obviously, if the fire conveniently just fitted the area blocked out by tracks it would be nearly 30 km long by 10 km wide, aligned in the direction of the prevailing wind. A rectangular block just containing the fire would be nearly 30 000ha in area. If this was the size needed on a *bad* day, then what would happen on the more common days when fires of this magnitude would be most unlikely? Either the fire would have to be put out within the area, large areas of fuel would be deliberately burned out to avoid putting in tracks during an event or smaller blocks would have to prevail.
4. **Of a size suited to a prescribed fire** – Prescribed fires may be very small if hand ignited near valuable assets, or quite large in inaccessible terrain. In eucalypt forests, 200 ha per day was considered reasonable for a ground party, but up to 15 000 ha per day when aerial ignition was carried out with ground support (Luke and McArthur 1978, pp. 144–145).
5. **Of a size where edge-based suppression measures reach the entire block** – For example, if tankers that spray up to 30 m from the track are used, then blocks 60 m wide may be considered ideal. If hose relays are laid out to 1000 m (Gould 2004), then 2000 m is the significant metric of block edges. Thus square blocks would be 400 ha in area.
6. **Of varying size created by encircling the fire with a new track (containment line) for every fire** – If this is done then any adverse effects of tracks are multiplied, even if the effects are short lived; all the original tracks are there yet more are created. Bulldozer tracks or grader tracks are likely to be visible for many years. Firefighters may deem such tracks necessary to halt the fire quickly, to lessen the chances of death of people or damage to property. In a major fire, suppression lines may be created at successive stages if earlier ones fail. An alternative to the

suppression lines being created within a block is the use of an established perimeter track as the place to fight the fire. Choosing the place to fight a fire can be important. By selecting the perimeter track, there may be a perceived need to accelerate the burning of fuels within the block (when weather has ameliorated sufficiently) by igniting the internal edge of the track and dropping incendiary devices from aircraft onto the unburnt fuel. As a result of this tactic, the length of burnt perimeter is increased so that chances of rekindling – proportional to the perimeter length – are increased. On the other hand, the fire lit at the edge is likely to be of relatively low intensity, compared with a fire of high intensity predicted to occur later – when it might easily cross the perimeter track. An implication of the burn-out-the-block tactic is that no matter what the size of the fire, the whole area is burnt.

The shape of blocks, as well as their area, may be a consideration. For example, blocks can be elongated in the direction of the wind prevailing at the times of extreme fires. With an intense fire, containment may be possible within the elongated shape when conditions ameliorate at night. Indirect attack from the flank of the fire may help. If the fires are kept narrow, then the potential rate of spread may not be realised. The widths of heading fires enabling speeds near the potential rate needed to exceed 125 m in grasslands and 175 m in woodlands, when winds were up to 17 km hr^{-1} at 2m height (Cheney and Gould 1995); higher wind speeds and heavier fuels could increase this width. On the other hand, the optimistic approach is that elongated blocks at right angles to the direction of prevailing winds allow more opportunity for suppression – direct attack of the head fire (that part of the perimeter burning with the wind).

Table 2.2 summarises a number of criteria associated with the formation of track networks in a fire-suppression environment, and a number of attributes associated with each. Density, the focus of this section, is included along with other criteria from previous sections. The range of criteria present and the attributes needed in particular places will vary widely with local circumstances, including neighbouring land uses.

Table 2.2 Track network characteristics and various track attributes associated with them in relation to firefighting.

Criteria for track network	Relevant attributes
Access	Track density, track width, track surface quality, slope and arrangement.
Burning out	Block sizes, track width and quality and verge treatment.
Chance of ingress or egress of fires from a block	Perimeter track width and quality (e.g. lack of fuel on track) and verge treatment.
Cost	Length, intersections, creek crossings, terrain, surface type, soil characteristics and drainage.
Firefighter safety	Number of dead ends and intersections, track widths for passing and turning, track quality, cleared areas providing greater safety of people in trapped vehicles and accessibility (see above). (Verge or block treatment effects are not considered here).
Passive suppression	Block sizes and shapes, track width and quality and verge treatments (for maintenance, effectiveness and potential rehabilitation after severe fire and suppression operations, such as width enlargement).
Prescribed burning	Block sizes, slope, track width and quality and verge treatment.
Travel times	Circuitousness/directness, width, surface and slope.

There is no single answer to the question of what sizes and shapes blocks enclosed by tracks should be because so many variables are involved. Important variables not considered here in any detail are the heterogeneity of vegetation within the block, the type of vegetation and type of terrain (e.g. dune fields and plateau–gorge country). Next, the proposition that areas defined by tracks within a reserve would be unnecessary if aerial suppression was completely effective is examined.

A track-free or minimal-track reserve

General

In this section, we take as the starting point a premise diametrically opposite to that which opened the chapter. Instead of assuming that a track network with access to all points is needed, why not use aircraft or all-terrain ground tankers for fire suppression and have no formal track at all, other than the perimeter track? Given the growing media prominence of aircraft use in firefighting, and the number actually used, could it be that tracks are unnecessary for firefighting? This is not a facile conjecture, as it has been informally advanced, or implied, even in professional fire-suppression circles. If practicable, this would minimise environmental problems associated with the normal, obvious tracks – wouldn't it? This section is designed to provide a framework for informed discussion, rather than present a point of view.

There are a number of ways that fires in a relatively track-free zone could be fought:

- By air – If the fire could be reached quickly by air, couldn't any fire be put out without the need for tracks? Or, by using narrow hand trails put in by remote area firefighting teams (RAFT) dropped into the area by helicopter, could the formal formed track network be redundant? 'Throw everything at the fire at the start no matter what the expense and save on costs in the longer term' is a way in which this proposition has been put.
- By burning out the fuel ahead of any fire from natural fuel breaks, so the fire is deprived of fuel.
- By ground tankers, using only water and travelling cross-country.

It could be argued that tracks are not needed on reserves where grasslands predominate, because vehicles can cross open landscapes in order to reach the fire. This is true to some extent, but there may still be difficulties in crossing streams, bogs and gullies, traversing steep or rocky country and passing through fences. Hummocks and tussocks can impede speed. Hidden rabbit warrens and pointy stumps of shrubs can stop a vehicle or even damage it. Fires may be too intense to control and passage across country may cause environmental damage, such as soil compaction. Suppression vehicles and people might be more readily trapped by fire where there is no track, especially when there is abundant spot-fire development.

Aircraft and fires

Aircraft come in many forms and sizes, with fixed wings or rotary wings (helicopters). They have a range of efficiencies in carrying out their various tasks in different circumstances. Roles they play are to (see Wareing and Flinn 2003, p. 216):

- Aid fire detection during or preceding known events
- Ferry firefighters to and from the fire ground, including rappelling crews (RAFT) into fire areas
- Facilitate infra-red and other scanning of the fire area
- Rescue people
- Suppress fire
- Supervise air attack operations
- Carry out aerial ignitions
- Assist in reconnaissance (including that of fire behaviour).

In this section, we are only concerned with the role of aircraft in suppression. In the north-eastern Victoria fires of 2003, approximately 43% of flying hours were spent on fire suppression – water bombing (data from Wareing and Flinn 2003, Appendix 7). If an average of about 400 m of wet fire line was laid down per hour (Table 2.3; I. Dicker's average estimate for fires at the same time in New South Wales, pers. comm., 2004), a relatively high rate in general (McCarthy *et al.* 2003), then about 570 km of line was laid down in these fires. This can be contrasted with the 9000 km of bulldozer trail (assumed from 'fire control lines requiring rehabilitation') in the same Victorian fires (Wareing and Flinn 2003, p. 211). The cost-per-kilometre for aircraft-established wet or retardant lines appears to be an order or two more expensive than ground-based methods (Table 2.3). It is also slower, yet applicable, in more difficult terrain. Calculated costs are affected by the inclusion of the costs of dozed or other bare-earth lines that are needed to support wet lines (Gould 2004) along with any costs needed for rehabilitation of lines.

Costs are dependent on a number of variables, such as turnaround time for water pickup and the type of aircraft, for example. In addition, effectiveness depends on the stage of fire development, current deployment of firefighters and when they might become available to work in the target area, and the value of the assets at risk (I. Dicker, pers. comm., 2004). Cost-effectiveness will therefore depend on circumstances.

Table 2.3 Some indicative figures[4] for suppression lines (wet or dry) laid down during fire-suppression operations. Wet lines do not involve soil disturbance; dry lines are fuel breaks. Rates of line construction are affected by the experience of the operator. Rates of line construction from the air can be strongly affected by time for refilling and return.

Type	Rate of construction (m hr^{-1})	Cost (2004 Australian dollars per hour)	Cost (2004 Australian dollars per km; from previous 2 columns)	Source
Wet lines – aircraft				
1400 litre capacity medium helicopter	<400 (5–15 min turnaround)	$4,700[a]	<$11,700	McCarthy *et al.* (2003) – forest; I. Dicker (pers. comm., 2004)
2500 litre capacity fixed-wing bomber	<140 (30–60 min turnaround)	$3,500[a]	<$25,000	McCarthy *et al.* (2003) – forest
3000 litre Air Tractor 802	400 (15 min from base)	$2,900	$7,520 ($24,500 with retardant)	I. Dicker, (pers. comm., 2004)
Sky Crane		$11,000 (on standby $25,000 per day)		I. Dicker, (pers. comm., 2004)
Wet lines – ground tanker				
4000 litre	<2000 (forest)			McCarthy *et al.* (2003) – forest
Dry fuel breaks				
D4 dozer	<700 (forest) <1000 (ACT)	*ca.* $180[b]	<$180	McCarthy *et al.* (2003) – forest; ACT Bushfire Service (undated)
D6 dozer	<900 (forest) <1000 (ACT)	*ca.* $215[b]	<$215	McCarthy *et al.* (2003) – forest; ACT Bushfire Service (undated)
Grader	*n.a.*			
Handline trails	<23 per person (forest) <60 per person (ACT)	Volunteer crews assumed		McCarthy *et al.* (2003) – forest; ACT Bushfire Service (undated)

[a] Costs supplied by Nick Ryan, Victorian Department of Sustainability and Environment (pers.comm., 2004).
[b] Includes 10% Goods and Services Tax (GST) and the cost of a trailing safety vehicle. It does not include dozer fuel. Standby costs (costs to have vehicles ready to use) can be the same as operating costs of the dozer only (say $A150 or $A190 per hour for D4 and D6 dozers respectively or as little as 50%) (D Broderick, pers. comm., 2005). Costs are approximate and designed to give an idea of the amount, not a quotation on any operation.

4 Figures vary considerably according to various terrain and human factors.

Environmental impact of aircraft suppression operations

While there could be an assumption that the use of aircraft in suppression would have no environmental impact at all, this cannot be sustained. There could be the effects of:

- Helipad establishment and operation (Mohr 1994), including service tracks
- Artificial water storage development and maintenance
- Water contaminated with seeds and spores of unwanted species from external sources
- Retardant chemicals and suppressant foams used to aid suppression (Adams and Simmons 1999) – discussed in further detail, below
- Spills of chemicals (see Adams and Simmons 1999), fuel and oil
- Depletion of scarce water resources from farm dams.

The first two of these are likely to occur in particular sites in the landscape, not just anywhere. These sites could be prime sites for biodiversity. These prime sites are not necessarily hot spots of exceptionally high diversity, but may have rare or threatened species present. Given an awareness of the possibility, sites can be chosen with biodiversity implications in mind. All except the first of these effects apply to ground-based suppression operations also.

The effects of chemicals added to water to enhance the effectiveness of suppression operations needs to be quantified (Adams and Simmons 1999). There are two basic types of chemical additives. The first are the plant fertilisers based on nitrogen and phosphorus, such as ammonium phosphates, which retard flaming combustion. Their delivery may be accompanied by thickeners, colouring material and corrosion inhibitors. Adams and Simmons (1999) describe a variety of types of retardants and their composition. The second are foams, wetting agents and surfactants, usually under the title *Class A foams* (as opposed to structural firefighting foams that are in another class), which contain ammonia (Adams and Simmons 1999). There may also be an effect from chemicals that are found naturally in the water source, such as the salts in sea water.

Aerial suppression

Retardants are able to act when dry, whereas suppressants can only be effective when wet. This has important implications for aerial application, because maintaining wet line is more difficult than maintaining a retardant line in hot–dry conditions – wet lines may last for as little as 30 minutes (I. Dicker, pers. comm., 2004).

For aerial suppression to be effective on a newly started fire, the rate of aircraft delivery of suppressant (e.g. water or water plus foam) would have to be quick enough so that, on reaching the area, enough suppressant could be effectively applied to the entire fire perimeter to indirectly extinguish or directly *knock down* (i.e. put out the flames) the fire. On a longer-term basis, suppressant would have to be applied at an effective rate that was faster than the rate of perimeter growth. This sounds simple enough, but the variables involved are not only the properties of the fire – rate of spread, intensity, spotting distance (see further details, below) – but also the numbers of available crews and equipment and suppression-response variables, such as:

- Time elapsed from ignition to detection (usually short, but not always – except in remote areas)
- Time from detection to the despatch order being given (usually short)
- Time from the despatch order being given to the departure of air and/or ground crew (usually short)
- Travel time to assess the fire situation (usually short)
- Time to collect suppressant agent and travel to the fire (or *vice versa*) (perhaps 15–30 minutes in south-eastern Australia)
- Time to lay down a wet line (see Table 2.3)
- Length and width of effective line – Influenced by number and types of aircraft and ground tankers, use of water, foam or retardant, rate of application of materials, their evaporation rates and dry effectiveness and the roles of fire properties, vegetation type and weather.

This simplified view of the problem does not take into account the periodic need for refuelling or maintenance and repair of aircraft during the operation. It gives no idea of the continually changing circumstances, as fire properties change diurnally and with the passage of weather systems and as fuel arrays and terrain change. The distances between water supplies, the shift in the fire line with time, and the location of the nearest water supply might also alter. Local water supply may be limited and run out during an operation. Implicitly included are delays due to safety considerations for the operation of aircraft (e.g. too windy or too dark; Wareing and Flinn 2003, p. 207; see below). Finally, the cost of the operation may be prohibitive. Table 2.3 suggests that costs per kilometre of suppression-line construction from the air can be an order of magnitude greater, or more, than those on the ground.

The implicit limits to aircraft use, mentioned above, need to be made explicit because, upon doing so, it becomes clear that using aircraft alone under such conditions could be ineffective and lead to much larger fires – and potentially greater economic loss. In short, the limits to aircraft use, apart from those due to fire properties, such as intensity (Le-Ray-Meyer 2003), are:

- High cost (Table 2.3)
- Inoperability at night (Le-Ray-Meyer 2003)
- Inoperability or limited operability in dense smoke, dust and extreme or turbulent winds (Le-Ray-Meyer 2003; McLeod 2003, p. 105; Wareing and Flinn 2003, p. 207)
- Inoperability during poor visibility where powerlines are present.

It is also likely that the perimeter of the burnt area will be surrounded by a surface track (Gould 2004) to minimise any opportunity for reactivation of the fire and its continued spread; an exception may occur if rain extinguishes the fire. 'A mineral-earth fire line [i.e. fuel break around the perimeter of the fire] is considered necessary in Australian forest fuels' (McCarthy *et al.* 2003). The mineral-earth fuel break can be a simple rake-hoe line put in by hand or a bulldozed track. In general, tracks are likely in areas set aside for conservation, but various methods can be used to minimise their number and increase their effectiveness.

Researching the effectiveness of aircraft, across a range of fire behaviours, can only be done with great difficulty, because of the immediacy of operational circumstances and the need to instantly be available in the right place at the right time during the bushfire season. However, as an alternative to this, McCarthy and Tolhurst (1998) studied the effectiveness of aircraft in putting forest fires out early in their development (initial attack) by looking at the influence of fire behavioural circumstances indirectly – through a knowledge of fuel arrays and weather conditions at the time of fire outbreak. Their results, expressed in the form of a statistical model, indicated that the probability of successful suppression was very low to nil when fuels were abundant and the McArthur forest fire danger index (FFDI; McArthur 1967) was above 50 (extreme). A shift in just one category of overall forest-fuel levels, however, caused the chances of successful initial suppression to rise to a maximum of 70% and a minimum of 10% for the same FFDI range. Where fuels were in low to moderate categories, the success rate under any weather conditions, even the worst possible, was modelled as greater than 90%.

The above considerations have not been explicit as to the type or number of aircraft used in any suppression operation. Loane and Gould (1986) concluded that 'a combination of helicopters and agricultural aircraft' – perhaps supplemented by a large fixed-wing aircraft in 'very severe fire seasons' – would be better than the use of 'any single aircraft type alone'. There will likely be optimum mixes of aircraft types and capacities for fire-suppression operations depending on assets threatened, budgets, travel distances, water sources, weather, fire circumstances, terrain, and fuel arrays.

In steep landscapes, the use of machines on the ground is limited, but aircraft can still operate, perhaps with some support from ground crews with hand tools. It is possible that aircraft could be the only means available for initial suppression in remote and steep country. Helicopters may have an advantage over fixed-wing aircraft for pinpoint suppression operations of recently ignited fires in steep country, and the possible follow-up using aerial delivery of RAFT.

Discussion

Table 2.4 is a summary of some of the features of tracks and the categories into which they are placed. The idea is to show that there are many variables to consider (see Figure 2.9 also). Having so many variables makes the task extremely difficult in some areas; however, simplifications are possible in many circumstances as not all variables apply in every situation.

Table 2.4 Track and verge networks (not including wet lines laid down by aircraft or natural edges), their characteristics and their assessment in terms of cost and effect on conservation.

TRACKS	
Property	**Categories**
Permanence	Temporary, dormant, permanent
Width	Suited to floats, tankers, light units
Lanes	One, two or more
Passing bays	None, present (frequency)
Surface	Bitumen, gravel, consolidated earth, scraped earth
Density	See Table 2.1
Drainage works	None; spoon drains (S); S and mitre drains (M); M, S and rollovers[5] (R); M, S, R and culverts (C); M,S,R,C and bridges
Location	Fire perimeter, block perimeter, reserve perimeter
Main purpose	Fire suppression, travel, visitor access
Maintenance	None, ends closed until fire occurs, routine
VERGES	
Property	**Categories**
Width	None, narrow, wide
Drainage works	See above
Treatments	Clearance; soil disturbance; slashing/mowing; herbiciding; planting of grasses, shrubs or trees with or without mulch; tree cutting; burning; grazing
Maintenance	None, rare, routine
BLOCKS/NETWORKS created by edges/tracks	
Property	**Categories**
Area	Small, large (or area given in hectares)
Number/density	Low, high
Shape	Square, rectangular, irregular
Intersection types	Two-way (corner), three-way (e.g. T-intersection), four-way (crossroads)
Dead ends	None, few, many
ASSESSMENTS	
Measure	**Categories**
Dollars	Number or high, medium and low
Species' extinction (indigenous *spp.*)	None, a possibility, certain
Species' population (indigenous *spp.*)	Known to be affected by tracks, not affected, not known
Habitat effect	Hollows of various sorts, habitat complexities of various types (e.g. foliage-height diversity)
Cultural relics	Fire prone, not fire prone
Aesthetics/spiritual quality/ untrammelled nature	Sites of significance, others
Affect on water yield and quality: amount, chemical composition, suspended solids	Within limits, outside limits
Weeds and ferals	Unaffected, promoted

5 Rollovers are earthen humps across steep sections of track to slow or divert water and to catch sediment.

Tracks and fire suppression

What is considered to be *enough* for a track network is likely to be perceived very differently in different regions. In sparsely settled, western New South Wales, after the extensive 1974–75 fires, the need was expressed in the following way: 'a system of firebreaks [i.e. fuel breaks] would have to be constructed to enable ready access for firefighters and to be used as a basis for back-burning operations'. A 'huge firebreak system' was inaugurated with blocks of approximately 32 000 ha each (Bushfire Council of New South Wales, extract, 1970s**)**. Blocks of this size, with woody fire-prone vegetation, would not be tolerated, or likely to occur now, near the edges of cities.

Creating a track network can have adverse environmental impacts. In itself, a track network will not stop all fires. However, the efficacy of a track network for fire suppression can be improved to an extent, dependent on the capacity and efficacy of suppression forces available, along with local fuel treatments. The use of aircraft could decrease the density of track networks perceived to be necessary for fire management. There is no one simple answer, but perhaps many partial answers, to the quest for the ideal system of tracks for fire suppression within a biodiversity management context. Cost, suppression capacity, firefighter safety, environmental effects, weather, vegetation, fuel treatments (see next chapter), proximity to economic assets and terrain all influence the outcome. Biodiversity conservation is rarely the sole aim of management. It usually occurs alongside others for water supply, recreation, cultural conservation and utilities. Tracks of one type or another are inevitable in moderate- and large-sized reserves, where various assets are threatened by fire occurrences.

Real world situations

It could be argued that all present systems have limited effectiveness where extreme fire weather and continuous fuels occur. Experience during the multiple fires of 2003 in south-eastern Australia in three jurisdictions at the same time (Victoria, New South Wales and ACT), suggests that the best aerial and ground suppression resources currently available will not be sufficient to stop a fire during extreme weather. However, in the absence of a track network, suppression system and fuel-modification program (considered in the next chapter), the situation could be worse. These three components – tracks, suppression capacity and fuel modification – need to be considered together, not separately, and an appreciation of their combined effectiveness sought to achieve economic asset protection and biodiversity conservation objectives. Subsumed in the track part of this consideration will be firefighter safety, cost of installation and maintenance of the network, ease of burning out fuels (suppression tactics) and environmental effects (Table 2.4).

In real terrain, which is often mountainous and forested, the creation of track networks will not only be constrained by the issues raised above, but also by environmental matters – some in relation to the effects of the tracks, but others simply as a matter of cost effectiveness. For example, mountain ranges are often aligned in a particular direction with more or less parallel ridges and valleys. Spurs may run down into valleys from ridge lines at sharp angles, demarcating ever-widening catchments. Plateaux, cliffs and gorges may occur in places. Geological substrates may vary, affecting soil type and erosion potential, along with slope angle and length. Fuel arrays may be diverse. However, tracks will tend to favour ridge lines for economic, environmental and scenic reasons. River or gorge crossings requiring bridges will be avoided, cliffs will be impassable and valley lines may be subject to flooding, thick vegetation and narrow, steep upper reaches. Yet wide grassy valleys may be the best place to suppress fires, rather than on ridge tops with woody fuels. Temporary perimeter tracks can sometimes be butted onto drainage lines, rather than directed down steep wooded spurs. However, proximity to drainage lines may increase the chance of erosion and sedimentation, and therefore the disturbance of water supplies to cities and local communities. Ridges and upper slopes may be the best places for prescribed burning to take place from tracks because ridges typically have drier fuels and fires can be ignited so that they travel against the slope and achieve only low intensities.

Ever present in the conservation manager's mind may be the effects of the tracks on rare and threatened plants and animals, feral animals and weeds, cultural sites, water quality and quantity, disease organisms, unwanted ignitions and passive control of fires entering the area – as well as suppression considerations.

At the time of a campaign fire, there can be a single mindedness to establish new tracks wherever possible. The tracks created may not be in the best interests of the achievement of the overall aims of management, including fire management. A suitable guideline for incident management teams has been summarised in the term *minimum impact suppression tactics* or M.I.S.T. (Mohr 1992–3; Mohr 1994; Mohr and Curtiss 1998). The tactics for particular terrains and aims of management can be determined before the campaign fire occurs. Mohr's publications are a useful starting point for the development of local guidelines, such as those in the Annex to the New South Wales Bush Fire Coordinating Committee Policy 2/2006. The guidelines can include the post-fire repercussions of the tactics for rehabilitation as well.

Tracks have a role to play in fire management, but there is no simple way to *optimise* the design of a track network for suppression effectiveness and firefighter safety, let alone avoid all deleterious environmental effects and minimise installation and maintenance costs. At one extreme there is but a bare-earth area – the ultimate wide track across the estate, an absurd proposition. At the other extreme is no track at all to access any fire – say in a *wild-ness* area. Such a proposition would concern authorities charged with suppression responsibility on behalf of the public, because by the time the fire emerged from the area it may be uncontrollable until weather abated and supporting suppression forces were reinforced from other jurisdictions, even international ones. Even with an effective aerial suppression operation at hand, there are limits to effectiveness and the economic cost will be substantial.

Are there any guidelines that can be offered about track networks? This chapter finds that there is a complex set of criteria to be used in the determination of track networks (Figure 2.9). Efficacy of fire suppression is just one of these criteria. For suppression – including the use of aircraft for reconnaissance and rapid deployment of firefighters, not just water bombing – a mix of air and ground methods will be usual. Ground methods will remain essential and some form of track network will usually be necessary to support them. Having a perimeter track, not necessarily on the exact edge of a reserve, could be important in some reserves. Its width and quality could be determined by fuel type, the chance of spotting of fires to various distances and the sorts of ground suppression available (e.g. only dry methods in most of the arid zones). The effectiveness of tracks may be enhanced by verge and block treatments, considered in the next chapter. Internal tracks may be useful for enhancing the safety of firefighters and for treating fuels. Internal tracks can be thought of as permanent, dormant or temporary. Of overall significance to decision making during operations is conformity to the overarching management objectives.

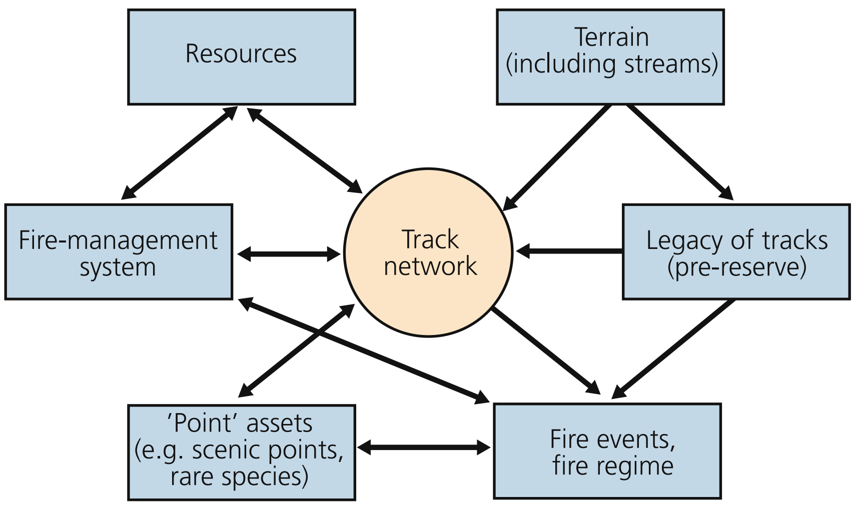


Figure 2.9 Simplified view of some of the main variables affecting a real-world track network and their interactions, in a reserve set aside for biodiversity conservation. Not included here are fuel types and their distributions, which can affect the track network, management system, fire regime and so on.

The fire-management concept – within the context of biodiversity conservation – is that track networks, fire-suppression capacity and potential fire behaviours need to be considered together, not alone, and in relation to the short- or long-term risk of extinction of plants and animals. Factors that limit track effectiveness need to be addressed in the short- and long-term – e.g. funding, terrain, environmental effects, equipment and knowledge.

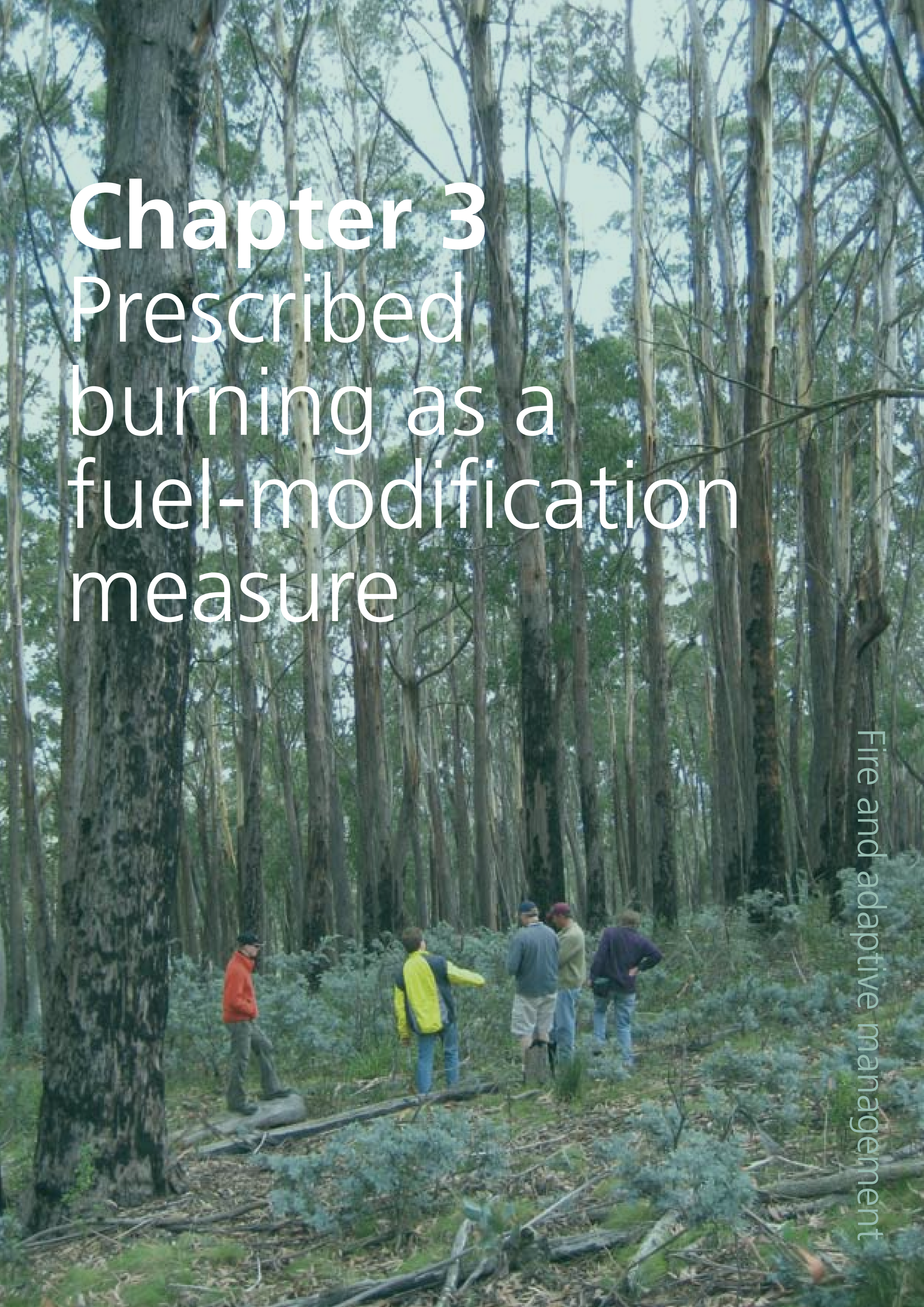
Conclusion

Tracks and roads are seen as an integral part of any firefighting infrastructure, but they have indirect and direct effects on the environment, including biodiversity. In short, these adverse effects may relate to: drainage lines, erosion and sedimentation; native animals and plants; weeds, diseases and feral animals; and unwanted ignitions. How can tracks be kept to a minimum and negative impacts ameliorated? A first premise in examining this question is that suppression effectiveness is maximised by high accessibility. When a grid of points is used to define a model area, the shape of the shortest (cheapest?) model track network to reach all these points can be described. Having done so, it becomes clear that factors other than access (e.g. firefighter safety, ease of burning out blocks and continuous perimeter protection) need to be considered. To accommodate these matters, model networks can be revised to include perimeter tracks that define a series of areas of land suited to prescribed burning. Tracks vary in their characteristics – width being particularly important to the effectiveness of suppression in grassfires and for traffic flow during fires.

If aerial suppression is possible, are tracks needed at all? In forests, aircraft can construct wet tracks at about half the rate of bulldozers and cost something like an order of magnitude more per kilometre. Bare-earth breaks of some sort almost invariably accompany them, as wet lines only provide a temporary holding function for fire spread. Bare-earth perimeter tracks around fires are usually necessary to prevent rekindling of the fire and further outbreaks. From a biodiversity point of view, it would be a mistake to think that aerial suppression can have no adverse effect, as effects of fire suppressants and infrastructure on the ground can be significant. All methods of suppression have limitations. No system is totally effective unless it is a complete bare-earth system – an impractical, nonsensical extreme. For suppression alone, the concept is that track networks, suppression capacities and fuel modification need to be considered in concert with the fire characteristics expected. For the best possible outcomes, suppression operations need to be addressed in relation to land management objectives.

There is no single simple answer to the question, 'What is the length of track suited to fire management of a reserve?', partly because length is not the only criterion for the establishment of a track network. Other criteria include: the activities that are to be serviced by tracks (e.g. access to beauty spots, access to fires, egress from fires, prescribed burning, fire suppression, servicing of boundary fencing and feral animal control); the safety context of tracks (e.g. for firefighters, general road users and park rangers); the establishment and servicing cost-implications of the track network; and the limitations to local track networks imposed by the terrain (e.g. swamps, rivers, estuaries, gorges, ridges and sand drifts).

Chapter 3
Prescribed burning as a fuel-modification measure



Chapter 3 Prescribed burning as a fuel-modification measure

Introduction

In chapter two, the value of tracks in fire suppression was examined in the context of biodiversity conservation. From the point of view of stopping a fire, the fire property examined in grasslands (without trees and other significant sources of brands) was the length of flames in relation to the width of the track in the fire's path. In eucalypt forests in which the production and spread of fire brands is significant, tracks were useful for stopping fires, but only over a limited range of intensities. Therefore, the modification of the fuel array throughout a significant proportion of a forest area, to minimise the production and dissemination of airborne burning materials, may be advocated. With reduced fuel loads and modified fuel structure, the potential for spread by fire brands can be reduced for a number of years.

A common cost-effective way to reduce fuel loads and modify fuel structure, especially where woody fuels are predominant, is to use fire under defined and low-risk conditions – prescribed burning (Text Box 3.1; Plate 3.1). Prescribed burning reduces the fuel load, fuel continuity, loose lower bark of trees and, initially at least, the proportion of dead-to-live fuel contained within the fuel array. Therefore, by definition (see Chapter 1), the potential fire intensity is reduced. It may then be argued that this in turn increases the chance of controlling the fire, and therefore decreases the chance of the loss of human life and economic assets.

Another broad-area technique is the use of grazing animals to reduce fuels, especially where grassy fuels predominate – even in forests. This will be examined in the next chapter. In this chapter, prescribed burning as a fuel-reduction measure is considered in light of conservation objectives.

Plate 3.1 Prescribed low-intensity fire for fuel reduction (Graham 2004).

Text Box 3.1. **Prescribed burning**

The term *prescribed burning* can be used in a number of ways. In Australia, the definition can be expressed as: the deliberate and safe use of fire to achieve an explicit outcome under specified conditions of fuel, weather and ignition in a designated area. Therefore, it does not include a fire that is unplanned (wild), even a low intensity one that fits a prescription – as is used in the USA (Bunnell 1995). It is also not ad hoc burning off; however, it does include a number of diverse circumstances, which include:

- The burning of woody debris (slash) after logging, often at high intensity, to remove as much woody material as possible (Plate 3.2) and create a seed bed for the regeneration of trees
- The burning out of fuels during suppression of an unplanned fire
- The application of fire to achieve a particular ecological purpose
- Broad-area treatment under mild weather conditions to reduce the quantity, arrangement and continuity of forest litter or shrubby/grassy fuels in an area (Plate 3.1).

In this chapter, the emphasis is on the last of these, sometimes called fuel-reduction burning, but given that fuel is what burns, this term is a tautology (Esplin *et al*. 2003, p. 74). Fuel reduction by low-intensity fire is a better description. Fuel modification may be preferred by some as the arrangement of the fuel can be important. Note that if a low-intensity fire kills, but does not consume, the green component of a grassy sward, shrubs, lower canopies of trees or green bracken, fuel available for future fires may be generated as well as reduced (see also Gould *et al*. 2007, p. 79). If the prescribed fire enhances shrub germination or the growth of bracken, for example, then fuel conditions also change.

Both prescribed burning and grazing will affect biodiversity, as both affect fuels in some way. Fuels may be partly seen as species or habitat – elements of biodiversity. Invertebrates may have at least part of their life cycle in habitats that are associated with live or dead plant or animal parts that are found above or below ground, exposed or concealed and in or out of water of various types. Non-vascular plants – including lichens, fungi, bacteria and algae – have a similar wide range of habitats. Fuel management always involves habitat, whether this be grass, bark, foliage of woody plants, standing or fallen logs or litter.

In this chapter, the links between vegetation, fuels and habitat are briefly considered before a discussion of aspects of fuel dynamics, the efficacy of prescribed burning for fuel modification and fire control, and the effects of fires on the environment.

Plate 3.2 Burning of woody debris (slash) after logging (Fisher 2005).

Fuel, vegetation and habitat

Fuel in a landscape is not often a simple object, such as a matchstick. Fuel for landscape fires can be dead litter, grass, tree crowns, live plants, micro-organisms, insects, houses, fences, cars or haystacks. Is it fuel before it burns? If it never burns, can it be considered fuel? What if it rarely burns? In practice, fuel is used as a shortcut term for material that has the potential to burn – the class of materials that are combustible.

While fuels are the materials that burn, not all potential fuels burn in all fires. In grasslands, consumption of much of the fuel present may occur on most occasions, but in many eucalypt forests and woodlands this is usually not the case. In forests, litter on the forest floor dominates the carriage of the fire front but grasses or low shrubs can have an important role too (Cheney *et al*. 1992; Gould *et al*. 2007). Also, forest canopies can catch alight and burn, as can peaty substrates and logs on the forest floor. Thus much of the above-ground vegetative complex (i.e. excluding living tree trunks and branches) can potentially burn in many fire-prone vegetation types but do not always do so.

What is considered fuel by some observers may be considered habitat for animals or floral biodiversity by others. This suggests that there may be parallels between the ways in which vegetation can be measured as habitat and as fuel (Peter Catling, pers. comm., 2002). Newsome and Catling (1979) developed a Habitat Complexity Score (HCS) to describe the habitat of small ground-dwelling mammals in south-eastern Australia. This score included the cover of ground herbage, shrub canopy, tree canopy and 'logs, rocks and debris' on the ground. The potential overlap with potential fuel arrays may be apparent (Table 3.1).

It is important to note that the HCS is for ground-dwelling mammals only and not for birds or arboreal animals, for example. Hollows are a significant feature of the habitat of arboreal animals and many birds, and fires appear to both help create and destroy hollows (see Gill and Catling 2002). Tree hollows can also aid the combustion of trees but do not feature in HCS or fuel classifications. Lateral cover of foliage, as a function of height above ground in forests, seems to be important to a variety of birds, some preferring more open conditions, others denser forests (see Gill and Catling 2002). Habitats are many and varied for the plethora of vertebrate and invertebrate animals. Brown *et al*. (1998) use a combination of macro (vegetation) and micro (e.g. soils, burrows, litter and tree trunks) attributes to define habitat for the fauna of the Australian Alps National Parks in south-eastern Australia (after Woods 1996). Thus ways of defining habitat depend on the fauna of interest and the degree of overlap between classifications of habitat and fuel will vary.

For example, fuels may be classified on the basis of vegetation structure (e.g. forest, woodland, shrubland and grassland), life form within a structural type (e.g. grass, shrub and tree), plant taxa (e.g. wire grass, bracken, *Lantana* and Gamba grass), fuel-component type (e.g. peat, bark and litter), ignition likelihood (availability) or a combination of these. Sandberg *et al*. (2001), in 'characterizing fuels in the 21st century' in the United States, highlight the 'kind, quality and abundance' of fuels as a means of inferring their 'physical, chemical, and structural properties'. Gill and Zylstra (2006) discuss the flammability of fuels in eucalypt forests at different scales.

Forest fuel in southern Australia is now often depicted as an array of materials – rather than just a monolayer of litter, for example. Sneeuwjagt and Peet (1985) incorporated shrub components in their Western Australian fire behaviour tables, while McCaw (1991) described the components of forest-fuel complexes being adopted by Australian Forestry authorities. Cheney *et al*. (1992) found that height and dead-fuel moisture content of near-surface fuels contributed to a prescribed-fire rate-of-spread model in regrowth eucalypt forest in New South Wales. They described the fuel array in terms of surface fuels (e.g. litter), near-surface fuels (e.g. grasses and short shrubs) and elevated fuels (e.g. taller shrubs, up to 2 m in height). In Victoria, Wilson (1992a, 1992b, 1993) initiated a descriptive system for forest fuels. This system, designed to assist in predicting the difficulty of suppression, included litter (t ha^{-1}), bark on trees (a rating) and elevated fuel (a rating) in an overall fuel array. The bark effect on the overall rating was contingent on a threshold litter level of 4 t ha^{-1} (Wilson 1993). Essentially, these systems provide a weighting of different fuel components in relation to their effects on rate of spread and/or suppression difficulty in eucalypt forests. The Overall Fuel Hazard[1] (OFH) Guide (McCarthy *et al*. 1999a) provides an overall rating system for the fuel array based on Wilson's concepts.

Following the trend for fuel rating, described above, Gould *et al*. (2007, in Table 3.2) have depicted forest-fuel arrays in south-western Western Australia and given ratings a numerical score. In doing this, their aim was to develop a better understanding of the way in which fuels in different parts of the array affected fire behaviour in the dry eucalypt forests of their study. They described their fuels in relation to five layers within the forest: surface, near surface, elevated, intermediate and overstorey (pp. 17–20). Bark was the predominant fuel of the intermediate and overstorey layers. A 'combustibility score calculated from the sum of the products of hazard score and cover in each of the five fuel layers' was correlated with rates of spread of their experimental fires (p. 73). However, their final rate of spread model used the variables wind speed, 'fine-fuel moisture function', 'surface fuel hazard score', 'near surface fuel hazard score', 'near surface fuel height' and slope (p. 91).

From the above consideration of forest fuels, the value of the structural approach is apparent. There may be further issues to resolve in terms of thresholds and contingencies – as in Wilson (1993) where 4 t ha^{-1} was a precondition for overall fuel rating – and the weightings given to each of the layers of a forest (Gould *et al*. 2007, p. 30). What is apparent is that there is considerable overlap in the attributes that are important to habitat (for certain animals) and fire behaviour (for determining rate of spread or degree of suppression difficulty), and vegetation classification also. Managers seeking to keep track of both habitat and fuels may choose to rate or measure attributes at the same sites in order to minimise the effort needed to monitor them. Table 3.1 compares attributes of the HCS, OFH and the vegetation classification scheme of Specht (1970).

The use of profile diagrams to depict fuel arrays, habitat and vegetation change may be worth exploring. For example, the initiatives for depicting fuels described above could be supplemented using the profile diagrams, such as those of Walker and Penridge (1987). In this way, the vertical profile of fuels could be depicted in terms of the variation of cover and fuel load and type with height, in particular. Profile diagrams could provide a visual image of changes with time. Their use would facilitate the recording of fire occurrence and severity at monitoring stations (taking severity to be a measure of post-fire scorch (fire-browned leaves), fire defoliation (black canopy), intact green canopy and bark char; thereby forecasting scorched-leaf fall, bark shed in smooth-barked eucalypts and branch contribution to post-fire fuels and nutrient cycling.

Table 3.1 A comparison of the component variables of the Habitat Complexity Score (Newsome and Catling 1979), Overall Fuel Hazard (McCarthy *et al*. 1999a), vegetation classification (Specht 1970) and a potential vegetation-profile method (after Walker and Penridge 1987) for Australian forests. Attributes used for each purpose are marked with a '+'.

Attribute	Vegetation classification	Habitat complexity Score	Overall fuel hazard	Fuel-profile diagrams
Bark type and condition	–	–	+	+
Ground herbage cover or near-surface fuels	+	+	+	+
Litter depth/cover/loading	–	–	+	
Logs, rocks, debris etc	–	+	–	
Moisture in the soil	–	+	–	
Shrub canopy cover	+	+	+	+
Shrub canopy height	+	–	+	+
Tree canopy cover	+	+	–	+
Tree canopy top height	+	–	–	+

1 Fuel hazard may be considered a misnomer as fuel per se is not a hazard until it supports a fire that threatens specified assets (Esplin *et al*. 2003 p. 76).

Fuels for prescribed burning

When the focus is placed on fuel loads, there are limits to fire control in forests due to, or correlated with, fire intensity (Chapter 1). Given that, by definition, fuel is a contributor to intensity, there is potential through prescribed burning to keep fuel loads down, reduce potential intensity and make suppression easier. If fine-fuel loads in forests are kept below about 8 tha^{-1}, then fire intensities on level ground are kept to 4000 kW m^{-1} and are therefore just controllable under a Forest Fire Danger Index of 100 (Gill *et al*. 1987*b*). If fine-fuel loads do not rise to 8 t ha^{-1}, then fuel treatment may be deemed unnecessary. If weather never becomes extreme, then higher forest-fuel loads might be tolerated. If ground is not level, then lower limits to maximum fuel load for fire control under extreme conditions are implied (Bradstock *et al*. 1998*b*; Esplin *et al*. 2003, p. 79).

The quantity quoted as the maximum upper limit of forest-fuel load for fire suppression and control is based on the fine-fuel load only (Gill *et al.* 1987*b*); this calculation may be regarded as unsophisticated. It takes no account of the spatial variation, nor the probable different effects of fine fuels present as bark on trees or dead and live material in shrubs. It assumes a level of fire-suppression capacity too. Thus it provides a worked example, an approach and an aim, rather than a definitive answer to the problem of fire suppression across all forested landscapes.

Whether fine fuels are grasses or forest litter, they follow a path to a quasi-equilibrium level (Figure 3.1) as they accumulate with time after fire (Walker 1981 for grasses, shrubs and litter). The shape of these curves is based on a constant input of fuel each year and a decomposition constant. The equation is of the form:

$W_{tsf} = W_{max} (1 - a.e^{-k.tsf})$ Equation 3.1

where W_{tsf} is the fuel accumulated since the last fire in a standard area, such as a hectare; W_{max} is the quasi-equilibrium maximum fuel load for the equivalent area; a is a constant; k is the decomposition constant and tsf is the time since fire in years (Olson 1963). Sub-year intervals are ignored here, but see Mercer *et al*. (1995) for a consideration of these. W_{max} equals A/k where A is the annual accession using the same units as for W. A is assumed to be constant in the Olson (1963) model but any change in A with time could be accommodated in the same equation.

The equation favoured by Gould *et al.* (2007, p. 26) for components of the fuel array in forests has the form:

$W_{tsf} =(a*tsf)/(b + tsf)$ Equation 3.2

Or, with altered values of the constants a and b:

$1/ W_{tsf} = a + b/tsf$ Equation 3.3

The form of these graphs is similar to that of Olson (1963), above, and is simpler to fit to data, but it does not reflect the processes of accession and decomposition, such as in Olson's one (Equation 3.1).

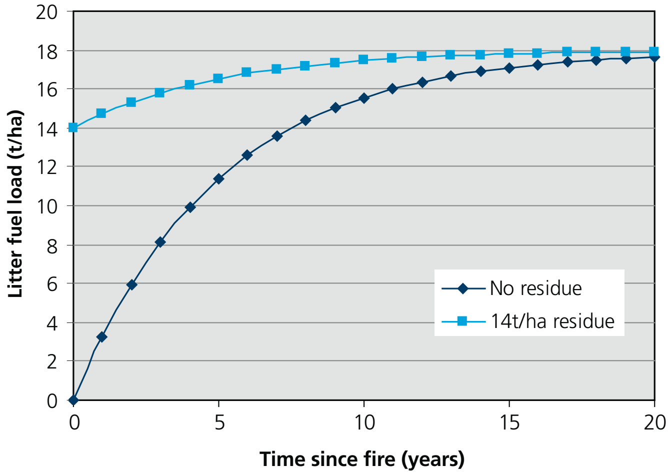


Figure 3.1 Fine-litter fuel load as a function of time since fire in *E. marginata* (Jarrah) forest (parameters with zero remnant fuel from Walker 1981 using data of G. Peet). The complete removal of fuel by prescribed fire (Equation 3.1 with W_{max} as 18 t ha^{-1}, *a* as 1 and *k* as 0.2) is contrasted with the illustrative circumstances in which 14 t ha^{-1} remains (equation as previously, except *a* is 0.222) – see the intercept on the Y-axis.

In Figure 3.1, the classic curve of Olson (1963) is contrasted with the situation in which some fuel persists, or is generated, immediately following the fire (O'Connell 1987). If large quantities of fuel remain, then the effectiveness of prescribed burning for fuel reduction would be questioned. For example, in Figure 3.1, a hypothetical illustration only, an average of 14 t ha^{-1} remains. This may not be considered to be a satisfactory result for fuel reduction per se. In practice, *k* varies considerably: Raison *et al*. (1983) found values between 0.11 and 0.31; Walker (1981) found values from 0.2 to 0.43; and Cary and Golding (2002) found *k* to vary from 0.03 to 1.

The form of the graphs given by Equations 3.1 and 3.2 for fuel measures against time since fire can be applied to all components of the fuel array in a forest, albeit with different parameters (see Appendix 2 in Gould *et al.* 2007). However, some of these components are likely to be unaffected by prescribed fire and variously affected by different intensities of unplanned fire. For example, Gould *et al*. (2007, p. 132) assumed – in developing a firebrand model – that 'elevated and bark fuel are not consumed at fire intensities less than 1000 kW m^{-1}'. The effectiveness of prescribed burning for the modification of fuels only, is considered further in the next section.

Effectiveness of prescribed burning for fuel modification

Prescribed burning may be undertaken for ecological or other purposes, but more often it is for the purpose of fuel reduction and modification (Text Box 3.1). In forests, where the practice has a long history (Gill and Moore 1997), the aims are stated in various ways, but Moore and Shields (1996) are more comprehensive than most. They state that the aims of fuel management – hazard reduction burning – are to:

- Reduce the total weight of fuels to reduce the rate of spread and intensity of a fire
- Reduce height of the fuel bed and therefore flame height
- Remove fire-brand material, principally fibrous bark, and therefore the chance of spot fires.

As a result, fire suppression capability is increased, the impact of fire on forest assets is reduced and the safety of firefighters increased (Moore and Shields 1996).

An immediate effect of fuel reduction by burning in State Forests of New South Wales was to reduce fine-fuel weight up to 70% over 30 to 60% of the gross area being treated (Moore and Shields 1996). For strategic corridors and fuel management over broad areas of forest, burning up to 60% of the area treated was desired (Moore and Shields 1966). In asset-protection zones in Victoria, the aim for the percentage of area to be burnt is 90% (McDonald 1999). In strategic fuel-reduced corridor zones it is 80% and in the broad-area fuel reduced mosaic zone it is 50% (McDonald 1999). In Tasmanian Buttongrass (*Gymnoschoenus sphaerocephalus*) the figure is >70% (Marsden-Smedley 1993). Thus not all the fuel is burnt over a chosen area, even for asset-protection burning.

What is actually achieved in the field depends on conditions that prevail during the burning operation. For the improvement of later operations, assessment and evaluation of each burning event in relation to objectives, field conditions and logistics is necessary. Poor weather may hamper operations and heterogeneity in fuel moisture may impede fire spread or prevent it in parts of the burn area. Such factors may lead to a lesser amount being burnt than intended. Mapping of outcomes (Heemstra 2006), in terms of unburnt areas and those burnt with various intensities, would be a useful part of assessment of the success of the event, let alone for ecological reasons.

Prescribed burning may be used to create linear buffer strips by dropping incendiary capsules from an aircraft in a line, then relying on the timing and placement of capsules to allow fires to join but then extinguish at night or when they reach natural and artificial barriers. This process can be particularly effective in tropical savannas (Crowley *et al*. 2003, p. 52).

While the protection of economic assets is often a prime objective in prescribed burning, it may be carried out for ecological reasons too (Fire Ecology Working Group 2004). The trend seems to be for more control of fire regimes for ecological purposes (Esplin *et al*. 2003, p. 115).

Assessing the effects of burning on fuel

In the previous section, the detailed objectives for prescribed burning events were given. The purpose of this section is to look at questions pertaining to the effectiveness of reducing, or otherwise modifying, fuel as an issue in itself, not in relation to wider questions of, say, threats to biodiversity conservation or to property.

Fuel modification can be examined in isolation or in a specific context, such as that of the urban interface (e.g. Bradstock *et al.* 1998*b*). Calculation of the risk of unplanned fire and its cost (e.g. economic, environmental and social), may help the manager decide whether or not the cure (fuel reduction) and its cost is worse than the disease (high fuel loads). In practice, monitoring the effectiveness of fuel-modification measures should be undertaken in relation to outcomes (especially social and environmental), not just in relation to fuel loads or scores.

If the fuels, such as some dominated by grasses, return to a level unacceptable to managers within one year, then burning or other means of fuel modification are necessary every year; if the effects of this situation are to be avoided. Frequent burning may reduce the chance of shrubby fuels dominating the grasses and changing the fuel load and structure in the longer term. In an international review of prescribed burning emphasising forests, Fernandes and Botelho (2003) concluded that 'Fuel accumulation rate frequently limits prescribed burning effectiveness to a short post-treatment period (2–4 years)'. However, what happens in particular areas depends on local fire weather, growing conditions, species present and suppression capacity, for example.

The most recent work on fires in Australian forests, by Gould *et al*. (2007, p. 79), emphasised both loads and structure in assessing the effects of fuel reduction by prescribed burning on fire behaviour. These authors suggested that in the Jarrah forests of Western Australia the effects of prescribed burning on future fire behaviour may last for 10–15 years. However, because only the bark on the lower parts of the tree bole are affected by low intensity fires (p. 114), bark characteristics on upper boles will continue to change in a way that favours the development of fire brands and spot fires. Consequently, they argue, 'There may be a good case for alternating mild spring burns with hotter autumn burns' (p. 79).

In grasslands, thresholds for fire control are unknown, although the discussion of fuel-break width provides some leads (Chapter 2). If a threshold fuel load can be identified, then some indices of effectiveness, or lack of it, are:

- The time (years) the fuel load is above the threshold as a proportion of the interval between fires
- The cumulative sum of the weight of fuel each year it is above threshold loads
- Potential maximum fire intensity.

The critical time, or time to reach the threshold, tsf_c, is:

$tsf_c = -1/k[\log_e(1 - W_c/W_{max})]$ Equation 3.4

Thus if, as in Figure 3.1, $k = 0.2$ and $W_{max} = 18$, and if the critical fuel load, W_c, equals 8, then $tsf_c = 3$ years. If burnt at this time, the interval between fires is 3 years. If $W_c > W_{max}$ then tsf_c become infinite, then no burning will be recommended based on this evidence (but may be on other evidence).

The time above the threshold fuel level, tsf_c, as a proportion of the between-fire interval, tsf_b, is given by $[(tsf_b - tsf_c)/tsf_b]$. Consider a grassland in which the fuel load reaches much the same, but undesirable, level within a year and stays there in successive years. Pre-emptively burning every year precludes any time above the threshold level – the ultimate for 'protection'. Burning every second year means that fuels are above (and below) what the manager may desire for protection objectives half the time.

In a forest, the fuel loads (or scores) may be as in Figure 3.1, with zero fuel left after fire; fuel is able to increase above a certain critical level for years. Perhaps the load and time above the threshold is a useful concept. This is given in part by the following equation in which only positive values are considered:

$(W_{tsf} - W_c) = W_{max}(1 - e^{-k.tsf}) - W_c$ Equation 3.5

This is equivalent to drawing a horizontal line through W_c on graphs such as Figure 3.1, and looking at values above the line each year to give the sum of these levels in the interval between the threshold year and the year that burning takes place. For example, say the critical fuel load is 8 t ha^{-1} and this is reached in three years, as in Figure 3.1. Then the sum of fuel loads above the threshold value for a six-year interval between fires is approximately 11 tonne-years, while that for a 10-year interval is approximately 38 tonne-years.

The fine-fuel load is just one of the fuel variables that may be of interest from a fire or environmental point of view. Curves, such as those in Figure 3.1, may be drawn for shrubs and bark on trees, logs on the ground, dead material in shrub canopies and crowns of trees and shrubs (see Gould *et al*. 2007 Appendix 2, for example). The value of the decomposition constant, *k*, is likely to vary for each component, as are the quasi-equilibrium loads (or scores). The remnant fuel loads (or scores) will vary according to component and to the properties of the last fire (e.g. in the absence of crown scorch all the canopy is remnant).

As illustrated in Figure 3.1, there can be fuel left after a prescribed fire or an unplanned fire. After a prescribed fire, a certain amount left covering the soil may be seen as being in line with land-management objectives. Tolhurst *et al*. (1992*a*) reported that 40% was often left unburnt in their prescribed burning experiments. They found that the percentage of fine fuel actually burnt in areas where the fire passed over was a linear function of the Keetch-Byram drought index (Keetch and Byram 1968). However, this relationship varied widely from autumn to spring; a much higher drought index being necessary in spring to burn the same percentage of the fuel.

The indices, above, have been considered in relation to a particular point on the ground rather than to an area. For a burning block, landscape or region, four related indices of effectiveness, or lack of it, are:

1. The total amount of fuel on a unit landscape at any one time, a combination of the amount of fuel accumulation with time since fire and the proportions of landscape with different fuel ages; an average fuel load (or score) may be obtained by assuming equal proportions of land affected and the use of Equation 3.1 for total removal each time
2. The geographical spread of certain fuel quantities in the landscape, or fuel quantities graphed by the area they occupy
3. Proportional area with fuel loads below, or above, the threshold amount
4. Patch sizes of fuels in different amount categories

The effect of the spatial arrangement of fuels for fire control is considered in Chapter 7.

Assessing the effectiveness of fuel reduction for fire suppression

In the last section, the direct effects of prescribed fire on fuels was considered. Here, the focus is on the efficacy of prescribed burning for fire suppression and how that might be measured.

Difficulty of suppression, and therefore control, is difficult to define because of the numbers of variables involved. For example, Gould *et al*. (2007, p. 32) mention visibility through a forest, access and difficulty of working machinery, flame height and spotting potential. Fire intensity is a convenient, single measure of the fire side of the story, because, in forests at least, suppression is likely to fail when spotting occurs and this may be linked to fire intensity (e.g. Gould *et al.* 2007, p. 117; see Chapter 2 also). McCarthy *et al*. (1999, p. 27) provide equivalent fuel loads for various components of the forest-fuel array so that they can be linked with fire behaviour guides. Thus an array intensity could be estimated. However, the focus of their method is to predict the probability of success of first suppression actions based on the overall fuel hazard rating.

Three groups of methods for assessing the effectiveness of fuel modification and decreasing fire intensity to aid suppression are suggested:

1. Analysis of case histories in which suppression is usually explicit
2. Scenario modelling in which the reduction in the extent of unplanned fires is the measure
3. Analysis of fire-area statistics in which the measure can be the same as in (2).

Group one includes:

- Probability modelling of the effectiveness of initial attack by firefighters (McCarthy and Tolhurst 1998)
- Direct observation of fire behaviour when it reaches an area that has been prescribed burnt (e.g. Grant and Wouters 1993)
- Probability modelling of the decline in the rate of heading-fire rate of spread and its consequences for suppression (McCarthy and Tolhurst 2001).

All these methods show an effect of fuel characteristics. The higher the fuel load (or score) and the Forest Fire Danger Index (FFDI) – a measure of weather effect on fire behaviour – the lower the chance of success, in general. Benefits for the few years after fire in forests and shrublands are summarised by Fernandes and Botelho (2003) as increasing the safety of firefighters, decreasing the extent of firefighting resources needed, moderating the overall suppression strategy and lessening the amount of mopping up.

In group two (the modelling of prescribed-burning scenarios in relation to the extent of unplanned fires) there are various levels of detail, including:

- Conceptual, graphical models (Bradstock *et al.* 1995; Cary and Bradstock 2003)
- Site-based models, such as those of Bradstock *et al*. (1998*b*), which use the number of days a fire was deemed to be uncontrollable as an index – a measure that includes weather, slope, aspect and suppression thresholds
- Full landscape depiction in a sophisticated GIS simulation – King (2004) and King *et al*. (2006) simulated fire-area frequency distributions with and without prescribed burning in south-west Tasmania, and found a decrease in unplanned fire areas as prescribed-fire area increased.

These methods provide useful ways of exploring options and comparing unplanned fire sizes, with varying proportions of the landscape being subject to fuel reduction by burning.

In group three (a set based on fire-regime ideas using landscape data) there are:

- Comparisons within a fire season – The effect in the monsoon tropics of burning fuel in early dry season (mostly of low intensity) on fires in late dry season (mostly of high intensity). Gill *et al*. (2000) showed that there was a trade-off between the area burnt in the early dry season with that burned in the late dry season in lowland Kakadu National Park, east of Darwin, Northern Territory irrespective of ignition sources.

- Comparisons between fire seasons – The effect of a prescribed burning program on the area burnt by unplanned fires (Lang and Gill 1997). In this case, caveats are usually necessary because of the changing nature of suppression capacity over time, systematic changes in proportions burnt by prescription, time lags in the system and other factors (Lang and Gill 1997). The results of Lang and Gill (1997) suggested that there may be a threshold level (10% per year) in which prescribed burning is most effective in reducing the extent of unplanned fire in south-western WA. The results of the extreme fires of January–February 2003 in south-eastern Australia show that the extent of prescribed burning in north-eastern Victoria (say 1 or 2% of the area per year) had little effect under such circumstances. Unless fuel-modification measures can restrict fire growth early, and unless sufficient suppression capacity is available and on-the-spot soon after ignition, large fires can result under conditions of pervasive drought, high FFDI and multiple ignitions.
- Comparisons between regions – The south-western forests of Australia are often regarded as exemplary in terms of their fire management, and south-eastern Australia compared unfavourably. More prescribed burning is practised in south-western forests, but there has been no formal comparison between regions as yet (Esplin *et al*. 2003, p. 113). It should be noted that the extent of prescribed burning has been systematically changing in the south-west for decades (Gill and Moore 1997).

Various performance measures for *agencies*, based on fire statistics, have been suggested (Esplin *et al*. 2003, chapter 11).

The results of these studies suggest that the effects of fuel modification vary according to the ignition, weather, fuel and suppression circumstances, but that prescribed burning generally increases the chances of successful suppression and decreases the amount burnt by unplanned fires. Adding to the extent of prescribed fire could alter the total amount burnt in various ways (Bradstock *et al.* 1995), including increasing it (King *et a*l. 2006), thereby altering the mean fire interval.

Effects of fires on the environment

The importance of prescribed fires for fire suppression and control has already been outlined. In reserves for nature conservation, such matters are important but attain added significance in view of their potential effects on the environment generally, and in particular the organisms that are part of it. As far as the environment is concerned, it does not matter by who or what a fire is lit; it does not matter whether it is prescribed, started by an arsonist or the result of a lightning strike. Rather, the immediate effects of a fire depend on the fire's properties (e.g. intensity, flame duration and flame height) and the state of the system at the time (and often soon after). Longer term effects are likely to depend on the history of fire events – the fire regime. Here, an introduction is given to the effects of fires and fire regimes on the environment.

Environment is a broad term. In this context it consists of:

- Biodiversity – the variety of life – technically, the diversity of organisms, genes and communities in natural locations
- Land – including soils, sedimentary deposits, rock types and terrain
- Air – chemical composition and physical characteristics
- Water – for humans as well as animals.

While each of these elements is important, the complexity of the biodiversity issue – the focus of this publication – needs to be emphasised. In Australia, there are over 20 000 species of native vascular plants (Orchard 2000), and a large and growing number of naturalised species (Groves *et al.* 2003). There are about 2000 species of terrestrial vertebrate animals (Ross 2000) and 200–300 000 species of invertebrates (Wells and Beesley 2000). These species, let alone others like the estimated 3000 species of lichens, 1500 bryophytes (McCarthy 2000) and 250 000 species of fungi (Grgurinovic 2000), interact with each other and their respective physical environments to various extents. Large numbers of nameless microbes exist too.

The focus here is on biodiversity at the species level, rather than on the gene or community, because species' conservation – avoiding extinction of indigenous organisms (Bradstock *et al.* 1995) – is the main imperative of conservation management. Conservation of genes is the conservation of populations of all species, while conservation of communities isthe conservation of species' associations while maintaining the habitat of all species.

Various authors have proposed that fires can pose a threat to species' persistence, but fires are often mentioned as one of a number of potential threats. In such cases, threats are worth noting, but are usually hypotheses rather than historically or experimentally verified possibilities. Over half of the 53 rare or threatened marsupial species are believed to be threatened by processes that include inappropriate fire regimes (Maxwell *et al*. 1996). Similarly, there are species of rodents (Lee 1995), reptiles (Cogger *et al.* 1993) and 55 species of birds (Garnett 1992) affected. A recent compilation on the Australian Alps found that there were 18 species – eight mammals, six birds, one reptile and three frogs – threatened by fire (Coyne 2001).

A counterpoint to the management goal of maintaining all species of indigenous organisms in a reserve is the aim of causing the local extinction of all exotic species. Intermediate positions on the place of exotic and indigenous organisms in relation to management in reserves also occur. Thus introduced biological control agents may be desirable in order to keep populations of weeds at low levels. Another example comes from the Top End of the Northern Territory where, in order to improve protection of infrastructure through the establishment of fuel breaks or buffer strips, the native *Sarga* (*Sorghum*) *intrans*, a dominant grassy fuel species, can be temporarily eliminated by burning during the wet season, rather than in the normal dry season (after Stocker and Sturtz 1966).

The immediate effects of fires can be related to the properties of the fires, as noted above, and the state of the environment at the time. Fire intensity represents the rate of heat release of a part of the fire perimeter and depends on rate of spread and fuel consumption (Byram 1959); it is a fire property that is associated with particular flame dimensions and effects such as leaf scorch (e.g. Cheney *et al.* 1992 for prescribed fires). Prescribed fires for broad-area fuel modification are usually of relatively low intensity, a property which can vary according to the manner in which a fire is lit – point or line, upslope or downslope, multiple or single ignitions at dispersed or concentrated locations (Tolhurst and Cheney 1999). High intensities are a feature of the forward edge of fires burning before a strong dry wind in long-unburnt forest-fuel arrays uphill; intensities may be much lower where the fire is burning against the wind and at night.

Fires can affect individual organisms and, collectively, populations, in any single event (e.g. see Gill 1981). Fires can kill individual organisms, cause seed to germinate or enhance flowering of some species. Fires can cause some plants to coppice or root sucker. Plants may become more palatable after fire and attract herbivores. There are many possibilities. The geographic scale of the fire event or the distance from an unburnt fire edge can even be important to some event effects (Gill 1998).

The effects of single fire events can set the ecosystem parameters for the next fire event and, in turn, the effects of that event can set the stage for the following event and so on. Previous fire events affect the current system state, and thereby have an influence on the effects of the next fire. Even where a suite of new species appear after fire (ephemerals) then disappear before the next fire (see Gill 1999*a*), the extent of seed set by these species and the extent of loss of seed viability before the next fire may be important to the effects of a fire event. More than one fire's effects needs to be understood if the effects of the individual event are to be understood completely. The chain of fire events is called a regime. The fire regime consists of (Gill 1975, 1981):

- Type of fire (above or below ground)
- Seasonality of fire
- Between-fire interval
- Fire intensity (or other fire property).

Note that the type of fire in this scheme is not whether or not the fire occurs in the crowns of trees (crown fires[2]); rather, the scheme distinguishes fires burning in peat (or duff) from all others. Fires burning in such substrates have different fire properties to above-ground fires in that smouldering is predominant and long lasting while flaming is rare (Miyanishi 2001). In the case of a peat fire in coastal Victoria, the peat burned for three months to a depth of about 2 m, 'the ground sank, the creek bank collapsed, and large areas of burnt peat were washed away' (Wark 1997). Peat or duff fires may cause ringbarking and death of woody plants (Gill 1995), and can destroy roots, seeds and rhizomes (Miyanishi 2001). As a result, plants establishing after fire do so from seeds and spores, or spread vegetatively into the area from the outside edges of the peat (Wark 1997).

Seasonality

An example of the effect of seasonality of burning was given, above, for the annual grass *Sarga* in the Northern Territory. In this case, all seeds in the soil–seed pool germinate in the early wet season when live adult plants are absent; if there is enough fuel to carry a fire, the grass seedlings can be killed. Other seasonal effects concerning seedlings are also known. For example, Burrows and Friend (1998) found that burning a south-western Australian Jarrah (*Eucalyptus marginata*) forest in autumn produced a higher seedling density than burning in spring, but had 'little affect on species composition, measured 12 months after fire'. Seasonality also affected seedling recruitment in Australian and South African shrublands (Bond and Van Wilgen 1996, p. 100), but this effect may be confined to Mediterranean-climate ecosystems (Whelan 1996, p. 100). In the Australian arid zone, however, recent research has shown that 'seedlings of woody species were significantly more abundant following summer than winter fires' (Wright and Clarke 2007).

Seasonality may have a strong affect on flowering (Bond and Van Wilgen 1996, p. 100–101), a recent example is Lamont *et al.* (2000) who reported an effect of season of burning on the flowering of the grasstree *Xanthorrhoea preissii* in south western WA.

Seasonality may also affect resprouting behaviour. To illustrate, there is the case of a resprouting mallee (*Eucalyptus* spp.) in arid western New South Wales. Frequent burning in autumn caused much higher mortality than frequent burning in spring (Noble 1997, p. 51), an interactive effect with between-fire interval.

A direct effect of season on fire on invertebrates in Victoria has been noted. It was recommended that prescribed burning in forests 'should be scheduled for autumn rather than spring to minimise adverse impacts on the overall invertebrate fauna inhabiting litter/upper soil' (Neumann 1992).

In the subalpine belt of south-eastern Australia, for areas within 300 m of the breeding sites of the endangered Corroboree Frog (*Pseudophryne corroboree*) – such as *Sphagnum* bogs – it has been recommended that any prescribed burning should not be carried out in autumn, as this is the time when frogs move away from these areas into the surrounding vegetation (Osborne 2001).

Between-fire Interval

Most explanations of fire-regime effects have been related to between-fire interval (see Chapter 5 also). If intervals fall short of reproductive age, then local extinction will result sooner or later if there is no other form of regeneration. If intervals are too long and a species with no stored seed relies on bare earth for establishment, then fire intervals greater than the life span will also cause local extinction.

There has been an increasing realisation that *variation* in between-fire interval, around a mean interval, may be important to some species. Indeed it is expected that in any fire regime there will be variation in all the components. The nature of variation in interval has been the most studied; this variation is explored in Chapter 5. However, it is worth noting that the variation in interval is being linked to biological responses (e.g. habitat of animals by Mackey *et al.* 2001), and that time since fire is a common way of expressing changes in biota (e.g. Gill 1999*a*). The nature of variation in interval and time since fire is explored in later chapters.

2 Crown fires are seen to be a function of fire intensity.

Fire intensity

Intensity is the usual fire property considered in relation to immediate fire effects. It reflects the rate of heat release and flame length (Byram 1959; Catchpole 2002), and therefore the exposure to, and direct effects on, plants and animals (Whelan *et al.* 2002). Crown scorch (and death of fire-sensitive plants) can be related to intensity (Dickinson and Johnson 2001), while exposed animals in trees would be affected to a similar extent to the leaves and crowns that they live among (Gill and Bradstock 1995*a*).

Intensity is not relevant for fires in peat where rates of spread are extremely slow. The length of time that seeds and roots are exposed to temperatures above lethal levels seems more appropriate, but the three-dimensional nature of the problem, along with the extremely slow rates of spread, make the determination of this difficult. The depth of peat burnt or proportion of depth burnt (some *c.f.* all) may be better indicators that reflect the length of time of exposure of organs to heat, and what the suitable substrates for plant regeneration are (Miyanishi 2001).

Effects of fire regimes

The effects of fires on plants and animals depend on a particular fire regime and whether they are adapted to it (Gill 1975). If they are adapted to a particular fire regime or set of regimes, this means that they are not adapted to the complement of possible regimes, and may go extinct if under their influence. Extinction may be local but could also be over a wider area if the regimes that cause extinction are widespread; local niches with different regimes may allow remnants to survive, however. Mistletoes can be locally eliminated by a single fire because their individuals are readily fire killed and they have no seed store; their local extinction may be reversed by seed dispersal by Mistletoe Birds (*Dicaeum hirundinaceum*) (Gill 1996).

In contrast to the extensive literature on the effects of fires on vascular plants and vertebrate animals (Text Box 3.2), there has been little research on the effects of fires on non-vascular plants. Results from the grasslands of Western Victoria suggest that the vascular and non-vascular flora 'respond in different ways to fire and this should be considered in … conservation planning' (Morgan 2004). In frequently burned grassland, Morgan (2004) found that there were fewer mat species (non-vascular plants e.g. mosses) in his grasslands, but he noted that vascular plant species' richness is usually higher in frequently burned areas. In forested south-western Australia, Robinson and Bougher (2003) found that different fire histories 'result in fungal communities with similar total diversity but different species composition'. Wark (1997) found that non-vascular plants were common colonisers after fire. In a western New South Wales eucalypt shrubland (mallee), Eldridge and Bradstock (1994) found that the ground coverage of cryptogams (algae, mosses and lichens) increased with time since fire, algae being the main early colonisers.

The effects of fire regimes on biodiversity are:

- Changing species' presence to the point of plant-population reduction (Bradstock *et al*. 1997) or even extinction (Gill and Bradstock 1995*a*); evidence of local extinction may be observed historically or directly through the killing by fire of seedlings establishing in new territory (e.g. *Callitris* noted by Leigh *et al*. 1989) or the eventual establishment of adult plants in an area after a change of fire regime.
- Changing organic aspects of habitat for native animals, such as forest structure (e.g. Catling 1991; Gill and Catling 2002) or litter structure (York 1999).
- Changing inorganic aspects of habitat for plants and animals (including fish and stream invertebrates) through changes in soil chemistry (such as pH), water chemistry and temperature, erosion and sedimentation, and changed soil-moisture regimes.
- Changing species' composition of plant and animal communities as a result of *(i),* above, and indirectly through various interactions (e.g. burning–grazing and the likelihood of invasions by, or proliferations of, exotic species).

When regimes change, the effects may take many iterations (a series of successive fires) to reveal ultimate changes (Gill and Bradstock 1995*a*). For example, a change in greater dominance of the native grass *Themeda triandra* compared with *Cymbopogon plurinodis*, another grass, occurred with a program of frequent burning in South Africa. The new quasi-equilibrium took about eight fires to be achieved in the 16 years of observation (Trollope 1996).

Our knowledge of the effects of fire regimes on the biota is growing, but is still at an early stage on a national scale in Australia. For example, explicit knowledge of the immediate responses of vascular plant species to fires is incomplete, although no recent census has been conducted. There is a wide variety of information on the direct and indirect effects of fire on animals in a variety of habitats, from streams to forest and desert. The complexity of the food web and the effects of fire regimes on individual components of it is a challenge to our understanding. Computer simulation is one way of exploring possible effects of a fire regime, including prescribed fires (e.g. Bradstock *et al.* 1998*a*, 1998*b*, 2005), especially in a time of dynamic change of climate, fuels or fire regime. Also, effects can be examined empirically using long-term fire experiments that are, by necessity, dominated by low-intensity prescribed fires (e.g. Tolhurst and Flinn (1992) provide a progress report on a range of studies related to the effects of prescribed fires on forest plots in Central Victoria).

Text Box 3.2 **Literature on the effects of fire regimes**

There is considerable literature on the effects of fires on biodiversity. Bradstock *et al.* (2002) and Abbott and Burrows (2003) are recent multi-author books that tackle the subject in Australia. Whelan (1995) is an Australian text on *The Ecology of Fire*. In the USA, there are substantial recent reviews of the effects of fires on fauna (Smith 2000), flora (Brown and Smith 2000) and air (Sandberg *et al.* 2002). A range of topics is considered in depth in *Forest Fires: Behavior and Ecological Effects* (Johnson and Miyanishi 2001). Bond and Van Wilgen (1996) give an authoritative account of the effects of fires on plants, based on their South African experience; while Goldammer and Furyaev (1996) report on Eurasian Boreal forests. There are a number of commentaries on the effects of fires and fire regimes on various Australian animal species, including invertebrates (Brown *et al.* 1998; and various chapters in Bradstock *et al.* 2002; especially Whelan *et al.* 2002; Abbott and Burrows 2003 and Anderson *et al.* 2003).

Despite many advances in our knowledge of fire regimes and their effects throughout the world, a conservation manager may still not know how local species in the local jurisdiction respond to fire regimes. There may be a general lack of local information and answers from one area do not necessarily apply to another (Williams *et al.* 1994). It behoves the local manager to observe effects of fire regimes on the local environment in light of the literature, and thereby draw and record conclusions.

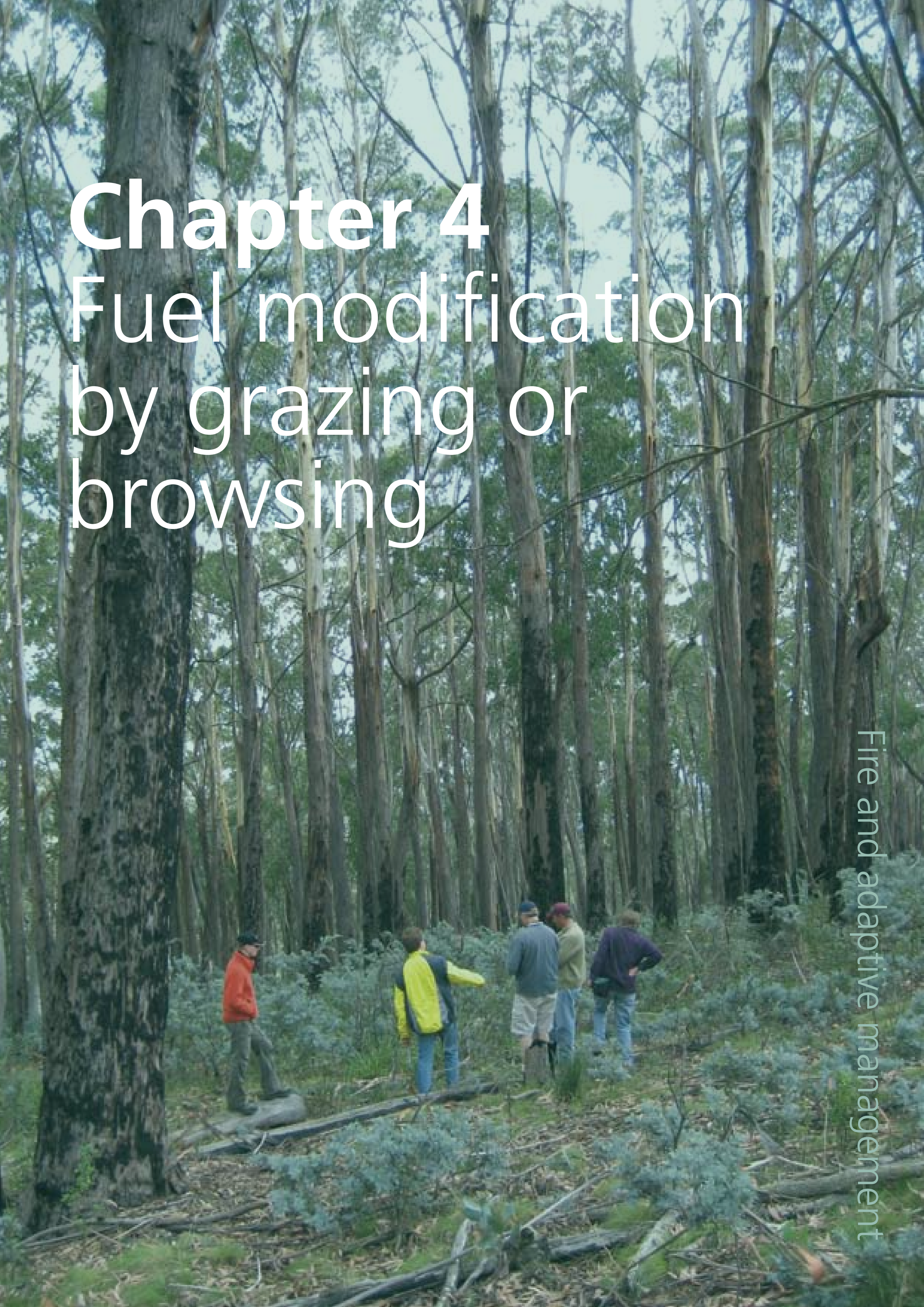
Conclusion

Prescribed fires are fires lit by managers of landscapes for specific purposes under specific conditions. These can be contrasted with other fires that are not for management purposes, and which may be ignited by unauthorised people or lightning. In forests, fires prescribed for broad-area fuel modification are generally of low intensity. In some places they may occur at shorter intervals than unplanned fires and occur in different seasons. In other areas they may be relatively rare. In some shrublands fires may burn with a high intensity whenever a fire will carry satisfactorily, and some grasslands may be burnt in mild weather during the usual fire season.

There are various methods for assessing a prescribed burning program, such as cost, influence on ease of suppression and ecological effects. Various possible performance criteria for assessing prescribed burning programs may be devised. Effectiveness may be viewed at a number of levels, such as the proportion of area burned, effect on fire suppression or immediate effect on the landscape. The ultimate measure of a program is its contribution to management aims.

In a conservation context, the effects of fires on indigenous and exotic organisms is of major interest; persistence of the indigenous species and extinction of the exotics may be desired. The fires occurring in a reserve, whether they have been ignited by people or natural means, have effects that are determined by the fire regime – type, interval, intensity and season – and the ecosystem context. Fire regimes may change as a result of prescribed burning, efficient suppression operations, reserve isolation or changing climates.

There literature on the effects of fire regimes is vast, but this does not necessarily answer local managers' needs for specific information. Information is accumulating rapidly and managers can act in ways that enhance local knowledge.



Chapter 4
Fuel modification by grazing or browsing

Chapter 4 Fuel modification by grazing or browsing

Introduction

Whether or not grazing by domestic stock has a role in conservation reserves is a longstanding issue in Australia, particularly in the southeast where grazing by sheep and cattle in the mountains has been controversial for at least 60 years. In 1943, domestic animal grazing was banned in what was soon to become Kosciuszko National Park (Clark 1992). Observation of detrimental effects of grazing (often accompanied by burning) in mountain environments go back to at least 1893 (Helms 1893), and elsewhere even before that, 'Since the country was stocked with sheep & cattle several native grasses have disappeared …' (1853 letter reproduced in Brown 1963). Domestic stock grazing in Victorian high-country national parks continued until recently (2005), amid considerable controversy. Grazing by domestic animals controversially continues in some conservation reserves outside the high country in the ACT (Canberra Nature Park; Plate 4.1; Lunt *et al.* 2007), Victoria (Wong *et al.* 2006) and undoubtedly elsewhere. In the forested escarpment country of north-eastern NSW, grazing by domestic animals was phased out as lands came under national park jurisdiction (Henderson and Keith 2002). Feral animals, such as goats, pigs, deer, rabbits, hares and horses, graze national parks and other reserves, along with native herbivores, such as kangaroos, wallabies, possums and potoroos. The grazing issue is relevant to conservation on farms, forests and pastoral properties.

Plate 4.1 Young exotic cattle grazing a conservation reserve in Canberra, ACT (Gill 2006). The subshrub in the foreground is the native *Hardenbergia violoacea*. The green graminoid in the foreground is the native *Lomandra* sp. The green grass at the base of the hill is the exotic *Phalaris aquatica*.

Some reserve managers may be tempted to use domestic stock as a means of fuel modification in reserves for biodiversity conservation. There are various reasons why this might be so, including:

- Reduction in grassy fuels and its perceived benefit to fire suppression and control without 'risky' prescribed burning operations
- Assisting drought-stricken, or other, local pastoralists ('good neighbour policy')
- A cultural legacy ('it's traditional')
- A way to reduce cover to allow indigenous species of plants to persist ('it works') (Kirkpatrick *et al.* 2005).

It may seem obvious that grazing would significantly reduce the fire proneness of vegetation, but this is not necessarily so if the dominant fuel element – as opposed to the grazing element – is not selected by the animals, even if it is grass: 'The most abundant and dominant herb is the snow grass … and as this is so obvious it is generally assumed that the sheep and cattle are grazing upon it. Close observation, however, shows that … the greater part of the rather wiry, tough, grassy tussock is not favoured by the animals except in years of intense drought. It is for this reason that most graziers, especially of sheep, have adopted a policy of regularly burning the snowgrass in the high country (Turner 1962).

Grazing for fuel reduction needs to be extensive and complete enough to have a significant effect on fire occurrence, intensity or rate of spread; recognising that fire can still spread across eaten out paddocks in high wind conditions (Cheney *et al*. 1998). In short, grazing that removes fuel biomass will reduce potential fire intensity (by definition). If it reduces grass height, it will reduce flame height (Cheney and Sullivan 1997, p. 24). Light grazing may have a relatively small effect on rate of spread, and even when eaten out, the rate of spread in grassland may be approximately 50% of that when there was no grazing (Cheney and Sullivan 1997, p. 39; Cheney *et al*. 1998).

A useful effect of grazing for the fire manager would be to reduce the proportion of dead matter (curing) in a pasture, which can assist in fire control. If the pasture can be kept green, and therefore moist, for longer periods than otherwise, then the chances of fire spread are reduced. This may depend on keeping pastures heavily grazed.

Grazing will have most influence on potential fuels where dominant species are highly palatable and accessible. In western New South Wales, grazing pressure has lead to the death of grasses and their replacement by woody plants formerly kept in check by repeated fires (Daly and Hodgkinson 1996). This agro-ecosystem seems to have changed from a largely fire–grass–shrub system to a sheep–grass–shrub system to a shrub system (see also Janssen *et al*. 2004). In the review of Lunt *et al*. (2007), the demise of dominant palatable chenopods in Riverina shrublands, and their replacement by grasslands, is described – the reverse of the previous example, as far as the sequence of life forms is concerned.

The presence of exotic grazing animals in a conservation reserve is usually seen to be contrary to the aim of biodiversity conservation. Removing exotic animals, domestic or feral, is usual when a reserve is declared. Thus there are programs to control the European Rabbit (*Oryctolagus cuniculus*) in conservation areas, presumably because it is exotic as well as a species causing damage to the assets of the area. Other feral animals are usually controlled as well. Furthermore, grazing has been identified as a threatening process (potentially causing extinction). Garnett and Crowley (2000) noted that grazing by sheep and cattle is a threat to 53% of mainland birds. Leigh *et al.* (1984) stated that 'Grazing by domestic, feral and native animals is believed to have been responsible for the presumed extinction of 31 species' (p. 35) of plants; 55 extant species were considered endangered by grazing in 1984. Eleven species of fauna were threatened by grazing in the Australian Alps according to Coyne (2001).

In its simplest form, opposition to grazing by domestic animals in conservation areas is a philosophical one. The presence of an exotic herbivore, especially if it is in large numbers, is the antithesis of the aim of conserving the indigenous biota. In its more complicated form, the issues centre on the:

- Environmental effects of grazing regimes in general (with potentially positive and/or negative effects)
- Possible extent of improvement of fire suppression due to the effects (including no significant effect) of grazing regimes on fuels
- Extent to which longer-term changes to vegetation type, animal habitat or landscape processes affect conservation quality or fuel type due to grazing regimes.

What is seen as the important environmental effects of grazing regimes varies widely according to the criteria applied. Pastoralists may primarily be interested in economic returns from livestock, but secondarily wish to maintain as many indigenous plant species as possible. Conservation managers may primarily be interested in conserving indigenous plants and animals, but see livestock as a way to reduce fuel loads. The primary concern here is with conservation of indigenous biodiversity in reserves, but the practice of grazing livestock in reserves draws attention to grazing effects, both positive and negative, in relation to fire-and-fuel management and biodiversity conservation.

The environmental effects of grazing are examined, as much of the literature is concerned with grazing versus no grazing rather than with the effects of grazing regimes. Grazing regimes are then defined in parallel with fire regimes – type of animal, intensity of grazing, intervals between grazing and seasonality of grazing, and their effects discussed.

Environmental effects of grazing

The environmental effects of grazing are readily observed. Differences in vegetation across fence lines are sometimes marked in terms of biomass, structure, composition and colour (due to curing, flowering, growth stage or composition). While the causes of these changes can be confounded with other treatments, such as fertilising, sowing, harrowing and herbiciding, enough evidence can be seen close to the fence to indicate when changes are largely due to grazing.

This is not an attempt to comprehensively review the literature on the effects of grazing; rather the purposes are to provide examples of grazing effects and to elicit principles. Recent overviews of the impacts of grazing by domestic animals on biodiversity in Australia include those by Dorrough *et al*. (2004), Lunt (2005) and Lunt *et al*. (2007). Caution is required because, as Dorrough *et al*. (2004) points out, 'We still have only a very basic understanding of the effects of different grazing strategies and pasture management on biodiversity'.

Information on the effects of grazing

The main sources of information about grazing effects are:

- Historical observations – 'A striking effect of the wool industry on Australian vegetation has been the virtual elimination of *T. [hemeda] australis*' (Moore 1962). Moore also noted, 'The trend under grazing [by exotic animals] is towards communities of short-lived alien species'. Woinarski and Fisher (2003) quote Finlayson (1945) as saying, 'It is not so much, however, that species are exterminated by the introduction of stock, though this has happened often enough, but the complex equilibrium which governs long established floras and faunas is drastically disturbed or even demolished altogether'. Lunney (2001) attributed the extinction of 24 medium-sized mammal species in western New South Wales during the first 60 years of settlement – 'the greatest period of mammal extinction in Australia in modern times' – to the presence of 'sheep, and the way that the land was managed for the export wool industry'.
- Historical legacies – quantitative studies of the effects of grazing practices (including burning) have been carried out in north-eastern New South Wales by Henderson and Keith (2002) and Tasker and Bradstock (2006); quantitative comparisons across different land tenures that have historically experienced different, unknown grazing regimes, such as paddocks, roadsides and reserves (e.g. Prober and Thiele 1995).
- Observations made across fence lines (Noble 1997) or at camping spots, sheltering spots, dung heaps (for certain species), watering points and yards compared with areas outside them. Moore (1962) noted that in sheep 'camps' there was an increase in the exotic genera *Marrubium* (Horehound), *Urtica* (Stinging Nettle) and *Onopordon* (Scotch Thistle). The effects of different historical grazing regimes have manifested in spring in Canberra, ACT, where massive blue flowerings of the exotic Patterson's Curse (*Echium plantagineum*) can occur in paddocks grazed by domestic animals. In contrast, across the fence and along the adjacent roadside, numerous yellow flowers of the native *Bulbine bulbosa* and the absence of *Echium* have been seen (Plate 4.2). In farm surveys Reseigh *et al*. (2003) found that 'the never grazed and infrequently grazed sites had significantly higher native species richness than grazed sites'. Driessen *et al*. (1990, pp. 75–76) showed that the greater the intensity of grazing by stock, the fewer the number of indigenous Tasmanian Bettongs (*Bettongia gaimardi*) found (as indicated by their diggings).
- Direct observation of diet – bite rates for sheep and goats in a Mediterranean shrubland were directly observed by Papachristou (1997).
- Measurements made at different distances from water sources (Landsberg *et al*. 1997) or from intensively grazed areas, such as near rabbit burrows (Leigh *et al*. 1989).
- Experiments – 'Rabbits reduced the cover and biomass of 39 species of forb, in some cases to zero' (Leigh *et al*. 1987; Plate 4.3). In the studies analysed by Kemp *et al*. (2003), 'where the herbage mass was maintained between 2 and 4 t DM/ha [dry matter per hectare] then species were maintained and productivity [for sheep] was optimised'.

Dung counts may be used to help quantify the grazing intensities of different animals (Allcock and Hik 2004; Archibald *et al*. 2005; Kirkpatrick *et al*. 2005), especially when uncontrolled numbers of native and feral animals are involved, as well as controlled numbers of domestic stock.

Plate 4.2 The yellow native *Bulbine bulbosa* predominates over the exotic Patterson's Curse (*Echium plantagineum*) on the roadside (foreground), while the reverse is true over the fence in the horse-grazed paddock (background) (Gill 2003).

Just what grazing is needs to be outlined. Grazing encompasses the relevant actions of herbivorous animals – eating, trampling, lying down, pawing or digging the ground, rubbing against trees, eating bark (ringbarking), defecating, urinating and carrying seeds externally or internally, all with flow-on effects to biodiversity, resource distribution (e.g. by camping) and water quality (e.g. Alexiou 1983; Friedel and James 1995; Robertson 1997; Noble 1997, p. 47). Needless to say, different animals have different grazing characteristics. Added to the direct effects of grazing are the effects of infrastructure associated with grazing, such as fences, watering points, yards and tracks. These have their own effects on the environment or on aims of management, such as recreation and biodiversity conservation. In some places, the use of horses and ground vehicles for mustering may have an effect.

Plate 4.3 Grazing exclusion plot (on right) established in 1989 at Wilsons Promontory to study the impact of native and introduced herbivore grazing pressure in Coastal Grassy Woodland communities. The abundant *themdia* sp and other palatable species present inside the exclusion plot have been heavily suppressed outside by grazing (Whelan 2008).

Indirect effects of grazing

There are less direct effects of grazing. Grazing can affect biodiversity by reducing soil productivity (Friedel and James 1995). Excessive grazing can remove vegetative cover, thereby allowing wind erosion, loss of productivity, loss of soil seed and loss of safe sites for seed lodgement (see Noble and Grice 2002). Carr (1962) noted that shrubs in the high country of Victoria were not affected by grazing per se, but by trampling. Eldridge and Rath (2002) found that kangaroos in semi-arid woodland in western New South Wales affected soil properties by creating frequent shallow hollows – hip holes – thereby considered 'important elements in the maintenance of heterogeneity'. Hobbs (2001), in reviewing practices in south-western Australia, found evidence for substantial effects of trampling by livestock, as determined by increased soil density and decreased infiltration. The presence of cattle in the high country of south-eastern Australia, through the alteration of drainage patterns in bogs and wetlands, may have altered the habitat of indigenous plants and animals (Ashton and Williams 1989). The effects of naturalised Water Buffalo (*Bubalis bubalis*) in the Northern Territory included 'accelerated soil erosion, channelling of floodwaters, saltwater intrusion … and reduction in the diversity and abundance of wetland flora and fauna …' (Corbett 2004).

Seed transport is another less direct aspect of grazing. Jolaosho *et al.* (2006) contrasted the seed carriage of cattle, sheep and goats in a field experiment on native pastures in Nigeria – in terms of numbers of seeds per gram dry weight of faeces, germination rates of voided seeds and numbers of species of voided seeds. Cattle had the highest number of seeds per gram dry weight of faeces (1.8), and sheep the least (0.4). However, germination rates were least for cattle (5%) and similar for sheep and goats (32 and 28% respectively). Cattle excreted the highest richness of seeds (13 spp.), while sheep and goats voided nine and five species, respectively. Thus effects of grazing can be more substantial and subtle than the effects of herbivory.

Effects of grazing on vegetation types

For various Australian vegetation types, the negative effects of grazing by livestock or feral vertebrates have been described as follows:

- Mallee vegetation – Parsons (1981) stated, 'It is clear … that the most serious single threat to the future regeneration and conservation of mallee vegetation is the destruction of seedlings by grazing [feral] rabbits. Unless controlled, rabbits will certainly lead to the eventual disappearance of numerous species and communities'.
- Desert communities – degradation is attributed to 'over-use … by domestic livestock as well as by uncontrolled grazing of feral animals, such as rabbits, horses, donkeys, camels and goats' (Cunningham 1981).
- Alpine/subalpine vegetation – 'the practice of summer grazing of sheep, cattle and occasionally horses for a period of more than 50 years, combined with the associated practice of burning off' had consequences such that 'Plant cover was reduced, soil erosion promoted and some plant species … became rare and disappeared locally' (Costin 1981).
- Natural grasslands – Groves and Williams (1981) suggested that 'Substantial differences in stocking rates (sheep and cattle per unit area per year) cause some differences in botanical composition but usage has to be severe and prolonged for permanent changes in vegetation to occur'.
- South-western Australian woodlands – Hobbs (2001), in reviewing the effects of livestock grazing, pointed out that the 'most obvious changes were in understorey composition, with a reduction or removal of the native shrubs and herbaceous perennials' and increases in cover of non-native plants.
- South-eastern Australian grassy woodlands – Prober and Thiele (1995) concluded that, 'For the maintenance of native plant diversity and composition in little-grazed remnants, it is critical that livestock grazing continues to be excluded'.
- Eucalypt-dominated dry forests and woodlands in Victoria – regular livestock grazing reduced the probability of finding eucalypt regeneration (Dorrough and Moxham 2005).

More generally, it has been noted that, 'Historically, stock grazing has caused enormous damage to many Australian ecosystems' and 'there is no scientific dispute that grazing stock continue to degrade ecological values in other areas, such as alpine grasslands in Victoria' (Lunt 2005, p. 1). However, Leigh (1994) suggested that, 'Too often, conclusions about the alleged deleterious effects of grazing are drawn without there being adequate controls in the form of ungrazed plants'.

Effects of grazing through diet selection

'Grazing of native plants by introduced livestock is … highly selective as regards the sites and the species of plants …' (Costin 1983).

Importation of exotic grazing animals into a reserve, especially a grassy one, can lead to a quick reduction in herbaceous fuel load (non-woody biomass), according to the density of animals per hectare and the palatability of the dominants. A first impression would be that such a practice would compromise biodiversity values. However, just what the effects of grazing would be depends on the diet of the animals, in the first instance. While there are many processes in the plant-life cycle that can be affected by grazing animals – and many animal habitat features that can be affected directly and indirectly – attention is drawn here to the immediate effects of grazing through diet selection. Animals are selective feeders to the extent that some herbivores may even prefer certain parts of a plant above others, or plants of a certain sex within a population of a single species, as in the case of sheep feeding on the native Bladder Saltbush, *Atriplex vesicaria* (Walsh *et al*. 2005). If the animals feed preferentially on certain flowers or seeds, they may disadvantage one species of plant over another. By reducing biomass of one species, they may allow another to out-compete it. By removing seedlings of trees, they may influence the gross structure of the vegetation.

Some examples of plant species' selection follow:

- Between broad categories of plants – Australian Swamp Wallabies (*Wallabia bicolour*) have a reputation as browsers (i.e. feeding on shrubs) rather than grazers (i.e. feeding on herbaceous species). So too do feral goats (*Capra hircus*) (Henzell 2004), whereas feral camels (*Camelus dromedarius*) in the arid zone prefer both shrubs and forbs to grasses (Dorges and Heucke 1995). An extreme example is in parts of western New South Wales, where the grasses have been completely eliminated by grazing and unpalatable woody plants now dominate; fires which may have kept shrubs at relatively low cover in the past became much rarer, thereby allowing the shrubs to persist and dominate at the expense of the grasses (Friedel and James 1997; Noble 1997, p. 39). Shrub-grass relativities can be altered.
- Between plant species by livestock – 'Sheep showed a high degree of species selection when grazed in different seasons on 3 contrasting pastures' (Leigh and Holgate 1978). Free-ranging cattle in the United States had preferences in the following order: Smooth Bromegrass (*Bromus inermis*) > Crested Wheatgrass (*Agropyron desertorum*) > Western Wheatgrass (*Pascopyrum smithii*) > 'native range' plants (Fehmi *et al.* 2002). In Israel, the 'changeover from goat to cattle grazing encouraged the expansion and invasion of thorny shrubs formerly eaten by goats' (Bonneh *et al*. 2004).
- Between plant species within native vegetation – Leigh *et al*. (1987) provided a list of palatable and unpalatable species, from the point of view of the European Rabbit, in the high country of New South Wales. Included in the palatable-forb category were 32 species, including *Brachycome* spp. (daisies), *Bulbine bulbosa* (a lily), a number of orchids, *Wahlenbergia* spp. (Bluebells) and some exotic species. Among the 54 non-palatable forbs were native *Geranium* spp., *Ranunculus* spp. (Buttercups) and *Stylidium graminifolium* (Trigger Plant). Leigh *et al*. (1989) provide lists of species in similar categories for central-western New South Wales.

Herbivores do not recognise plants as native or exotic; they choose food in terms of palatability (see Lunt 2005 p. 20, from Vallentine 2001, for characteristics of palatability) and accessibility. Some native and exotic plants are toxic (Everist 1981) to some animals, and may therefore affect their selection. There is no reason to expect that certain native or exotic animals will select exotic weeds over natives or dominant fuel species over other less common species. The abundances of plant species in relation to their cover or biomass, in addition to their palatabilities, will help determine the effects of grazing on potential fuels.

In Figure 4.1, the plant species are arranged in order of cover, presumed to be proportional to biomass, all of which is assumed to be fuel in this grassland example. The curve represents very high dominance of one species and drastically lower cover classes for other species. In this grassland context, the most abundant species may be referred to as a 'fuel species' (Gill 1999*a*). At least one widespread grassland species in Australia – *Themeda australis* – is highly palatable (e.g. Allcock and Hik 2004) and a dominant fuel species; while at least one exotic dominant – Serrated Tussock (*Nassella trichotoma*) – is unpalatable (Lunt 2005 p. 24), but also a fuel species. The dominant grass species changes spatially, sometimes within short distances.

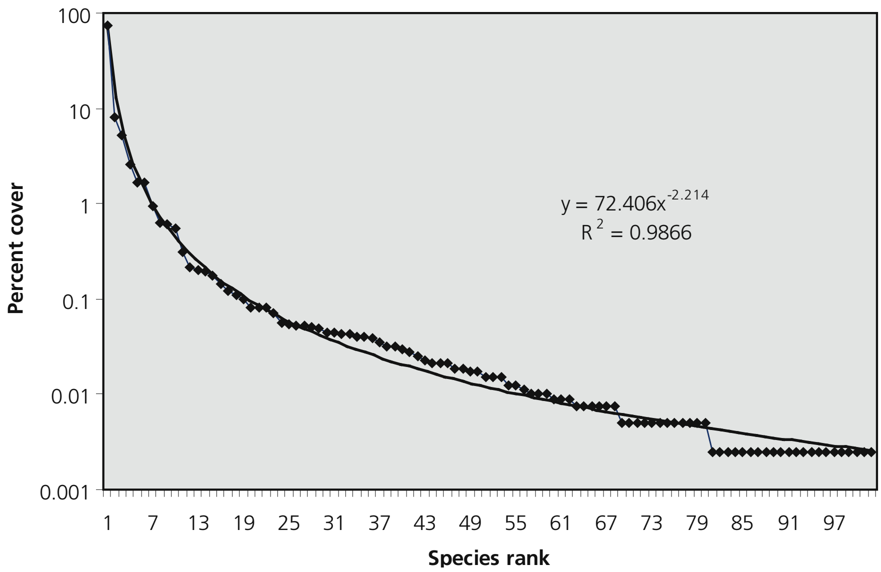


Figure 4.1 102 species of a montane grassland in Namadgi National Park, ACT, arranged in order of their overlapping cover values; 95 species have less than 1% cover (Graph courtesy of Dr R Godfrey, CSIRO, 2006). Note the logarithmic Y-axis.

Relatively small rare herbs in the tail of a frequency distribution, such as that in Figure 4.1, in the herbivore context, may be most readily envisaged as locally rare. The dominant species, in the fuel context, may be regarded as having ubiquitous occurrence. In a conservation context, at the landscape scale, species may persist because of heterogeneity of habitat. For example, niches for herbs desired by grazing animals may occur among unpalatable grasses or prickly shrubs or in rocky terrain where a herbivore's foraging rewards are less pronounced. While this may be the case in some landscapes, Landsberg *et al.* (2002), working in north-west South Australia, found that selectively chosen and uncommon or short-lived species, 'are more likely to decline everywhere', not just near the more heavily grazed water points.

Grazing regimes

In this section, the concept of the grazing regime is explained. It is addressed at the level of a plant-sized patch or point in the landscape – as opposed to an area e.g. paddock. This reflects the scale of the animal's immediate influence through herbivory and recognises that some plants or plant parts are palatable while others are not.

The components of a grazing regime are:

- Type of animal
- Intensity of herbivory (e.g. amount or proportion of plant removal on each grazing occasion)
- Interval between disturbances (e.g. time between consuming the shoots of the same grassy plant)
- Seasonality (of shoot removal).

The parallel with fire regimes may be apparent. Interactions between grazing regimes and the environment in which the animals live are to be expected. The grazing-regime approach is different to that of traditional farm experiments, where intensity of grazing is given as a 'stocking rate' for a paddock as a whole, rather than for a point within a paddock (as above). Point approaches allow for diet selection and spatial variation to be expressed through probability measures. When the concern is with hundreds of plant species within an area, rather than a general effect of feed on the growth of animals, or fuel for fires, a point approach seems appropriate.

Type of animal

Differences in diet have been identified between introduced animals, such as Alpacas (*Lama pacos*) and sheep (McGregor 2002); native and introduced species, such as the native Bettong (*Bettongia lesueur*) and the introduced Rabbit (*Oryctolagus cuniculus*) (Robley *et al*. 2001); and native species, such as Wallabies (Nailtail Wallaby, *Onychogalea fraenata*, and Black-striped Wallaby *Macropis dorsalis*) (Evans and Jarman 1999). 'Cattle, goats, kangaroos and rabbits have a different pattern of selectivity [and] have different impacts on the vegetation …' (Wilson and Harrington 1984).

Similarities, or partial similarities, in diet may lead to competition for food. A significant conservation example is that of the endangered Golden-shouldered Parrot (*Psephotus chrysopterygius*) on Cape York Peninsular in Queensland: Cockatoo Grass (*Schizachyrium* spp.), an important seed source for this parrot, is selectively targeted by feral pigs and is also subject to overgrazing by cattle, thereby depleting seed production (Crowley *et al*. 2003, p. 41–42).

While there may be an assumption that only vertebrates need to be considered in terms of removal of grassy fuels, invertebrates (e.g. termites) can also be important consumers with a similar aggregate body weight to cattle in the arid zone (Watson *et al*. 1973). Grasshoppers can be significant herbivores at times also.

Allcock and Hik (2004) studied the effects of sheep and cattle (exotic), rabbits (exotic) and kangaroos (native) on the survival and growth of transplanted seedlings of two species of tree and one of grass, over a period of almost three years. They summed up their results, and perhaps the topic generally, as, 'While our study was limited in its ability to tease apart the individual effects of each herbivore species, it provided substantial evidence that herbivore identity is important in determining the outcomes of herbivory in multi-species communities, and that these effects vary with habitat characteristics associated with grassland and woodland communities'.

Intensity of grazing

The intensity of grazing at paddock scale may be controlled by adding or removing domestic animals; culling feral herbivores, such as rabbits, deer, pigs, goats, sheep, horses, buffalo, cattle, donkeys and camels; and culling native species, such as kangaroos. A land-use combination of farms and reserves may increase numbers of [native] kangaroos in reserves, a circumstance that has the potential to increase grazing intensity in the reserve. Such increased intensity may occur due to the permanent availability of water in paddocks or the extra feed there. Native animals, such as kangaroos, can potentially overgraze native-plant communities like those in some of the Mallee region of Victoria, but the situation can also be confounded due to the presence of rabbits and goats (Sendell 1995).

Kemp *et al*. (2003) assessed the results of ten livestock grazing experiments in southern Australia, with up to seven treatments examined over three to four years. Numbers of 'native [plant] species increased by 1 or 2 ... where the grasslands were less intensively used … but decreased in more heavily grazed treatments'. In the high country of south-eastern Australia, Leigh *et al.* (1987) found that 'Rabbits reduced the cover and biomass of 39 species of forb, in some cases to zero'.

The effects of removal of grazing are also pertinent to the understanding of the effects of grazing intensity. Noble (1997, p. 61) documents the 'substantial increase in species richness, especially in terms of perennial grass species … achieved by excluding all vertebrate grazing animals [sheep and kangaroos presumably] for the last twenty years confirming the overwhelming effects of past grazing history' at a site in western New South Wales. Lunt (2005, p. 42) provides examples of a number of such studies, and points out that palatable weed species may also increase when grazing pressure is removed.

When indigenous species conservation is the aim of management and diets are selective, the intensity of grazing needs to be determined in relation to the species of concern. In relation to fuel, intensity of grazing needs to be determined with respect to species which contribute the bulk of the fuel ('fuel species'; Gill 1999*a*), and at larger scales than for relatively rare species. In simple terms, when the palatable species have been consumed, animals turn their attention to less and less palatable species, 'when these minor species had been consumed, sheep were forced to eat the dominant grasses' (Leigh and Holgate 1978). The intensity or grazing varies with palatability and may be seen to apply at a point scale. In Leigh and Holgate's example, at first the grazing intensity was relatively high for the minor species, but very low for the dominant species.

Interval between grazing events

At the paddock scale, the interval between grazing events can be measured by the timing of livestock additions and removals each year. The interval may be rated as short term or seasonal, measured in days or months. As with Tscharke (2001) – noted by Lunt (2005) – long-term outcomes are unavailable as yet. The interval for an individual resprouting plant – considered to be at a point – may be measured in days, weeks, months or years. As with intensity, interval may be determined differently according to the purpose of the measurement.

If the dominant species, a fuel species, of a natural sward is the most palatable to domestic stock, while the lesser ranked species are less palatable, then the periodic removal or reduction of the shoots of the dominant may be beneficial for conservation and fuel minimisation. This may have been the case with *Themeda* pastures in Victoria where periodic, high-frequency (short-interval) burning favours the expression of a variety of herb species, while not eliminating the dominant (Lunt and Morgan 2002). However, the palatability of *Themeda* to domestic stock, probably under continuous grazing, may have led to its demise over wide areas, as surmised by Moore (1962) and supported by subsequent studies, including those of Fensham (1998) and Allcock and Hik (2004).

The effects on palatable and non-palatable (fuel) species at point, patch or paddock scales merge when large numbers of animals are imposed on the vegetation for short periods. In this case, the shoots of the more palatable plants are quickly removed. The animals then concentrate on the less palatable materials and may remove them before the shoots of the palatable plants recover sufficiently to again be available for grazing. It has been argued that this sort of treatment would have the minimum adverse conservation effect and maximum fuel-reduction effect, but the resilience and relative palatabilities of the species present in any local area would need to be taken into account. Furthermore, other potential adverse effects associated with the presence of the animals would need to be considered, including the creation of areas of nutrient concentration ('camps'), exposure of the soil to erosion, ringbarking of trees by rubbing, trampling effects, fouling of water bodies and the promotion of unpalatable or highly resilient exotic species.

Seasonality of grazing

'*Poa annua* [an exotic grass] was positively affected by past grazing in spring, and negatively affected by past grazing in summer' (Kirkpatrick *et al.* 2005).

'Seasonal effects' may be seen as an interaction between herbivores of various type and the feed on offer. Different plant species have different palatabilities and accessibilities at different stages of the year and at different times in the life cycle. Young succulent shoots of grasses have a higher palatability to many animals than coarse dead stalks. Flower heads or seeds that appear seasonally can be preferentially selected (as by rabbits in subalpine vegetation of New South Wales, see Leigh *et al*. 1987), a circumstance that may have consequences to plant reproduction in the short- and long-term. Thus if animals are introduced to an area for short lengths of time at different times of the year, the results could be different for different plant species.

Whalley and Lodge (1986) set out to manipulate pasture species' composition through livestock grazing in order to favour a 'desirable' native grass in a grazing system – *Danthonia linkii*. They succeeded by using heavy summer and early autumn grazing and then light winter and spring grazing. Sheep were more effective than cattle. The idea was to manipulate the grazing regime to minimise adverse effects on the desirable *Danthonia* and maximise adverse effects on an undesirable native grass, *Aristida ramosa*.

Conservation under grazing and burning regimes

Can livestock grazing enhance conservation?

Despite the many examples given of detrimental effects of livestock on the environment, there are several examples in which livestock grazing has been shown to enhance plant-species biodiversity in some Australian ecosystems (e.g. Fensham 1998; Kirkpatrick *et al*. 2005; see also the reviews of Lunt 2005 and Lunt *et al.* 2007). Does this mean that grazing by livestock should be adopted for conservation in reserves, or for fuel management there? In answering this, it is important to consider the grazing regime and it is worth remembering that Australia's native plants had not been exposed to livestock prior to white settlement.

The following hypothesis may apply to the situations where enhanced species' diversity resulted from livestock grazing (see also below):

1. The dominant species of a productive sward eliminates all but the soil-seed stores of a number of species that become suppressed as the dead material of the dominant accumulates in the absence of disturbance.
2. The shoots of the dominant species are highly palatable to many species of herbivore.
3. By grazing the sward lightly, the dominant species is the most affected in these systems, in terms of relative reduction in shoot biomass.
4. Reduction of the biomass of the dominant reduces its competitive ability, thereby allowing the suppressed species to germinate and establish.

Support for the above hypothesis is that positive results for conservation under livestock grazing are obtained when pastures are productive and have a highly dominant grass present. With the lessening of dominance due to grazing, other members of the plant community can express themselves (Fensham 1998). This is summarised by Lunt *et al*. (2007), 'The most common circumstance in which it [livestock grazing] may be a useful management tool for conservation purposes is where it controls the biomass of existing potentially dominant, grazing-sensitive, palatable plants (native or exotic) on productive soils'.

Some may use the above hypothesis to justify the use of livestock in reserves for nature conservation, despite the fact that livestock are exotic and exotics, by definition, are not part of the indigenous fauna. However, there are possible alternatives that may be more consistent with the ideals of indigenous-species conservation, namely: (i) any positive effects on conservation values due to grazing by livestock may possibly be mimicked by native herbivores, and (ii) it is possible that certain fire regimes could reduce the biomass of the dominant to the extent that suppressed species are expressed.

With respect to the use of fire regimes as an alternative to using livestock grazing, research at Boggy Plain in Kakadu National Park in the Northern Territory is instructive (see Davidson 2005). There, removal of the feral Asian Water Buffalo (*Bubalus bubalis*) allowed the native grass *Hymenachne acutigluma* to spread and dominate the wetland, and apparently cause the disappearance of many other plant species. However, returning to a traditional pattern of repeated burning, as practised by Aboriginal people, reduced the dominance of the *Hymenachne* and increased biodiversity.

Interactions between grazing and burning regimes

Burning may stimulate changes in animal habits or diets, as mentioned above, or populations. Habits may change because native animals are often faithful to home ranges – the area in which they feed and breed. If part of the home range is burnt, there can be increased grazing and browsing pressure on it, as the regenerating plants are usually more palatable than the mature vegetation. Animals may shelter in unburnt edges and feed in burnt areas (see Gill 1998). A concentration of insect herbivores may also occur at burnt edges (Knight and Holt 2005). Population changes may be significant. Leigh *et al*. (1987) noted that 'feral rabbit populations survived and multiplied on burnt areas but decreased on areas left unburnt'.

Langevelde *et al*. (2003) explained the grazing–fire interaction in African savannas as follows. Burning areas of grass attract herbivorous animals when regeneration begins. This increases the intensity of grazing locally, leading to a decreased fire intensity. In turn, lower fire intensities cause less damage to trees and thereby enhance woody vegetation. The same authors point out that browsing animals can cause a trend in the opposite direction – enhancing the damaging effects of fires on trees by reducing woody biomass and allowing compensatory grass growth that can support a more intense fire, which can lead to greater damage to trees and more grass. In higher rainfall *Acacia* shrublands of eastern Australia, Hodgkinson (2002) noted that over the last 120 years, where pastoralists have increased grazing pressure and suppressed fires, woody plants have increased and grassy ones have decreased.

The interactions between fires, woody plants and grass are likely to take place at a number of scales. Using a system of easily moved grazing exclosures and a program of documentation of effects may allow managers to discern changes due to herbivores that are otherwise very difficult to see (e.g. Sandell 1995). Also, the effects and significance of interactions between fire and grazing or browsing regimes will depend on their components and how they vary through time. Stocking rates and intensity vary naturally, but can also be manipulated. The same is true of fire regimes. Local susceptibilities of trees and shrubs, periodic changes in seasonal productivity and curing of grasses, and variations in fire weather and behaviour set different natural contexts for the expression and significance of the interactions mentioned. Mixes of animal species – feral, native and domestic – through time may cause different effects. A system of patch change, varying both spatially and temporally, may be envisaged across a landscape, but how stable this is in isolated conservation reserves is an open question.

Discussion and conclusion

The addition of livestock to reserves, removal of charismatic feral animals (such as horses and deer from reserves), or culling of native animals (such as kangaroos or koalas in reserves), raises strong passions in communities of south-eastern Australia. On the other hand, removal of feral rabbits is uncontroversial. This situation occurs despite the fact that, in part, overgrazing can be a common element in all cases.

Interventions, such as adding livestock, burning and mowing can affect fuel loads, fuel structure and fuel moisture by altering the proportion of live and dead material. Overgrazing can cause a shift to another system state, such as from a grassy community to a shrubby one (Janssen *et al.* 2004), thereby altering fuel conditions and fire regimes.

The effects of grazing regimes on grassland fuels and fuel species can be quite varied. Williams *et al*. (2006) investigated the slogan 'alpine grazing prevents blazing' after the widespread 2003 fires in Victoria. They found that 'There was no statistically significant difference between grazed [by cattle] and ungrazed areas in the proportion of points burnt'. There is no comparable data known to the author for lowland grasslands, but personal observation shows that fires can burn across eaten-out paddocks, albeit at reduced intensity given that biomass is low and rates of spread are reduced compared with ungrazed pastures (Cheney and Sullivan 1997, p. 39). In shrubby communities of the alpine area, Williams *et al*. (2006) found no reduction in the severity of the fire due to grazing.

Lunt (2005 p. 37) summarises the research to date on the effects of historic livestock grazing on native pastures of *Themeda* sp., *Stipa aristiglumis* and *Poa caespitosa* (Stage 1): from these species, the trend of change is through communities of *Austrodanthonia* spp. and *Austrostipa* spp. (Stages 2 and 3) to exotic species (Stages 4 and 5). Using this as a basis, he draws up another model that indicates how contemporary grazing intensity might affect small-scale diversity (p. 27). This model suggests that if the grassland is in Stage 1, increasing grazing intensity will reduce diversity; if in Stage 5, decreasing grazing intensity will either leave diversity the same or increase it; while, if in Stage 3, changes in grazing intensity may affect diversity up or down, with decreases and increases in grazing intensity respectively.

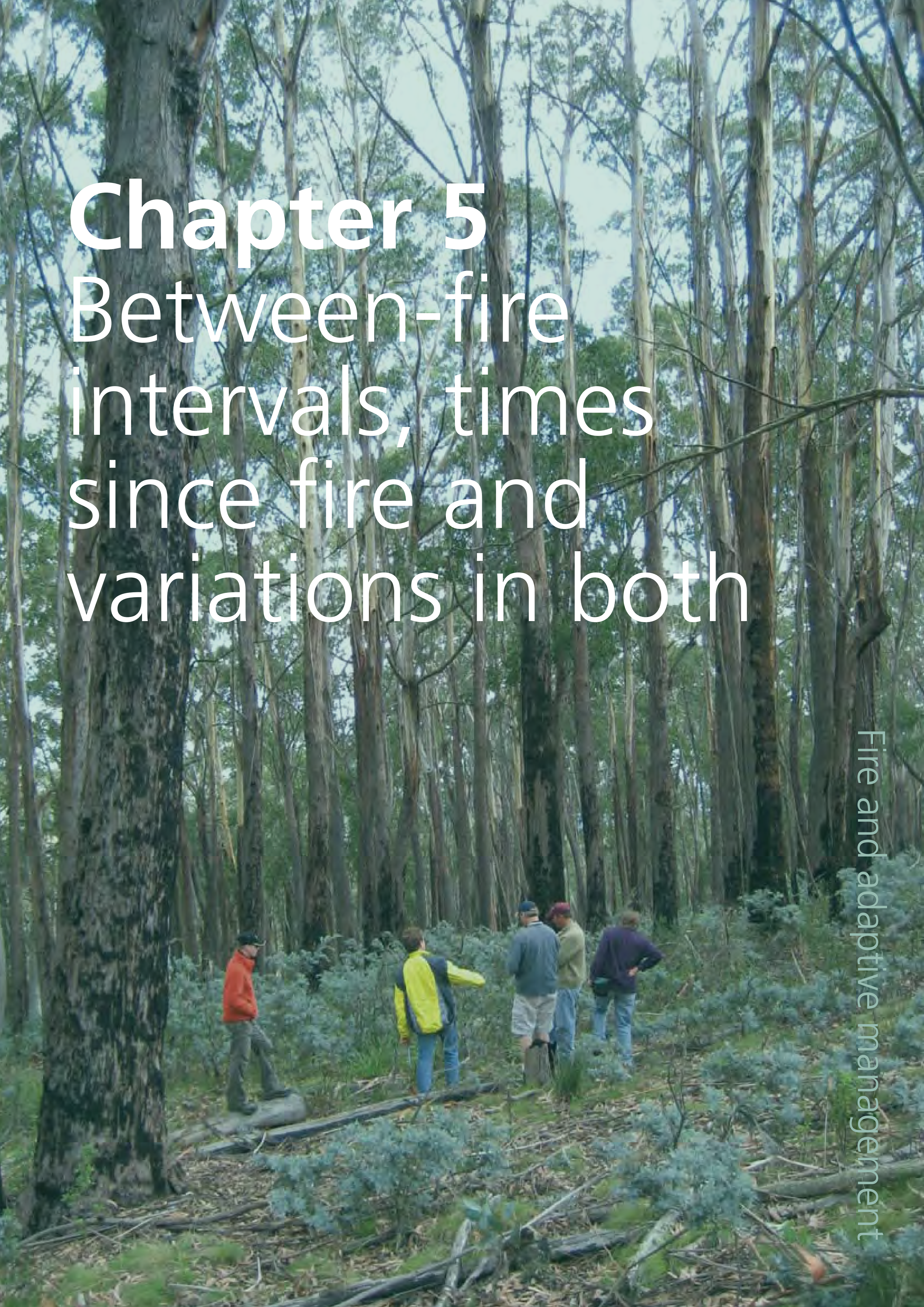
Lunt's (2005) model is useful as a basis for discussion. It is necessarily quite simple, relies only on grazing intensity and does not include the effects of different types of animals, different intervals between grazing and varying seasonality of grazing, together with the length of time that the regime has been imposed. The challenge for managers contemplating this model is to work out where their landscapes, or more likely parts of their landscapes, fit within the model and to what extent the model (like all models) meets their needs. For example, in Cooleman Ridge Reserve (part of Canberra Nature Park in the ACT and an area well-known to the author), there are gradients in grass communities in which the exotic annual Wild Oats (*Avena* sp.) is common on some ridges, natives such as *Stipa* are common mid slope and the exotic *Phalaris aquatica* often dominates lower slopes. Would the addition of livestock decrease the exotics and enhance the natives? A fence-line effect, associated with a farm grazed mostly by horses, suggests that *Phalaris* could be greatly diminished by horse grazing, while native *Stipa* would be enhanced (on lower slopes). However, a proliferation of exotic Patterson's Curse (*Echium plantagineum*) may accompany the change. At the same time, would adding livestock cause the diversity to decline on the mid slopes, where the native species are common? If the perennial *Phalaris* is to be removed, intervention of some sort seems necessary. If removal was successful, there may not be a soil-seed pool of native species to replace it and there is the possibility that replacement might be by annual exotics, such as *Echium*. The model suggests that the exotic annual grasses may persist with added grazing, but again the regime is important to what happens.

Grazing regimes based on exotic animals, such as livestock, can affect grassland biodiversity, as is evident historically. For reserves, removal of livestock can enhance biodiversity outcomes, while retention also seems to maintain it in some cases. The cautious approach, encouraged here, is to avoid livestock grazing in reserves where biodiversity is the main aim of management. There seems to be little known of grazing-regime effects over long periods, so no effect in a short-term experiment does not necessarily mean no effect in the long-term; recovery processes may take decades and be hard to discern. Positive effects on particular species or even communities may be achieved by changing the fire regime, rather than retaining a particular grazing regime (e.g. see Davidson 2005). The less cautious approach is to introduce livestock. Such a policy implies an obligation to assiduously measure, record and report the grazing-regime effects on biodiversity of native organisms, because of the lack of such animals in pre-European settlement of Australia; and that livestock grazing regimes can replace certain fire regimes. An intermediate position is to allow the grazing of livestock in asset-protection zones to reduce grassy fuels where biodiversity conservation may be subsumed as an aim below that of fire protection.

Influences on today's ecosystems are either quantitatively or qualitatively different from those in the past. New influences include fertiliser or herbicide drift and the effects of feral animals, domestic stock and exotic plants. Old influences that have been eliminated are those of native predators and other vertebrates and changed fire regimes (see Gill 2003). Many reserves have a problem with too much grazing by exotic, feral animals. Continual research and monitoring is necessary for further understanding of the effects of past and present livestock grazing in various regimes. The effects on plant species in reserves will depend on regimes, which indigenous or exotic plant and animal species are present and their dominance relationships. Effects of grazing or browsing regimes can affect many components of the ecosystem, not just the plants. Perhaps some of the complexity that may arise is evident in the case study of Burrows and Friend (1998), in which trees regenerating after fire were browsed by locusts to the point where the trees died (a burning–browsing effect) – 'permanently changing this site from a rock she-oak thicket capable of supporting [native] red-tailed Phascogales to a grassland affording little habitat for the species'.

Sophisticated computer simulations that allow the manager to explore the many possibilities of fuel management, at least, are being developed. Such simulations can be seen as decision-support tools that may allow the best choices to be based on available information; assessment of such choices through monitoring would remain important. Effects of fuel-management practices on biodiversity of any local area are much more difficult to model, and empirical methods are to be recommended.

Finally, livestock grazing is not just herbivory. For management it involves infrastructure, such as fencing, watering points, tracks and even yards, which all have their effects. Grazing, in itself, involves seed carriage, camping and nutrient concentration, baring of soil in heavily used areas, defecation, urination, breakage of woody plants and trampling, amongst other things. There is a plethora of possible grazing regimes, even if herbivory is considered as the only influence, but the effects multiply when these other aspects of grazing are taken into account.



Chapter 5
Between-fire intervals, times since fire and variations in both

Chapter 5 Between-fire intervals, times since fire and variations in both

Introduction

Fire regimes are invisible (Bradstock *et al*. 2005) but may be discovered in part by analysing fire maps, fire scars on trees and charcoal deposits in swamps. Maps may provide a record of only a short and recent period, swamp deposits one that is imprecise, while many tree-ring records are incomplete as not all fires cause a scar. Thus the determination of historic fire occurrences and intervals between fires is often difficult.

Determining past fire regimes and implementing what is likely to be a new fire regime through prescribed burning is often an aim of management. Fire interval can be seen as a primary consideration in this, not only because it is important for biodiversity conservation, but also because it is the first variable chosen in fire-regime planning. Once the year of burning has been decided, thereby setting the interval, the season of burning within the burning year can be selected, and the fire intensity variation within the chosen season considered. In this chapter – a technical and mostly theoretical one by necessity – the relationships between variations in fire interval and time since fire, and between fixed intervals and random intervals, are considered. The chapter provides general formulas. In the next chapter, the ways in which a manager might choose fire intervals are considered.

Both regular and random patterns of fires may be useful to the manager. In nature, between-fire intervals do not occur with metronome-like precision. They occur irregularly, apparently randomly, with a spread of intervals having an average (mean), thereby implying that a constant interval, to take the extreme case, is 'un-natural'. However, regularly spaced fires may be suited to maintaining minimum litter-fuel loads in asset-protection zones (Chapter 1) set aside for maximum protection of economic assets within or bordering reserves.

Intervals may be seen as being for an area or for a point. Both have some value. An example of the former is the map of Cheney (1979), which shows the frequency of occurrences of large fires likely to cause social disasters somewhere in the area. Areas can be important in animal studies, particularly (e.g. see Gill 1998), and in hydrology (Gill and Allan in press), but point-based methods are significant to plants. Areas (blocks) burnt by prescription may be mapped as having been burnt but usually give little or no indication of the proportion of the block actually burned or where that was located. Essentially, then, the block becomes a point – an irregular pixel.

Variation around a mean interval can be important to biodiversity, the simplest case being an interval that causes a species to become locally extinct. Other examples of the importance of variation about the mean interval follow (see also Gill and McCarthy 1998):

- Jackson (1968), in Tasmania, was perhaps the first to realise the importance of variation in mean interval and variation around the mean for the determination of vegetation type and therefore biodiversity conservation. He hypothesised that differences between vegetation types were due to differences in the overlapping probabilities of certain fire intervals.
- The variation of fire interval about a mean (see Gill and McCarthy 1998) can be significant for the survival of the habitat of some animals, such as Leadbeater's Possum (*Gymnobelideus leadbeateri*) of the Mountain Ash (*E. regnans*, Plate 5.1) forests of Central Victoria (Mackey *et al*. 2002).
- In USA, Clark (1996) modelled interval distributions based on fine-scale sediment-charcoal data, along with fire scars and tree rings. He found that the spread of intervals, even when the means were similar, was important in explaining the persistence of different tree species over the several centuries of the pollen record used as a test. Knowing the mean interval was not sufficient.
- Cary and Morrison (1995) and Morrison *et al*. (1995) in New South Wales found that the diversity of plant species is a function of the variation of between-fire interval in Sydney sandstone communities (Plate 5.2).

Plate 5.1 *Eucalyptus regnans*, a dominant species in this tall, open or wet sclerophyll forest, Wallaby Creek, Victoria. Individuals of this species are killed by fire that completely scorches the crown, a 'seeder' (see Chapter 6). Notice the size of the vehicle as a scale (Gill 1962).

Observation of the variation in interval, or the variation in other fire regime components, has led to a number of guidelines for fire management aimed at the conservation of biodiversity. Thus:

- Gill and Nicholls (1989) considered the circumstance of too-frequent fires and the ways in which this might be avoided for plant species; Noble and Slatyer's (1981) work was not set in a management context but their principles of species characteristics and the timing of critical events, such as first seeding, apply.
- Bradstock *et al*. (1995) referred to intervals likely to be damaging to plant-conservation value within a fire regime while emphasising that variation between limits is appropriate.
- Tolhurst *et al*. (1992*b*) suggested that 'Management of most ecosystems in Victoria may require diverse fire regimes rather than regular cyclic burning'.
- Keith (1996) maintained that 'a site must experience fires of varying intensity, season, size and at varying intervals if its full complement of species is to be maintained'.

Note that if species are adapted to a particular fire regime (Gill 1975), and therefore not adapted to the complement of fire regimes possible, then simple headlines such as 'pyrodiversity promotes biodiversity' (Martin and Sapsis 1991) – implying that any diversity in fire regimes is appropriate – need to be closely examined for context.

Plate 5.2 Effects of too-frequent fire. On the right-hand side, *Banksia ericifolia* cones are visible in a shrub skeleton in this recently burnt area in the Sydney region; on the left-hand side of the track, all the seeder Banksias have been eliminated by too frequent fire (Cary 1992).

What should the nature of variation of fire interval about a mean interval be? Will it always be the same? Constancy is unlikely based on the variation in nature (e.g. McCarthy *et al.* 2001*a* for Australia) but the question of variation cannot be fully answered yet in a quantitative way. The simplest model of random variation for between-fire interval and the mathematically related time since fire is when there is an even chance of a fire event in any year after the previous fire. More complex models can be derived on the basis of the relative rates of fire spread and area covered as a function of the time since fire (see McCarthy *et al*. 2001*a*). In the next section, the relationship between regular and random fire occurrence is explored.

Regular or random: a simple model

In this section, the effect of *variation* about the mean interval for time since fire and for between-fire interval is illustrated by modelling the burning of fixed patches or blocks in an imaginary reserve. In these cases, the patch can be considered to be the same as a point because the patches are uniform in size, shape and fuel. Relatively simple illustrations are used to assist in concept development here; their limitations are also instructive.

Imagine an area in which 50% can be burnt every year, a mean somewhat similar to that burnt in the dry season in the lowlands of Kakadu National Park (Gill *et al*. 2000), but used here as a fixed amount for convenience of illustration and not for its possible application to any actual locality. This is the simplest of examples other than 100% burned per year. Currently, proportions burnt in southern Australia on a landscape scale are much less than this.

A series of simple diagrams illustrate the possibilities for patterns of burning and the outcomes that emerge for both fire interval and its variation and for time since fire.

1. *Consider a fixed proportion (half) of the study area is burned each year and the same half is burned every time*
   In Figure 5.1, the same half of the area is burnt each year with the result that the burnt half has a between-fire interval of one year and the most recent time since fire is also one year. The interval and time since fire on the unburnt half is indeterminate (or can be set at infinity for this purpose). In this example, the number of blocks is minimal and there is no randomness between blocks or any variation as a function of time. It is a highly constrained system – deterministic and perfectly regular. Note that for the study area as a whole, the mean interval between fires is two years (the reciprocal of the proportion, or fraction, burned per year, *viz*. 1/0.5). In practice, this is obviously an undesirable way to determine mean interval; however, there is often an element of this effect in real-life statistics.

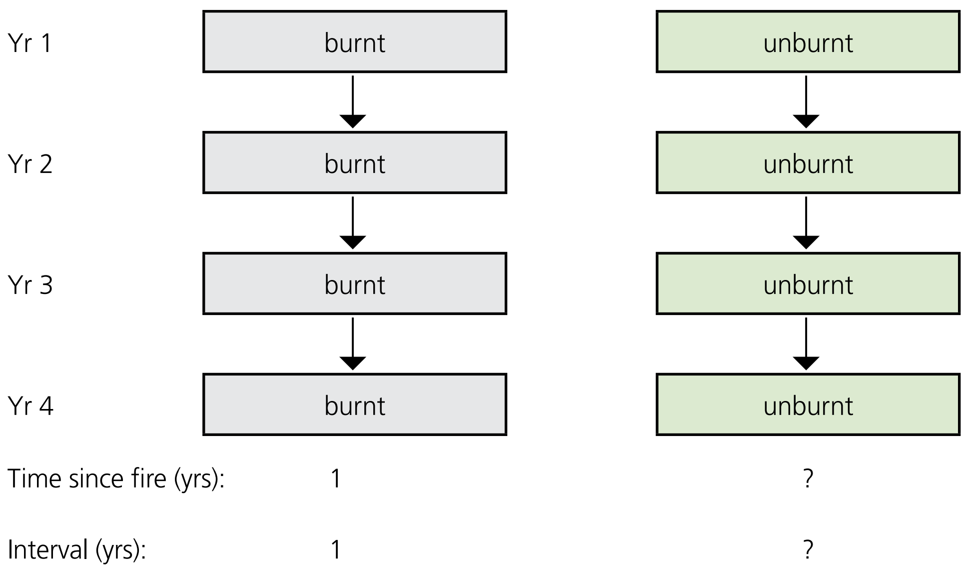


Figure 5.1 (a) The study area is represented by two columns, each one being half of the study area. 50% is burned each year and it is always the same half. The burnt half is burnt each year for four years. Grey blocks are burned while green blocks are not burned. On the burnt half, the time since fire and the interval between fires is always the same – one year – from the viewpoint of year five. There is a minimum one-year fire-free period – the minimum time step being considered.

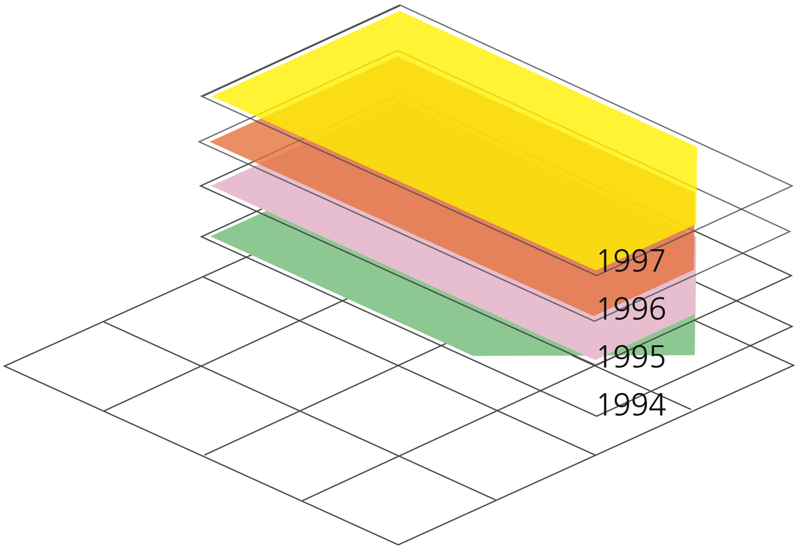


Figure 5.1 (b) A schematic of (a) with the study area a lattice of white squares and each annual fire represented by a coloured strip (Gill 1999*b*).

2 *Consider alternate halves being burned each year*
In this example, the same rules apply as in Figure 5.1 except that the blocks being burned are alternated from year to year. The average amount burnt per year remains at 50%. The interval between fires is always two years. The time since the latest fire is one year on one half of the area and two years on the other. The system is deterministic (regular), as in Figure 5.1, but is quite different in the way it is achieved. Note that the interval between fires is always two years, as above, while the time since fire can be either one or two years.

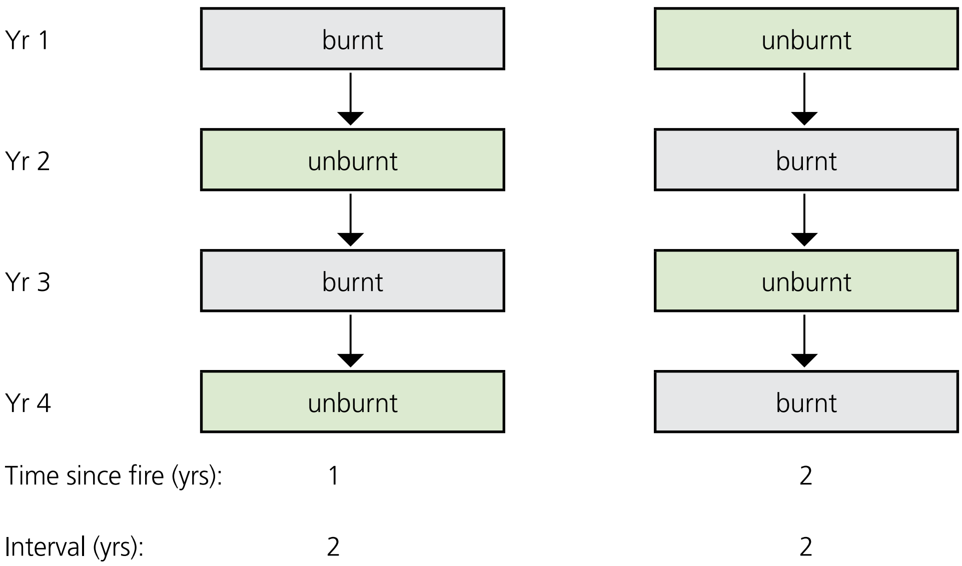
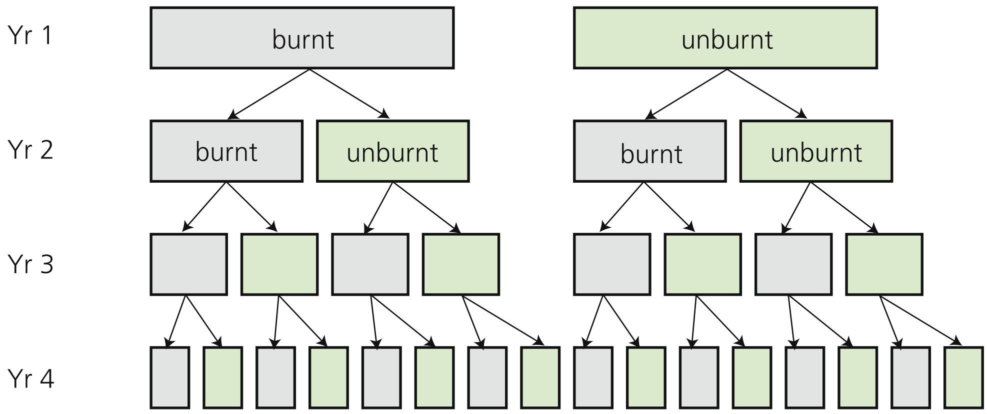


Figure 5.2 Half the area is burned each year, as in the previous figure, but blocks are alternated each year. Grey blocks are burnt while green blocks are not. Note that the interval between fires is always two years and that the time since fire on a block alternates between one and two with the passage of time.

3. *Consider half of each block, whether burnt or not the previous year, is burned*
   In Figure 5.3, half the area is burnt each year, as before. The burned patches become smaller with each successive treatment. Of course, this process has limitations but illustrates how greater variation can be introduced to time since fire (*tsf*) and interval (*int*) than above by increasing the number of blocks.



tsf (yrs)	1;	2;	1;	3;	1;	2;	1;	4;	1;	2;	1;	3;	1;	2;	1;	?
int (yrs)	1;	1;	2;	1;	1;	2;	3;	?;	1;	1;	2;	?;	1;	?;	?;	?

Figure 5.3 The study area remains the same but subdivision of blocks is allowed. Half of the study area shown at each row is burnt every year for four years. Abbreviation and colours are the same as for other figures.

The numbers of patches, represented by the Year 4 areas, with particular times since fire, show an approximate but incomplete negative exponential frequency distribution (i.e. ½ blocks are one year, ¼ are two years, 1/8 are three years, 1/16 is one year and there is one block undetermined – see further, below). Similarly, intervals between fires also suggest a negative exponential frequency distribution – modified from a strict sequence as the number of determinations is small. See Figure 5.6 for a full expression of a negative exponential distribution using the same mean interval.

Note that:

i. *tsf* and *int* values are not coincident but form the same statistical distribution
ii. One-year intervals between fires can be for the period between Years 4 and 3, 2 and 3, or 2 and 1; intervals are embedded at various time depths in the fire history of the area
iii. Annually splitting blocks into two is unwieldy and grossly limited from a practical point of view
iv. The mean interval of two years is set and kept static by declaring a fixed proportion of the area to be burned each year.

4. *Consider an area of 16 blocks in which exactly half are burned at random each year*
   The system depicted in Figure 5.3 has 16 blocks at the outset (Year 1) with a cluster of 8 blocks (subdivisions between blocks not shown) being burned in the first year, two clusters of four contiguous blocks burned in the second, eight clusters of two blocks in the third and finally 16 single blocks burned. Or a square study area of 16 blocks could be defined and fires of area 8 blocks forming a rectangle overlaid on it for the four years in a way that causes each area to be burnt in a contrived but still random manner (see Gill 1999*b*, Figure 5.4*a*).

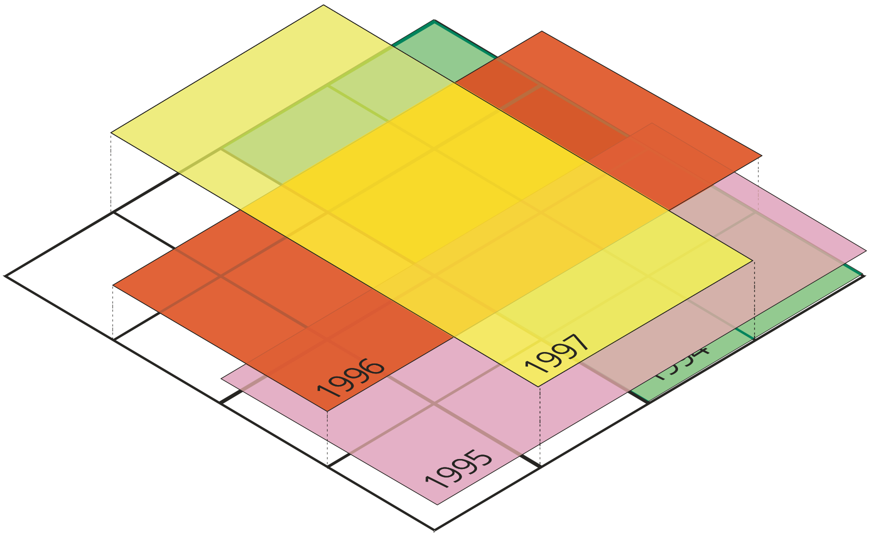
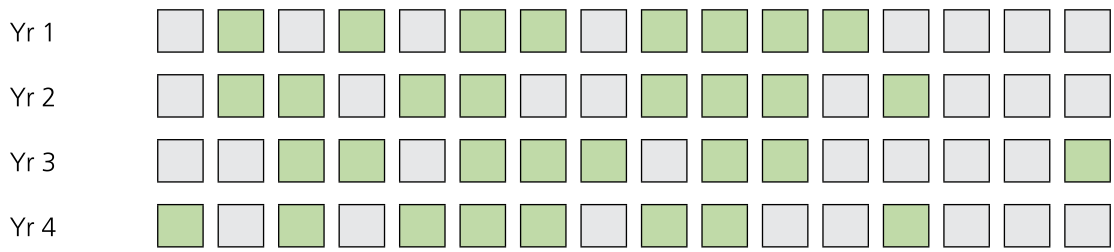


Figure 5.4 Random patterns

(a) A schematic of a study area – a lattice of white squares – being burned by annual fires for a period of four years, each fire being represented by a coloured strip and being applied with contrived randomness.



tsf (yrs):	2	1	4	1	2	?	3	1	2	?	1	1	2	1	1	1
int (yrs):	1	1	?	2	2	?	?	2	?	?	?	1	2	1	1	2

(b) The study area – a linear feature of separate blocks – has 16 blocks, half of which are burned each year at random. Grey blocks are burned while green blocks are not burned.

In Figure 5.4*b*, 16 blocks are depicted. Eight have been randomly chosen for burning each year. In this system, the number of blocks with particular time since fire has only an approximate negative exponential distribution (numbers of three year *tsf* are lower than expected, while indeterminate *tsf* are more numerous). Similarly, note that there are fewer blocks at one- and three-year intervals than the statistical ideal (infinite number of blocks with random treatment at the same mean). On the other hand, there are more blocks than 'ideal' with two-year and indeterminate (i.e. ?) intervals; however, such observations are to be expected in such a random system.

5. *Consider that half the 16-block area is burned per year but only on average*
So far, the number of blocks burned each year has been kept at exactly eight from a total of 16. In a real system, there is usually some constraint that prevents burning every year, from whatever cause, natural or artificial. The last example in this section keeps the same average amount burnt per year (over a large number of years) and shows an example for a run of four years, as above. In this case, the number of indeterminate intervals becomes quite high but one item of interest is that the number of unburnt blocks for each year is 9, 9, 8, 8 respectively. This change from the uniform 8, 8, 8, 8 pattern is small but significant. As the numbers of runs of four years are expanded, more variation is to be expected. In just one more year of data – not shown – chance dictated 13 of the 16 blocks were unburned for Year 5.

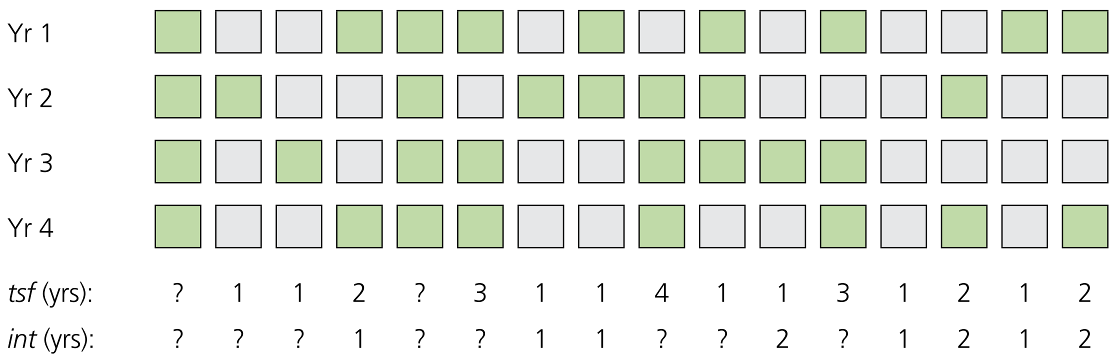


Figure 5.5 An average of eight blocks per year are burned (50%), but the number burned per year is not constrained to exactly eight. Symbols and colours are the same as in previous figures.

If the number of years of data or number of blocks in the model increased, then the average *tsf* and *int* would be expected to be close to two, and the frequency of occurrence of blocks in each age category fit the negative exponential relationship better and approach that in Figure 5.6. As the number of blocks decreases the variation in area burned per year increases, the extremes being 0% and 100% burned.

6. *A general model for random burning of a large number of blocks, one half per year on average*
   With the randomness illustrated above, a negative exponential curve for the frequency of blocks in each year's category arises according to the average proportion burned each year, *P*, in order of years beginning with Year 1:

   P; $P(1\text{-}P)$; $P(1\text{-}P)^2$; $P(1\text{-}P)^3$; *etc.* Formula 5.1

   P is also the probability of a block (point) being burned, while (1-P) is the chance of a block not being burnt (see McCarthy *et al*. 2001*a*). The fire cycle or average interval between fires is $1/P$ years. A graph of this relationship is shown in Figure 5.6, with a mean interval of two years. The effect of transforming the probabilities to their natural logarithms is shown in Figure 5.7. The R^2 values represent 'goodness of fit' and are equal to one, because the graphs are from an equation, not from real data.

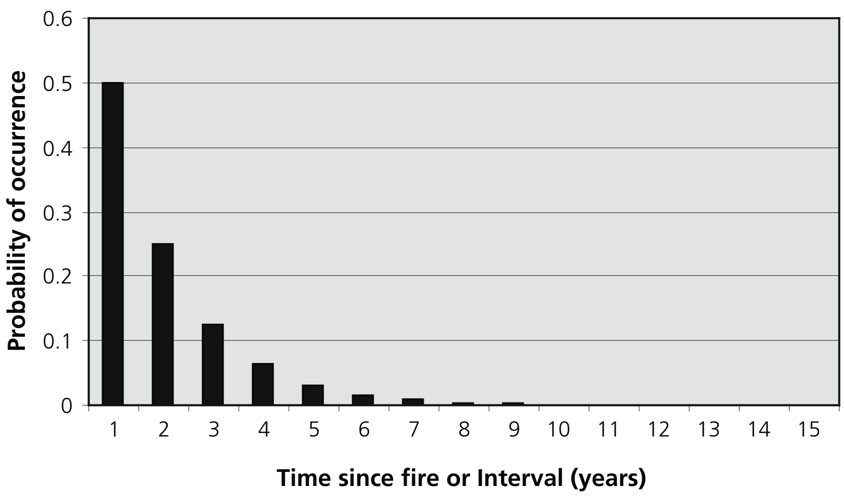


Figure 5.6 Frequency distribution of the intervals between fires, or times since fire, expected when the mean interval is two years. The chance of encountering an interval of 10 years is approximately 0.0009, or about 1 in 1000. The data are an illustration of theory only, so the data fit a negative exponential curve perfectly ($R^2 = 1$).

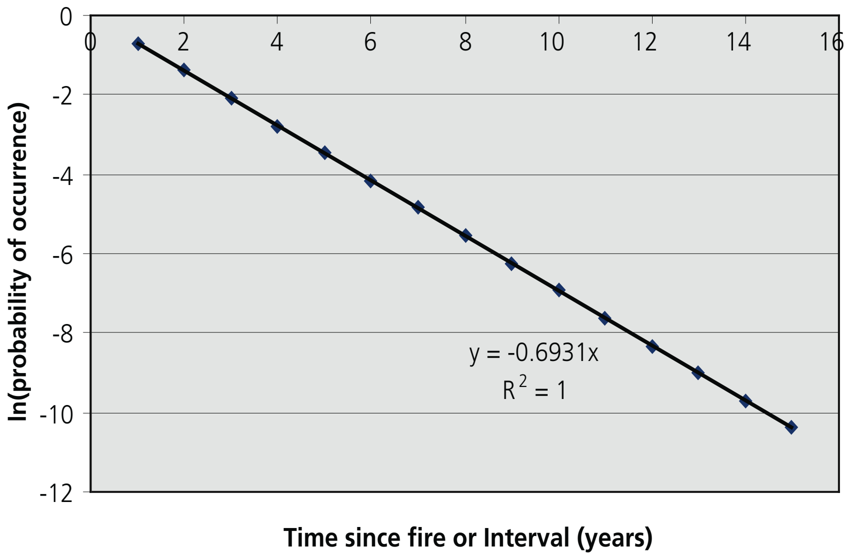


Figure 5.7 As for Figure 5.6 except the Y-axis data have been transformed into logarithms with base e. The data are an illustration of theory only, so the fit is perfect ($R^2 = 1$).

Section conclusions

The models presented are simple but have many lessons for the fire manager. They illustrate how:

- The same extent of burning each year can have quite different outcomes for time since fire and the last interval between fires
- The pattern of burning can be random or deterministic, the latter giving a regular, constant, interval
- Randomness increases the variation in interval and time since fire
- The latest interval in the record can occur at different times since the last fire
- Randomness can be between blocks within a year or for one block across years
- A negative exponential distribution of times since fire and of intervals between fires is expected from the one random model shown – sample size (number of patches or burning blocks) and number of years of maps affects the approximation to theory
- Unknowns (marked with a ? in the figures, above) occur even in simple models, but especially in reality where records are often relatively short in relation to the expected spread of intervals. When unknown intervals occur, the data is said to be censored and sophisticated mathematical methods have been proposed to deal with them (Polakow and Dunne 1999).

A feature of our examples is that they start with no prior information – no history – and this is often the case in practice.

Notice that in these examples only two fire cycles of average two-years' duration have been traversed. If we think of this in relation to the availability of data for cycles that average 30 years (estimated by Wouters *et al*. 2002 to apply in many Victorian communities), 60 years of data would be needed just to reach the stage of our examples that show so many unknown intervals and times since fire. Data custodians in most Australian jurisdictions, with the likely exception of south-western Australia, would be unable to claim 60 years of accurate maps of fires. Many would not claim a quarter of that. In 27 years of unplanned-fire maps in the Central Directorate of New South Wales National Parks and Wildlife Service, 29% of the area had no fire recorded, 28% had one and the remainder two or more. Thus intervals could only be determined for 43% of the area (de Ligt 2005). Considerable foresight and determination is needed if we are to ever have a satisfactory data set on which to estimate fire intervals.

In all the examples depicted previously, an average interval of two years between fires (the reciprocal of 0.5, the average proportion burned per year) was present in each case; as the mean interval increases, the range of values for *tsf* and *int* obtained in a limited data set will also increase. Note that the negative exponential models have an upper reach of infinity, while reality dictates that the graphs are truncated, even if the truncation point is unknown (see Finney 1995).

Random models show that there is much more area with short fuel-age classes than long in a time-since-fire distribution. This applies to the age of the surface fuels, not necessarily the age of forest trees. Old-growth forests may be classed as old either because they originated from the last fire that occurred a long time ago, or they have survived repeated fires and the trees are old but the surface fuels are not. Old-growth forests in a random model of tree ages across a landscape – as opposed to fuel ages – will have larger areas in the younger age classes than older.

Effects of a fixed-minimum interval

In this section, the effects of a period after fire in which no fire is possible, or in which fires are prevented for one reason or another, are examined. Having a fixed interval between fires, as discussed in the last section, defines a fire-free period; however, so far, mixtures of fixed fire-free periods and randomness have not been addressed. Mixtures of regular and random intervals may be expected in managed systems and some natural ones also. With a time step of one year, a fire-free period of one year is usual (as assumed in the previous models).

Minimum fire-free period

For various reasons it may not be possible for a fire to occur the year after the previous one, or even for many years (Table 5.1). Reasons for a minimum fire-free period greater than one year may be due to:

- Lack of fuel continuity – an obvious reason why a fire will not spread one or more years after another fire. This may be reflected in low fuel levels overall, or in a fuel having marked patchiness on a plant–plant basis. For example, while grassland fuels in the wetter areas of Australia are generally able to burn one year after the previous fire, those in the arid zone (Table 5.1) may require a very wet year or two to grow the short-lived grasses that provide the continuity of fuel between the distinctive hummocks of the perennial grass species (spinifex) when they cure (dry out) (Griffin *et al.* 1983).
- Management considerations, such as retention of biodiversity, protection of a young forest crop or inability to burn under mild weather conditions. This may lead to a fire-free interval greater than one year being imposed or sought.

While weather suitable for burning can be a factor affecting how soon an area can be prescribed burnt after a fire, the minimum periods given in Table 5.1 are considered to be due to factors other than weather.

Table 5.1 Length of fire-free period due to inability of fires to spread in the first years after a previous fire. Note the variation in this period from place to place, even for the same fuel type. Extreme fire weather may shorten the minimum period, as will the growth of interstitial fuels in hummock grasslands, due to a year or more of heavy rainfall.

Vegetation	Period (years)	Reference
Buttongrass Moorland (Tasmania, cool temperate climate)	2	Marsden-Smedley and Catchpole (1995)
Heath (Sydney region, NSW, temperate humid climate)	<2 (when exposed to headfire during extreme weather)	R. A. Bradstock (pers. comm., 2006)
Heathy woodland (Victoria)	2	Grant and Wouters (1993)
Hummock grassland (arid zone)	13	Craig (1999)
Hummock grassland (arid zone)	5 in Tanami region, 10 in Uluru region, and possibly up to 20 in the Great Victoria Desert	Allan and Southgate (2002)
Mallee heath (WA, Mediterranean-type climate)	<5 under extreme conditions, otherwise 8	McCaw *et al*. (1992); McCaw (1997)
Mallee (Victoria, Mediterranean-type climate)	10–15	Cheal *et al*. (1979), p. 44
Open Forest (WA, Mediterranean-type climate)	<2	after Underwood *et al*. (1985)
'Porcupine mallee' (south-eastern Australia)	15–20 on dunes, less than 3 in swales	Noble (1997), pp. 50–51
Shrublands, northern Sandplain, (s.w. WA)	3	See Abbot (2003)
Spinifex in the arid zone	3–10 years	Kimber (1983)

Table 5.1 shows a variety of minimum periods, even for the same vegetation type. The minimum period is unlikely to be fixed as fuel accumulation will vary from time to time and may be affected by circumstances and other factors, such as extreme fire weather and fuel-generating events. Not enough is known to show the expected variation over a long period of time.

Effect of a minimum interval between fires on the frequency distribution of intervals

To see how the fire-free period affects the dynamic of the system, another simple model (compared with those illustrated above) can be used, this time with a mean four-year between-fire interval. With a one-year fire-free period as minimum, the benchmark condition here, a negative exponential trend for proportion of landscape with various times since fire or between-fire intervals occurs (as before). A one-year interval or one-year time since fire has a probability of occurrence of 0.25 (i.e. ¼), then successive years have probabilities, P_{ti}, of 0.19 [i.e. $P(1-P)$], 0.14, 0.11, 0.08, 0.06, 0.04 and so on (see Formula 5.1). This sequence of numbers trends to zero and their sum trends toward one as interval increases. The sum of probabilities must eventually equal one as it is a probability function.

Now, let's consider that four blocks comprise our study area. On average, one block burns per year (*P* = 0.25). There are a number of alternative, random possibilities for burning an average quarter of the four-block study area each year (*P* = 0.25), including:

- One block, determined by chance, burns per year (i.e. the area burned per year is held constant)
- The number of blocks burned per year varies from zero to four, so the amount burned per year varies widely – the average area burned per year remains the same as above (i.e. 0.25)
- Another fire will not occur the year after a fire, or for a period of some years (a minimum interval, *M* years), perhaps because there is too little fuel to burn or it is considered undesirable for conservation reasons.

The consequences of the last of these possibilities is explored by varying the minimum period from one to four years using our four-block area, and burning only one block per year with *P* = 0.25.

In Table 5.2, the general method of the simple simulation is illustrated. In this example, the minimum interval is initially one year (the benchmark condition and the time step); eight years of data are shown. Thus it is possible to have fires in the same block in successive years. The times since fire and between-fire intervals noted in Table 5.2 are for the last interval and time since fire.

Table 5.2 Incidence of blocks being randomly burned each year when the average interval between fires is four years. One block is burned per year. 1 represents a fire and 0 the absence of fire. The minimum interval between fires is one year, so fires are allowed in the same burning block every year. *tsf* and *int* represent times since fire (years) and between-fire interval (years), respectively.

Year	Block 1	Block 2	Block 3	Block 4	Area burned per year (number of blocks)
1	0	0	0	1	1
2	1	0	0	0	1
3	0	1	0	0	1
4	1	0	0	0	1
5	1	0	0	0	1
6	0	1	0	0	1
7	0	0	1	0	1
8	0	0	1	0	1
Last *tsf*	4	3	1	8	
Last *int*	1	3	1	?	

Note that one interval cannot be determined (marked as ?) as the record is not long enough. The last interval of Table 5.2 varies from one to three years only, but from Formula 5.1 we know that the range can be extensive – the probability of occurrence of long intervals declines as the interval increases (as in a negative exponential distribution).

When the minimum interval is set at two years, the range of intervals is constrained as an interval of one year is not possible by definition. However, the time since the last fire may still vary from one year upwards (Table 5.3). Formula 5.1 does not apply in this case because of the change in the rules.

Table 5.3 As for Table 5.2 except the minimum interval between fires is two years. Years in which burning is not allowed, by definition, are shown in grey.

Year	Block 1	Block 2	Block 3	Block 4	Area burned per year (number of blocks)
1	0	0	0	1	1
2	1	0	0	0	1
3	0	1	0	0	1
4	1	0	0	0	1
5	0	0	0	1	1
6	0	1	0	0	1
7	0	0	1	0	1
8	0	1	0	0	1
Last *tsf*	5	1	2	4	
Last *int*	2	2	?	4	

When we move to a minimum interval of three years (Table 5.4), between-fire interval is necessarily restricted to three years or more. When the minimum interval is four years, no variation in interval is possible (Table 5.5). Note that times since fire continue to vary. Interval and times since fire behave differently, compared with Figure 5.6.

Table 5.4 As for Table 5.2 except with a minimum three-year interval. One interval cannot be determined as the record is too short. The years in which burning is not allowed, by definition, are shown in grey.

Year	Block 1	Block 2	Block 3	Block 4	Area burned per year (number of blocks)
1	0	0	0	1	1
2	1	0	0	0	1
3	0	1	0	0	1
4	0	0	0	1	1
5	1	0	0	0	1
6	0	1	0	0	1
7	0	0	1	0	1
8	1	0	0	0	1
tsf	1	3	2	5	
int	3	3	?	3	

Table 5.5 As for Table 5.2 except the minimum interval is four years – the maximum for a fire cycle of four years. The period in which no burning is permitted is shown with grey shading.

Year	Block 1	Block 2	Block 3	Block 4	Area burned per year (number of blocks)
1	0	0	0	1	1
2	1	0	0	0	1
3	0	1	0	0	1
4	0	0	1	0	1
5	0	0	0	1	1
6	1	0	0	0	1
7	0	1	0	0	1
8	0	0	1	0	1
tsf	3	2	1	4	
int	4	4	4	4	

With this simple set of examples it is apparent that the extent of variation in interval is going to decline to zero as the minimum interval, *M* years, reaches *C* years (where *C* is the length of the fire cycle in years and $C = 1/P$; see also Gill 2000). This sequence (Table 5.2 to Table 5.5) illustrates the spectrum from randomness to regularity. Randomness was between blocks within a year as randomness between years was disallowed.

When the minimum interval between fires is the same as the time step, *viz*. one year, the distribution of intervals looks exactly the same as that for times since fire when fires are at random (Formula 5.1). That is, the frequency of blocks burned in each category – equivalent to the probability of occurrence – will be in the form of a negative exponential graph (Figure 5.8 is shown in semi-logarithmic form), with successive yearly values for the probability P_{ti}, equal to 0.25, 0.1875, 0.141, 0.105 and so on [i.e. P, $P(1-P)$, $P(1-P)^2$ etc] as noted above. How will distributions change as a minimum interval between fires is introduced? A general theory is given below, and the results compared with the same conditions as those applied in the above tables.

Figure 5.8 Negative exponential distribution of times since fire or between-fire intervals when P=0.25. Compare with Figure 5.6 that has the same form of relationship but a value of P=0.5. The equation has a perfect fit ($R^2 = 1$) because it is theoretical.

General theory for between-fire interval and time-since-fire distribution with a given fire-free interval

The example chosen was for an area of four burning blocks. Fire was allocated to a block on the basis of random choice. Restrictions occurred according to the number of blocks in the target area (4), as well as restrictions due to declared fire-free status for various periods after the last fire. The time step was one year and one block (1/4) was burned each year, thus P=0.25.

For an area consisting of four burning blocks and no fire-free period other than the minimum time step of one year, the choice of burning blocks at any one time step is one out of four (1/4). When the minimum interval between fires is two years, the choice of burning blocks at any one time step is restricted to one out of three blocks (1/3). When the minimum interval is three years, the choice is one out of two blocks (1/2). Finally, when the minimum interval is four years, the choice – or lack of it – is of one block only (1/1). In this sequence there is a shift from random to fixed-interval burning.

A fire-free period declared following a previous fire determines the number of occurrences of a zero probability at the start of the frequency distribution of fire intervals: this is given by (M-1) years, where M is the fire-free interval. Thus when the minimum fire-free interval, the time step, is one year, there is no zero value for the occurrence of an interval of one year and the random probability distribution used above applies without restriction. However, when M is four years, there are three consecutive years (i.e. M-1 = 3) with a zero probability (for fire intervals of one, two and three years) followed by a probability of one at four years.

When the probability of occurrence is zero for one or a number of years, the probabilities for the remainder of the intervals in the distribution have to rise until, when M reaches four, the probability of occurrence of an interval of four years is one as in the last paragraph. They must do so as the sum of probabilities must equal a value of one. When there is a zero probability at an interval of one year (i.e. M = 2), the probabilities for the remaining intervals are as if the mean had shifted from four years to three i.e. (C-M+1) years, where C is the fire cycle in years (C = 1/P). When the minimum interval, M, is four years, this general formulation points to a probability of one, as is necessary. These probabilities are assigned after the zero probabilities have been assigned.

Consequently, the general formula for between-fire interval is:

i. A period of (M-1) years with P_{ti} = 0; where P_{ti} is the probability of an occurrence of a particular interval

following which

ii. There is a probability P_{ti} for successive years i.e. year M and onwards given by probabilities determined using a mean of (C-M+1) years.

In our example, there is a fire in the target area every year. If the minimum interval is one year (i.e. M = 1), then the probability of occurrence of a particular time since fire, P_{tt}, is given by the same negative exponential distribution as for intervals. If there is a two-year minimum interval between fires (i.e. M = 2), the times-since-fire distribution will start with P_{tt} = 0.25 for the first year (i.e. for M-1 years). With some of the probability distribution assigned – 0.25 in this case – the sum of the remaining probabilities is now [1 – (M-1)P] or 0.75. Thus the remaining probabilities can be assigned according to 0.75 times the probability sequence determined for a negative exponential distribution with a mean of three years, that is (C-M+1) years.

To take another example, when there is a three-year minimum interval between fires (i.e. M = 3), P_{tt} = 0.25 for the first two years (i.e. M-1 years), then for successive years is given by 2/4, that is [1 – (M-1) P] times the probabilities for a negative exponential distribution with a mean of 2 [i.e. (C-M+1)] years.

When there is a minimum interval of four years, P_{tt} is 0.25 every year for the first three years. In the fourth year, P_{tt} is given by [1 – (M-1)P], i.e. 0.25, times the probability sequence for a mean of (C-M+1) years, i.e. one year, a value of 1. This leads to P_{tt} = 0.25 in the fourth year, thereby giving four equal values of 0.25, totalling one.

To generalise for the probability of occurrence of times since fire:

1. $P_{tt} = P$ for the first (M-1) years
2. In successive years, P_{tt} is equal to [1-(M-1)P] times that for year one onwards when the mean is (C-M+1) years.

Simulation of interval distribution when there is a minimum fire-free period

In the previous simple situation, four burning blocks comprised the study area. One fire per year was randomly allocated between blocks but within the strictures of the minimum interval allowed. Now, a simulation is run for 40 years and every between-fire interval used for estimating probabilities of occurrence. Times since fire were noted for each block every five years. When the minimum interval is one year, the negative exponential graph is expected (Figure 5.9). As the minimum interval increases, the graphs change according to theory (Figures 5.10 to 5.12).

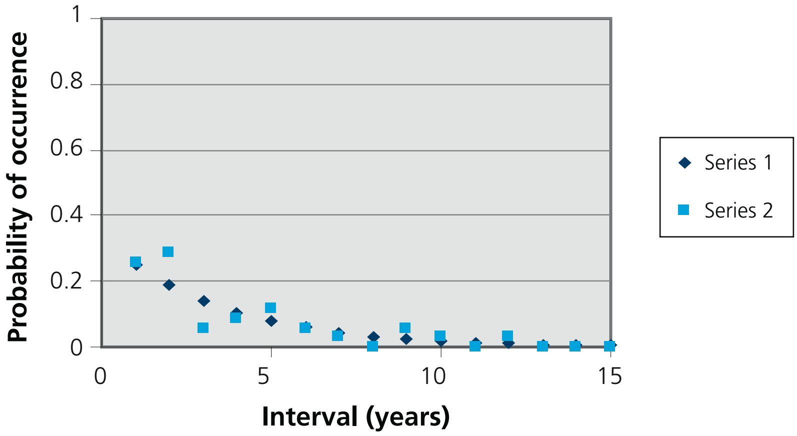


Figure 5.9 Interval distribution when there is a minimum interval of one year and one fire is randomly allocated to one of four burning blocks each year (P=0.25) for 40 years. Series 1 shows the theoretical result and Series 2 the results observed (n=35) in the simple simulation.

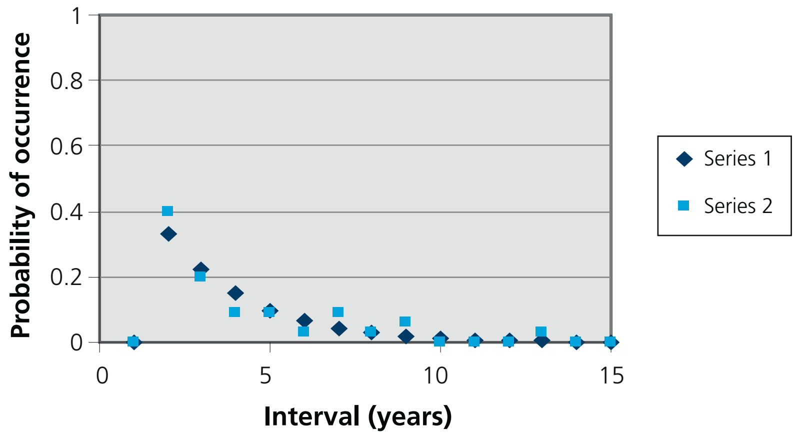


Figure 5.10 As for Figure 5.9 except that the minimum interval between fires is now set at two years. Note the zero value for an interval of one year and the change in maximum probability of occurrence. Series 1 shows the theoretical values and Series 2 shows the simulated values.

Fire and adaptive management

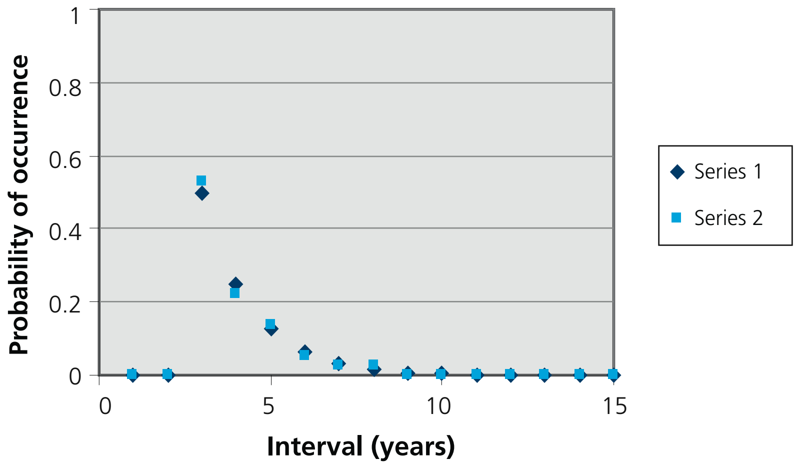


Figure 5.11 As for Figure 5.9 except with a minimum three-year interval between fires. Note the zero values for the first two intervals. Interval distribution according to theory (Series 1) and a simulation of 40 years for a four-block study area, a minimum of three years between fires, and one fire (1/4 blocks) burned per year (Series 2).

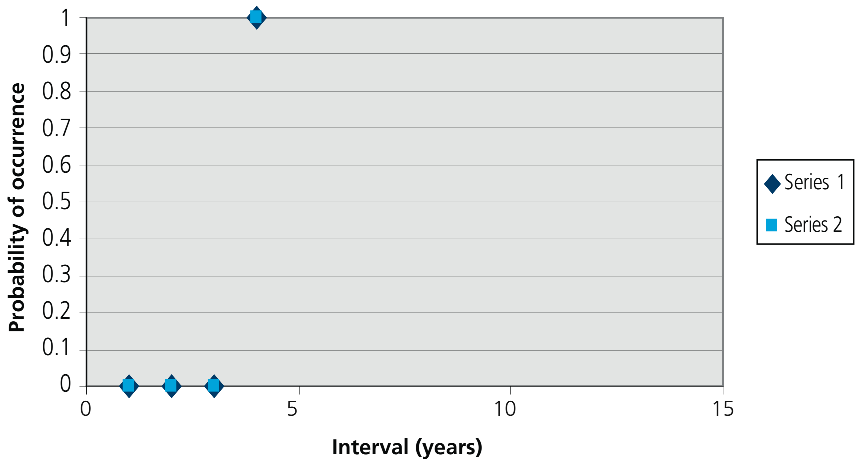


Figure 5.12 As for Figure 5.9 except minimum interval is four years. Note the zero values for the first three intervals. The maximum probability (and only option) is at four years. The scale has been left as above to illustrate the change. Observed and expected results are identical.

The effect of a minimum between-fire interval on the time-since-fire distribution

In the following series of graphs, a simulation with the same criteria as that for intervals in the above section was carried out, but for times since fire. As a fire-free period greater in length than the time step of one year (Figure 5.13) is introduced, the departure of these graphs from those above will be apparent (Figures 5.14 to 5.15). With a four-year minimum, all times since fire will have an equal probability of occurrence of 0.25; this is not illustrated.

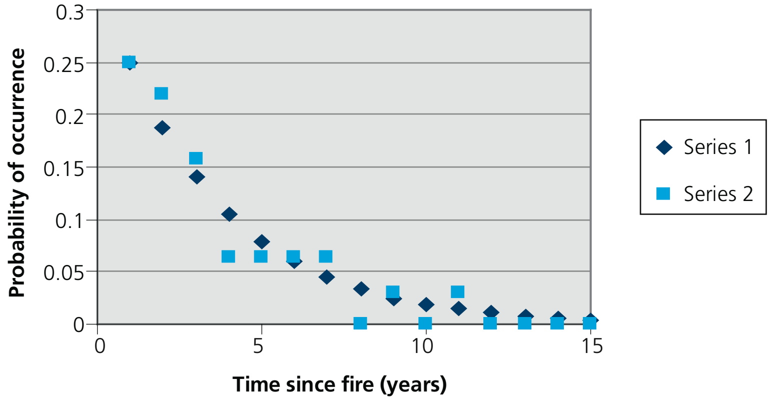


Figure 5.13 Theoretical probabilities of times since fire (Series 1) for four blocks with a minimum interval of one year (the time step). Observed values (Series 2, n=32) are from a simple simulation with one block burned per year randomised across four blocks for a period of 40 years.

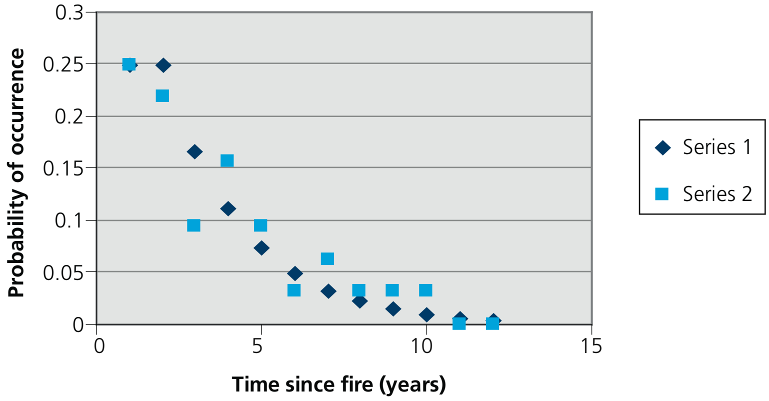


Figure 5.14 Theoretical probabilities of times since fire (Series 1) for four blocks with a minimum interval of two years. Observed values (Series 2, n=32) are from a simulation with one block burned per year randomised across four blocks for a period of 40 years.

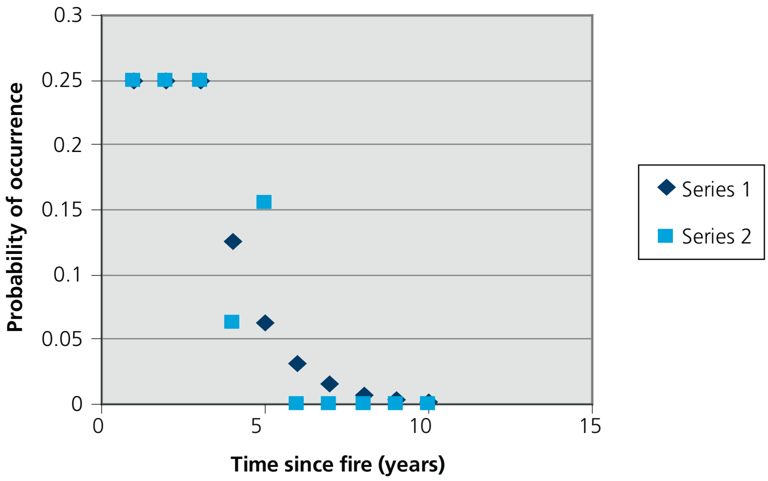


Figure 5.15 Expected probabilities of times since fire for four blocks with a minimum interval of three years (Series 1) and observed values (Series 2, n =32 times since fire) in a simulation with 1/4 blocks burned per year (i.e. one per year) randomised across blocks each year for a period of 40 years. Points for the first three years are coincident at *P*=0.25.

Removal of the constraint of one block only being burnt per year

In the previous simulations, it was declared that one block and no more would be burnt each year. The effect of this can be understood by removing the constraint for the simplest case – minimum fire-free period of one year – the minimum time step. A negative exponential graph would be expected as before (Figure 5.8). The main change is that instead of a steady one block per year (P=0.25) being burned, the number in one trial of 40 years and four blocks varied from zero to three, most of the values being zero (one or two blocks per year). The proportion expected in sequence from no fire in the four blocks to four fires in the four blocks is 0.316, 0.422, 0.211, 0.0469 and 0.0039. These proportions are calculated from the probabilities of fire or not in each block multiplied together, then multiplied again by the number of alternate states of fire/no fire probabilities between the four blocks. Thus the chance of no fire at all in any of the four blocks is given by $1(0.75_{*}0.75_{*}0.75_{*}0.75)$ = 0.316. The chance of no fire in a block is 3/4 or 0.75 and there is only one variant – the probability is the same for each block – so the multiple is one. The chance of one fire among the four is $4(0.25_{*}0.75_{*}0.75_{*}0.75) = 0.422$. The value 0.25 can go in only four possible positions, hence the multiplier is four. The chance of two fires among the four blocks is $6(0.25_{*}0.25_{*}0.75_{*}0.75) = 0.211$. The chance of three fires is $4(0.25_{*}0.25_{*}0.25_{*}0.75) = 0.0469$. The chance of four fires is now obvious at $1(0.25)^4 = 0.0039$. The sum of all these probabilities is 0.9998, which is close to unity – rounding off errors cause the difference.

Say that there were 40 blocks in the study area, instead of four, the proportion burnt per year on average was 0.25 and the minimal fire-free period was one year. As before, we would expect the average proportion of blocks having a one-year interval to be 0.25, but what variation around this would be expected? An indication of this variation can be seen in the graphs above, by the departure of the observed from the expected results (Figure 5.9 Series 2 compared with Series 1), but this is only one example.

A final point here is that the longer the mean interval, the greater the absolute variation to be expected around the mean. If the mean is 50 years, then some intervals may be expected to be more than 250 years on occasion.

Discussion and conclusion

An introduction to the topic of fire intervals and times since fire and their variation has been made in this chapter. It is to be emphasised that there is a great deal more to this topic. For the treatment of censored data and the use of various forms of randomness see Clark (1991), Johnson and Gutsell (1994), Polakow and Dunne (1999), McCarthy *et al.* (2001*a*), Clark *et al*. (2002) and Moritz (2003).

The practical questions about which intervals and what variation is appropriate for management remain. There is evidence of at least some species being affected by variation in interval; however, the insistence on variation for biodiversity conservation seems to be based largely on the observation that variation is to be expected in nature. If the biota evolved partly in the presence of human beings who had an influence on the fire regime, as in Australia, then the assumption is that their actions led to a variation that can be taken to be random, even if ignition was targeted at a particular place at the time of any one fire. Burning a set proportion of blocks each year is probably more practical than varying the proportion of blocks deliberately burned per year in a bid to attain randomness in time as well as space – if there are a sufficient number of blocks.

In real life, the variation in fire intervals is affected by fires spanning many orders of magnitude. The difficulty then, in theory and in practice, becomes the size of the area that is needed as a study area. The answer is that it should be substantially larger than the largest fire (e.g. see Baker 1989). However, if maps are the basis of the study, the larger the area the more heterogeneous it becomes and the greater the likelihood that the mean result is a compromise (see the appendix at the end of this chapter).

The examples chosen in this chapter have all been based on between-fire intervals and the related times since fire. They have not concerned the importance of fire properties, seasons of burning or fire type. This largely reflects the state of knowledge of the system. For the models of possum habitat and fire regimes in the Central Highlands of Victoria, described in Mackey *et al*. (2002), the death or otherwise of the trees was important – an outcome determined by fire intensity, the height of the trees and possibly seasonality of fire occurrence. As knowledge of the effects of fires on the biota grows, more Australian examples of the effects of variation within fire regimes are to be expected.

The curves for times since fire and for between-fire interval are mathematically related to the probability of burning at a point (*PBP*), as a function of time since fire. This is outlined in the appendix to this chapter.

There is no incontrovertible evidence that the variation expressed by the probability functions used here, and by McCarthy *et al*. (2001*a*), are exactly those expressed in nature; however, they are the best approximations available. When patterns of ignition, weather, climate, fuels, landscape dissection and fire-suppression capability are changing rapidly, so too is the nature of the probability of burning taking place at any given point in the landscape. The mean interval between fires may change, as may the nature and extent of the variation about the mean. The possibility of peat-fire occurrence may rise in a drying climate, seasonality of fires may shift due to changes in ignition cause, and intensities may change according to new fuel situations as well as climate change.

The principles outlined in this chapter for predicting the chances of occurrence of various intervals between fires and various times since fire, using a simple random model and varying the fire-free period, are timeless although the situations to which they are applicable may change. The same principles used for modelling fire intervals and times since fire could be applied to the grazing situation described in the last chapter.

Chapter 5a Probability of burning at a point in relation to time since fire and between-fire interval

This appendix provides a rudimentary introduction to the subject only.

Probability of burning taking place at a point in the landscape

A point on a fire-prone landscape is burnt most often when a fire is ignited somewhere else but spreads to the point in question. The probability of a point in the landscape being burned (*PBP*) can be graphed as a function of the time since the last fire. In a mesic grassland, the fuel may be restored within a year and so the chance of burning a point a year after the fire is the same as it was the year before. In this case, the chance remains constant with time after fire and was the form of randomness used as a basis for the chapter; the nature of the probability function varies as the minimum interval increases. To illustrate, the *PBP* graphs for *P*=0.25 are given in Figure A.1. *PBP* equivalents are sometimes called 7'hazard' functions, a misnomer for this application as fire probability is not the same thing as a hazard to something or someone.

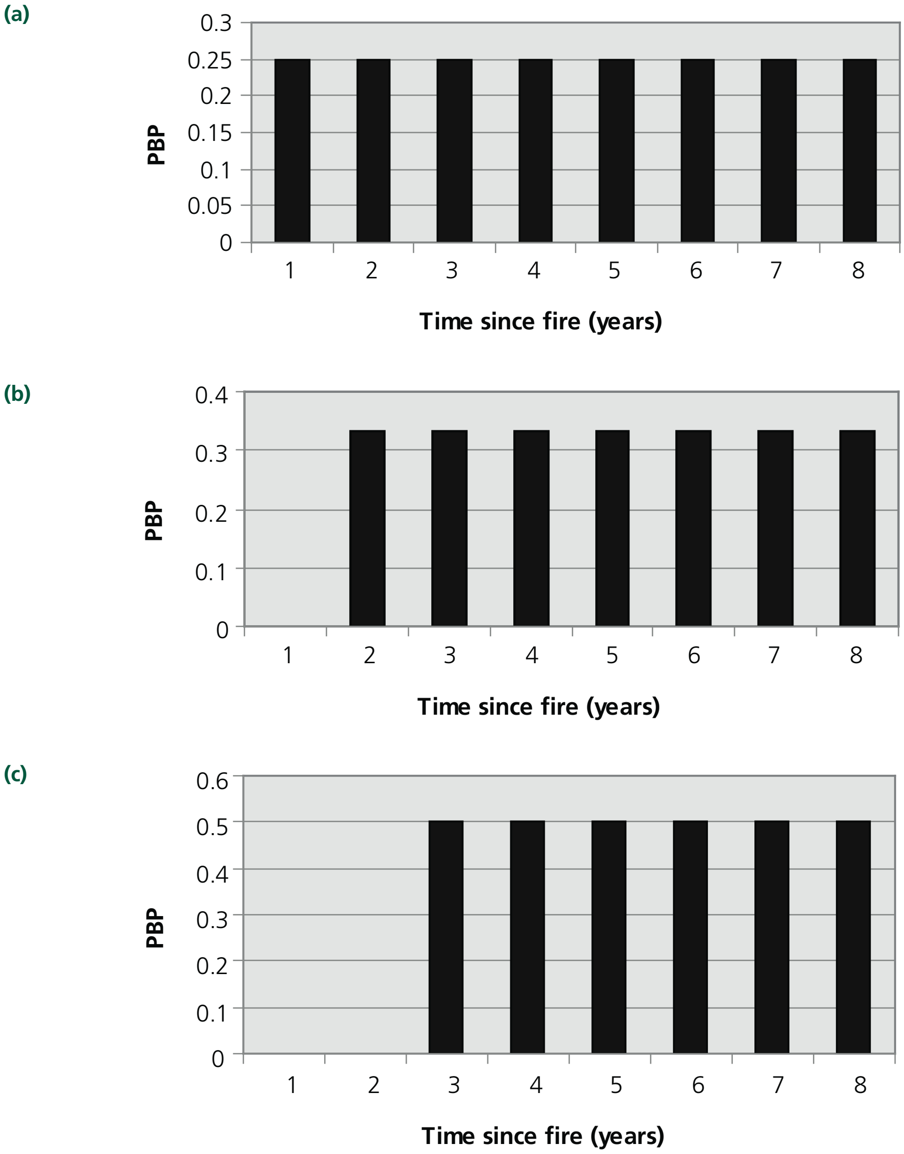
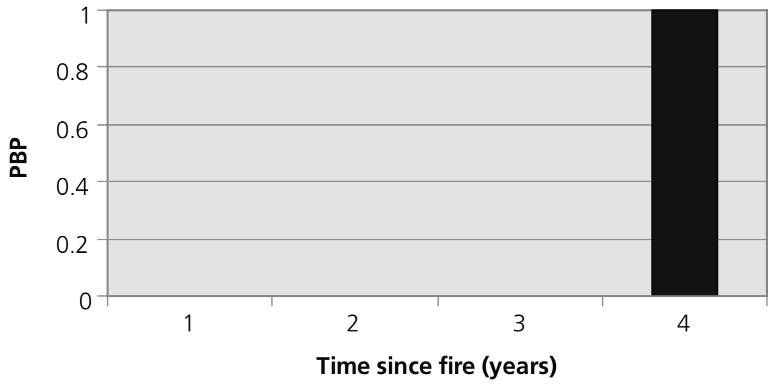


Figure A.1 Probability of burning at a point (*PBP*) as a function of time since the last fire in years. Minimum periods after the last fire affect the function, although the proportion of the landscape burned per year, *P*, remains the same. Note the change in scale of the Y-axis from (a) to (d). Minimum periods without fire are: (a) one year, the time step – a constant *PBP* with time; (b) two years – note the gap at year one; (c) three years; (d) four years – note change of scale on the X-axis.

(d)



PBP curves represent the combined effects of ignitions, topography, weather and fuels. In this sense, *PBP* functions can be regarded as basic. Also, they are basic because they give rise to the probability functions for time since fire and for between-fire interval. In those cases where sharply delineated minimum fire periods were suggested, a step in the *PBP* function occurs (Figure A.1 *b* to *d*). In reality, such sharp jumps from zero for the early years of the curve are unlikely. Knowing what the real curves are is a challenge influenced by a general lack of data and heterogeneity in the landscape. Rising to the challenge, McCarthy *et al*. (2001*a*) produced a series of hypothetical curves to depict a number of fire landscapes in Australia. They were able to show that the proportion of landscape with a particular fuel age, or time since fire, was more or less negatively exponential; however, the interval distributions arising from these more or less similar curves were quite different. This is an important practical point as time-since-fire data are more common than interval data. In McCarthy *et al.* (2001a), interval distributions included the negative exponential curve shown in Chapter 5, but also a series of somewhat distorted bell-shaped curves. This result highlights the importance of using interval distribution, rather than time-since-fire distribution, as a means of assessing the appropriate timing for burning. It then begs the question as to the most appropriate *PBP* curve.

Major points from McCarthy *et al.* (2001*a*) are:

- There are different *PBP* curves for different vegetation types
- The graphs of the proportions of landscape with different times since fire are all similar, while those for intervals are quite different
- When comparing interval distributions with time-since-fire distributions only those derived from a constant *PBP*, which is the negative exponential, are similar.

Heterogeneity within landscapes

The examples given so far have been for homogenous landscapes but this is never the case in reality. How will *PBP* curves, such as those in Figure A.1, vary as a result of heterogeneity? For our purposes, heterogeneity can vary:

- Within the same form of *PBP* (e.g. flat) but with two or more levels of *PBP*; a variant on this is that one level is zero – these areas do not burn at all
- Two or more different forms of *PBP*.

If the landscape has two levels of *PBP*, *viz*. 0.5 and zero (no fire) each applying to half the landscape, the average proportion burned per year will be 0.25. In this hypothetical and illustrative example, the bare areas are completely intermixed in the terrain. A negative exponential value equivalent to that for a mean interval of four years might be expected as a result. However, use of this value would be incorrect as the real mean for those areas that will burn is two years. Areas known not to burn, such as bare patches, should be removed from any analysis for fire interval.

If the landscape has two levels of *PBP*, the area of each covering 50% of the total area as above but both are above zero, the situation is more complex. For example, let's assume that one value is 0.5 and the other is 0.1. As in the above case, the distribution of combined values for each year for the study area will be half the value of those for a mean interval of two years (because P=0.5), plus half the values for a mean interval of ten years (because P=0.1). The combined result will not be the same as that for an interval of six years (average of ten and two); however, the combined probabilities will add to unity as each individual probability does. The combined distribution will be closer to that obtained when the proportion burned per year is 0.3 (average of 0.1 and 0.5) but will not be the same.

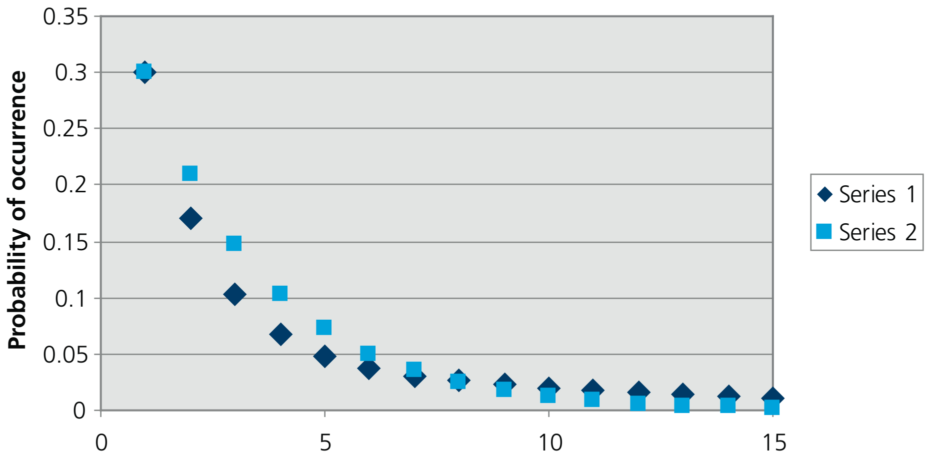
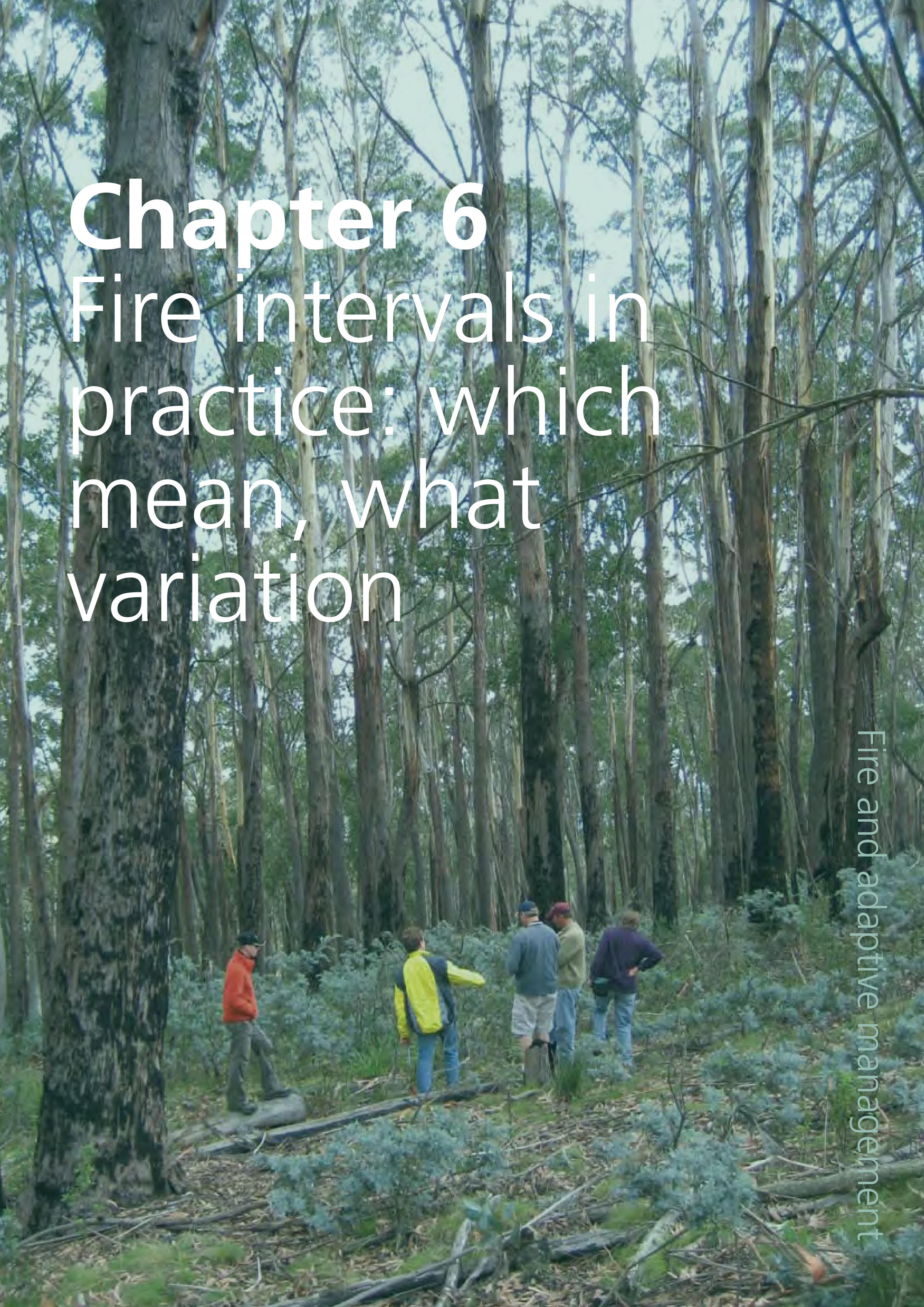


Figure A.2 Probability distribution of times since fire or interval for an area with half having *PBP*=0.5 and the other half having *PBP*=0.1 (Series 1). The combined distribution is compared with the distribution expected if the whole area had *PBP*=0.3 (Series 2).

Note that the two graphs in Figure A.2 are very similar; they overlap, which is necessary if the total probability in each case is to add to one. In the real world, these small differences may be impossible to detect.

Landscape heterogeneity is the reality. Where prescribed burning is practised and biodiversity conservation is the aim, avoiding the temptation to achieve 100% coverage by seeking out and burning unburnt patches – which reflect, in part, the underlying variation in landscape – is not recommended.



Chapter 6
Fire intervals in practice: which mean, what variation

Chapter 6 Fire intervals in practice: which mean, what variation?

Introduction

To the extent that fire regimes can be domesticated[1] – tamed – managers can determine the values of all the components of the fire regime and their variation. It is assumed that they will prescribe burn for fuel reduction or ecological purposes when conditions are deemed safe – particularly in forests because of the potential for very high intensities there. This usually means burning under mild weather conditions at a time of year when severe fire weather has a low probability of occurrence. It is a matter of choice which intervals are appropriate, and there are a number of ways this can be determined – the subject of this chapter.

There are three main ways in which the search for the most appropriate interval can be conducted in a biodiversity context:

1. Choose the mean fire interval that existed at the time of settlement
2. Choose to maintain the current mean fire interval
3. Create a new mean fire interval based on plant life cycles.

Each of these has its practical merits, but which mean, and what variation around it, is desirable for the purpose of conservation of biodiversity?

One school of thought is to predetermine a mean fire interval from various sources, such as those above, and then apply it – the calendar, recipe or prescription approach. Under this approach, an extreme, the manager responds to the calendar and never the resource and a guideline can become a recipe. Thus guidelines involving minimum and maximum tolerable limits (Wouters *et al*. 2002), recommended intervals (Tran and Wild 2000) or minimum and maximum intervals (Kenny *et al*. 2004) can easily become embedded in the management system. While the authors of guidelines may be cautious about them because they are based on data of variable comprehensiveness and quality, managers may not be as careful. They may not have the necessary staff or skills and time to check on them and may be inclined to leave it to the staff scientists.

The other school of thought (e.g. Gill and Nicholls 1989) is that it is not possible to know with certainty which mean interval – or indeed which fire regime – will achieve the aims of management most fully, so it should be worked out in practice. The manager responds to the resource, not the recipe. For this system to work, it is necessary to pay constant attention to what is happening to the resource and modifying management practice accordingly. An indicator can be used so that it will trigger the decision to burn when certain circumstances are attained; the efficacy of the indicator could well be checked also.

The 'calendar approach' is simplest and easiest, whereas the 'trigger approach' keeps the manager in touch with both the resource and the aims of management. Of course, it is possible to consider both approaches together. First, set a hypothesis for a desirable interval. Second, test it in the manner of the trigger approach. This adds a dimension of extra thought to the whole operation. However, what could happen is that, with a change of manager, the thought process is lost while the recipe becomes fixed. The key is to keep management as a vibrant process rather than one that is just a routine.

Fire intervals before white settlement

Policy makers may decide that the between-fire intervals presiding before white settlement are those that are desirable today, because these were the conditions under which our biodiversity evolved. This policy may be attractive as it implies that a return to these intervals means that biodiversity is maintained and monitoring is superfluous. This line of thought, implicitly or explicitly, assumes that:

- The flora and fauna has survived a regime, or regimes, of fires that was in place for thousands of years, up to the time of settlement
- The ecological, social and geographic context now is the same as it was in pre-settlement times, or has no effect on the outcome
- We know, or can find out, what that fire regime was before settlement
- We can reproduce the pre-settlement fire regime across the landscape.

1 Domestication may be considered undesirable by some managers but sought by others.

The biota has changed during the many thousands of years before white settlement. Taking a cue from the debate over the extinct Australian megafauna (animals weighing over 44kg) and the arrival of Aboriginal people (Johnson 2006), there have been major changes in biota, climate, fire regimes and human habitation during the last 50 000 years. These changes are reasonably well documented or inferred, even if the details are unknown (e.g. Miller *et al*. 1999; Roberts *et al*. 2001; Kershaw *et al*. 2002; Johnson 2006). Thus the environment in all its variety has been subject to substantial change in the past.

The rate of change has increased following white settlement. Change is obvious even within a human lifetime. There have been the effects of cities and ports, widespread agriculture, pastoralism and forestry, water harvesting, fire suppression, land-use heterogeneity, transport corridors and so on. Many native species have become extinct; others have expanded their ranges. There are many introduced species. Now there is rapid climate change, most likely caused by humans, which could have substantial effects on fire weather and fire regimes (Beer *et al*. 1988; Cary 2002; Hennessy *et al*. 2005; Lucas *et al*. 2007; Pearce *et al*. 2007).

Changes have occurred but they have not occurred to the same extent everywhere. Can we find out the regimes of the past? There are a number of ways the nature of the regimes of the past can be sought, for example, from:

- Fire scars on trees with annual rings (e.g. Banks 1989)
- *Xanthorrhoea* stems with annual increments and fire markings (for south-western Western Australia, see Lamont *et al*. 2003 and Enright *et al*. 2005; see Gill 2006 for commentary on the controversy over the method)
- Life cycles of plants and animals, habitat and age structure of vegetation combined in models to various extents to find plausible ranges of alternative regimes (e.g. McCarthy *et al*. 1999*b*; Bradstock *et al*. 2005; G Cook and A Liedloff, pers. comm., 2004; King *et al*. 2006)
- Anthropological investigations and explorers' notes (e.g. see Gill 2000)
- Varved or other sedimentary deposits of various types with included charcoal (e.g. Clark 1996; Kershaw *et al.* 2002).

These techniques can substantially improve our knowledge, particularly about fire interval, but may lack accuracy, spatial resolution, sufficient detail to ascertain seasonality and intensity, and ignore current social needs and the substantial changes that have taken place before, but especially after, white settlement.

In many areas, using effects that are observable now may be the best that can be done to establish appropriate fire regimes for the management of biodiversity. However, this is not a perfect system either. The biodiversity is only partially known and its interactions with unplanned and prescribed fires are even less well understood. Thus drawing evidence from as many sources as possible and evaluating the results seems to be the most appropriate way forward.

If a desired regime is declared, can it be implemented? To a large extent this may be possible in the more settled landscapes. Even then, however, there is a chance that unplanned fires will occur. Managers may also be constrained by budgets and other resources and numbers of opportunities to burn (see Esplin *et al*. 2003, Chapter 10, for a full discussion of the issue). Then there are often unwanted human-caused ignitions, and natural ignitions under extreme weather conditions, to complicate matters.

Intervals based on recent post-settlement history

The current pattern of fires may be deemed appropriate on the basis that a management pattern is already in place, costing is within bounds and there is no need, therefore, to alter anything. It has the dubious merit that it can take place without thinking about it too deeply. This system can also be the default system. The assumption, from a biodiversity point of view, is that there is no adverse change occurring – everything is as it should be.

In practice, such a system requires careful analysis as:

- The fire regime may not have been well defined – yet is believed to be so – because the records are incomplete or inaccurate and the scale is broad rather than sufficiently detailed to encompass fauna and flora that may live in limited areas with different fire regimes
- The fire regime may be changing but is not measurable at the time scale of management because of randomness in the system. If the mean fire interval is thirty years, for example, any change in the system, let alone the norm, is not readily detected over a period of a few decades
- Species may be in decline or expansion but population measures are not in place to show that this is the case. The changes from year to year (the 'noise') may be such that decline (the 'signal'), is not evident
- Co-variables may be changing – such as climate change – the effects of which are not readily apparent; the effects of weeds and feral animals may be undetected
- The regional geographic context of the reserve may be changing such that land-use influence on the margins of the reserve, at least, are having a chronic effect
- Tracks and visitor use may be changing and having an undetected effect.

Some of the methods mentioned in the last section may also be used to determine the current fire regime. In addition, the current interval and times-since-fire distributions can be determined from a series of annual fire maps set in a Geographic Information System. Determination is easiest where the length of record is long in relation to the mean fire interval; in Australia, this is in the savannas of the monsoon tropics where the mean is short (e.g. Gill *et al*. 2000, 2003). The mean interval is given by the reciprocal of the mean proportion of the fire-prone area burnt per year (the sum of all fire areas divided by the number of years of record and the fire-prone area), while times since fire can be determined directly. Intervals for individual pixels can be determined by longitudinal analysis – burrowing through the annual layers of the data atlas to find fires and measure intervals. However, given that there are inherent inaccuracies in mapping, a single pixel should not be used as a valid standalone measure of the interval distribution at that place. A frequency distribution of intervals based on maps of areas with presumed homogeneity is better, a process which depends on a knowledge of the ecological setting but which is bound to overlook some important local variation.

Frequency distributions of times since fire can look similar despite the interval distributions being very different (McCarthy *et al*. 2001*a*; Chapter 5). Thus it is better to use interval distributions rather than times-since-fire distributions as a guide to management (Gill and McCarthy 1998). The practical difficulty with this is that interval determination requires more data than time-since-fire data and there are few areas where data is long-term and reliable. The obvious response to this situation is to seek to map fires accurately each year and repeat this for long periods so that the problem is overcome in the long-term.

It is desirable to use a variety of methods when seeking to know the local fire regime, including fire interval. Mapping methods may fail to discern nuances of mean interval, or even considerable differences in mean interval, exhibited in small areas. Point measures from tree rings and fire scars can assist but may miss some fires (Richards *et al*. 2001).

Wouters *et al*. (2002) used the negative exponential age-class distribution (or time-since-fire distribution) as a benchmark for assessing the ecological burning status of Victorian plant communities, 'until further research offers a more sound and practical alternative model'. They used plant functional types and life history markers to determine the fire cycle (or mean interval) and to establish novel burning schedules for ecologically based management. This system is discussed later in this chapter.

Intervals based on plant species' characteristics: some background

A number of methods can be used to formulate desirable fire intervals for the conservation of biodiversity and fuel management, based on:

- Functional groups of plants – with the desired mean interval determined by monitoring plant responses of the most vulnerable types as indicators and measuring critical times in their life cycles
- The responses of a desired or major species – because of rarity, dominance or food-web significance (e.g. bird-attracting species)
- Functional groups of animals or knowledge of individual species
- Fuel load limits.

Some background to the use of plant characteristics as a way to guide the determination of fire intervals follows, as does a brief discussion of the use of fuel loads.

Functional groups of plants

Taxonomic groups are well known (species, genera, families) but functional groups much less so. Functional groups are groups of species that behave in similar ways. Members of these groups quite often cut across taxonomic lines, although some alignments may occur. 'Functional groups', as a term, will become familiar when examples of categories within a functional group based on life forms is brought to mind e.g. trees, shrubs and herbs. Trees, for example, can be of many species despite widely different taxonomic affiliations. Consider, for example, conifers and angiosperms, eucalypts and wattles, oaks and elms.

Functional groups defined in relation to fires have been created based on the death or survival of populations of species in the first instance (e.g. Noble and Slatyer 1980, the 'vital attributes' scheme). Adding a standard severity – degree of damage – as part of the criterion for 'death' is important in a classification of species based on 'function'. So too is a measure of plant maturity, as seedlings may respond differently to adults and young adults to senescent ones. The standard severity used by Gill (1981) was 'complete scorch'.

'Seeder' species are those with populations that die when fully scorched but survive as a species from seed. All species in which fully scorched populations die are in the same functional group but the method of species persistence varies according to the location of seed storage: for *Callitris* spp. and some eucalypts (Plate 5.1), seed is stored in woody fruits on the tree; for some wattles of this functional group, seed is stored in the soil; and for the mistletoes hemi-parasitic on woody species, seed must be sourced from outside the burned area (Gill 1996).

As well as mistletoes, some endemic Tasmanian conifers, such as *Athrotaxis selaginoides* (A Pyrke, pers. comm., 2007; Text Box 6.1), can be seeders with a need to source seed outside the burned area for their persistence. The dynamics of other conifers fitting this functional type have been studied in relation to fire and species' characteristics in North America, where even after large fires there are seeds able to disperse sufficiently from unburned sources to re-establish the species in the burned area (Greene and Johnson 2000).

Sprouters (Plate 6.1) provide a contrast to seeders (Plates 5.1, 5.2 and 6.2). Plants of species in this functional type, when just fully scorched, resprout. Sprouters may or may not have propagules that persist through the fire event (Pausas *et al*. 2004). Many leguminous sprouters have long-lived, soil-stored seed.

Plate 6.1 Eucalypts resprouting after the widespread Canberra, ACT fires of 2003 (Jean Geue 2005).

The third main category is species that are not visible at the time of fire but that respond to it, or the post-fire conditions, by germinating, growing, flowering and fruiting, mostly or entirely before the next fire. Species of plants in this group are called 'fire ephemerals', and are particularly important when dealing with the flora of arid and semi-arid regions; there, the numbers of species visible at the surface can more than double after a fire (Gill 1999*a*).

Plate 6.2 Shrubland of the dominant native seeder *Leptospermum laevigatum* (Coast Tea Tree) at Jervis Bay Territory. The most obvious species regenerating is the short-lived native shrub species – or ephemeral – *Solanum vescum*. Seedlings of the dominant are very short at this stage (Collins 2004).

Notice that this broad classification of functional groups does not account for the possibility that:

- Sprouters may succumb to repeated fires at short intervals (Noble 1997, pp. 51–53)
- Sprouters may sprout when fires occur in one season but have high mortality when fires occur in another (Noble 1997, pp. 51–53)
- Seeders may have provenances that resprout (see Gill and Bradstock 1992), even in small but perhaps ecologically significant numbers (e.g. *B. ericifolia* in Jervis Bay Territory, personal observation, 2005).

Explicit life history markers: age to maturity and life span

Functional groups, as broadly defined above, have no immediate value in determining mean between-fire interval because no time scale is necessary for their description. Their value rises rapidly when used in conjunction with time markers. Two time markers that are of particular significance are the 'time to first set seed' after germination and 'longevity' of the species.

The time to first flowering from seed – the juvenile period or primary juvenile period (Gill 1975) – is often used as a practical marker, rather than time to first set seed, as flowers are usually more visible than seed. Juvenile period can be defined as the number of potential flowering seasons after germination until flowering begins (expressed in years). This definition allows for seasonality in fire occurrence as well as the seasonality of germination and subsequent flowering.

Determination of the length of the juvenile period can be difficult for many perennial species due to variation between sites and variation in post-fire weather (see also below). Burrows and Friend (1998) noted that in south-western Australia, juvenile periods in the drier forests were 12–18 months longer than those for the same species in moister forests. In New South Wales, Benson (1985) found that first flowering can be restricted to a few particularly advanced plants, thereby implying that in good seasons or sites flowering may occur sooner than otherwise. Expressing juvenile period as a range of values is realistic.

While a juvenile period can be determined, a population having sufficient mature seed for reproduction is the ecologically significant state. Seed development can take a year once flowering occurs (Benson 1985), and first flowerings are not necessarily successful in setting seed. Bond and Van Wilgen (1996, p. 134) use the useful term 'age to maturation'; however, time to first seed production may be more explicit for non-ecologists. In Keith's (1996) review, juvenile periods for woody obligate seeders varied from one year (*Acacia suaveolens*) to twenty (*E. regnans*) for Australian plant species.

Knowing the age of a population at first seed set or when 50% plants have set seed, for example, can assist in predicting the vulnerability of a population to short between-fire intervals. If the population is a seeder that replenishes its population from canopy stored seed soon after fire, then the population is vulnerable to a second fire that scorches the crowns of the plant before there is seed available for replenishment (i.e. the between-fire interval is too short). Such species usually have short-lived seed once it is dispersed so there is no seed storage in the soil. If the seeder has dormant soil-stored seed that is only partially depleted by germination after fire, the species may survive a second fire that scorches the plant crowns within the time needed to set seed because viable seed remains in the soil. The population will die out when this seed store has been used up by repeated partial germinations following each successive fire.

Even reprouters can be at risk from repeated fires at short intervals if they cannot replenish seed stores between fires. Repeated firing at intervals less than the juvenile period – whether primary (i.e. after germination) or secondary (i.e. after sprouting; Gill 1975) – will eventually eliminate the species because plants are never able to produce seed during the inter-fire period, by definition, and death of plants will occur sooner or later.

Life span, or longevity, of a species is an important life history marker although a somewhat vague concept in practice, despite the fact that each individual plant will have an exact life span. Populations of plants show various 'thinning' curves with time, which represent the survival of the population after more or less continual dying out of individuals over time. Theoretically speaking, the longevity of a population is the period in years from first germination to the age of death of the last surviving individual. For a species, it is an even more difficult time to determine because of its likely variability. It is easiest and most accurate for annuals and biennials when the time step of concern is one year. For long-lived perennials, it is most difficult.

Longevity is important in relation to fire interval as some species only regenerate after fire. If the entire population dies from old age before a regenerating fire occurs, then the species becomes locally extinct unless there is some form of longer-term seed storage (Noble and Slatyer 1980). By way of contrast, some species can establish in the inter-fire period, an important component in the scheme of 'vital attributes' of Noble and Slatyer (1980). For seeder species with canopy stored seed, the critical measure of longevity may be the age at which the seed supply fails. This may be shorter than the longevity of the mother plants or equal to it.

By considering the life histories of plant species in different functional groups, an idea of which species will be most vulnerable under which circumstances can be determined. Then the most vulnerable species can be used as an indicator for the setting of an appropriate mean fire interval. For example, the species most vulnerable to short-interval fires may be an easily scorched seeder with the longest time to seed set and with a canopy seed store. Tall seeders, such as *E. regnans*, are less readily assessed as populations are not necessarily killed by every fire (see Mackey *et al*. 2002).

In nature, there can be a shifting mosaic of age classes (Clark 1991) evident in a landscape dominated by seeders dependent for their persistence on seed released at the time of the fire (e.g. *E. regnans,* Mackey *et al*. 2002). When a local population of the regional complex is killed, the population may be replenished by seed of its now-dead parent plants or from those plants within dispersal distance. Thus adequate dispersal can overcome the problem of juvenile plants being killed in some locations. In a conservation reserve, the question becomes whether or not the reserve area and plant populations are large enough to tolerate fires killing juvenile populations. There may be a chance that the depleted areas cannot be reached by dispersing seed. Managers may want to have a fire-free period long enough to ensure that no prescribed or other fire will impact on juveniles of such species.

If the minimum interval allowable for the maintenance of the species is set by the longest age to maturity of all the plant species present, and the maximum interval desirable is set by the longevity of critical species, then the framework is set for limits of fire intervals deemed to be suited to the long-term persistence of all plant species. What mean interval is appropriate given that variability is inevitable, even desirable? There are various methods for setting what appears to be a desirable mean interval in conservation reserves and these are found in the next section.

Choosing a between-fire interval: fuels and plant species

Here we examine two methods for choosing fire intervals. The first, fuel limits, is briefly discussed before a more detailed examination of the use of functional types of plant species, life history markers and models that help establish variation about mean fire intervals is conducted.

Fuel limits

Setting fuel-level limits as a trigger for prescribed burning indirectly establishes an interval for a fire regime. Such limits are usually based on a perception of the maximum amount of fuel, or limiting fuel condition that will allow fire control, even under extreme weather conditions (Gill *et al*. 1987b; Esplin *et al*. 2003). When fuel characteristics potentially allow a fire to occur with an intensity beyond the threshold for control, the fuel can be treated to reduce the probability of such fires occurring.

In grasslands, this method has less utility because the fuel loads could reach a quasi-steady state each year, irrespective of burning the previous year. However, out-of-season burning may reduce grass fuel loads in some places, such as the savannas of the Top End (northern part of the Northern Territory), where an early wet season fire may kill many germinating annual *Sorghum* plants – the main fuel species in some regions (Stocker and Sturtz 1966).

In steep country with erodible soils, litter fuel may be a stabilising influence (Good 1996; Leaver 2004), so a water manager desiring relatively high fuel loads to provide maximum soil cover, and thereby maintain stable soils, may also risk having an intense fire. Burning at short intervals may decrease the chance of uncontrollable fire but allow erosion to occur after each fire event. If high-intensity fires occur in heavy fuels over large areas there may be substantial erosion. There is no easy solution as to what the fire interval should be under these circumstances but the manager may be inclined to burn small areas under mild conditions outside the storm season (Gill *et al*. 2008). Prescribed-fire intensity may be low enough to avoid consuming all the fuel present, and such that the pattern and size of fires from spot ignitions may even restrict the extent of local soil movement. While this issue seems to be incompletely resolved scientifically, management guidelines based on expert opinion have been given (Good 1996; Leaver 2004).

'Tolerable' intervals based on plant species' characteristics

A very short fire interval can be adverse to many species; a very long fire interval can be similarly adverse. Sometimes this interval needs to be qualified, such as interval between tree-killing fires or interval at certain times of year; however, here, interval is considered in a general sense. Within the fire intervals possible in a plant community, there are *tolerable limits* (Wouters *et al*. 2002) or *critical domains* (Bradstock and Kenny 2003) for the persistence of species. In South Africa, Van Wilgen *et al.* (1998) considered a number of *thresholds of potential concern*, using variation in the components of the fire regime and fuel loads as criteria, to define critical domains. In this section, and the next, an appropriate variation in interval within a critical domain for species' persistence is sought.

Mean between-fire interval has been estimated by Wouters *et al*. (2002) for a range of Victorian plant communities by setting a minimum tolerable fire interval, a maximum tolerable fire interval and a value for the mean (fire cycle) equal to half the sum of these values. Most of the ecologically desirable mean intervals were estimated at thirty years, with the tolerable minimum fire interval being five to ten years and the maximum tolerable interval about 50 years. The theoretical negative-exponential distribution for a mean interval 30 years is shown in Figure 6.1, alongside data points for 52 intervals identified in a random-model series of 1500 years (sample mean 28.1 years). Note the variation around the ideal theoretical values.

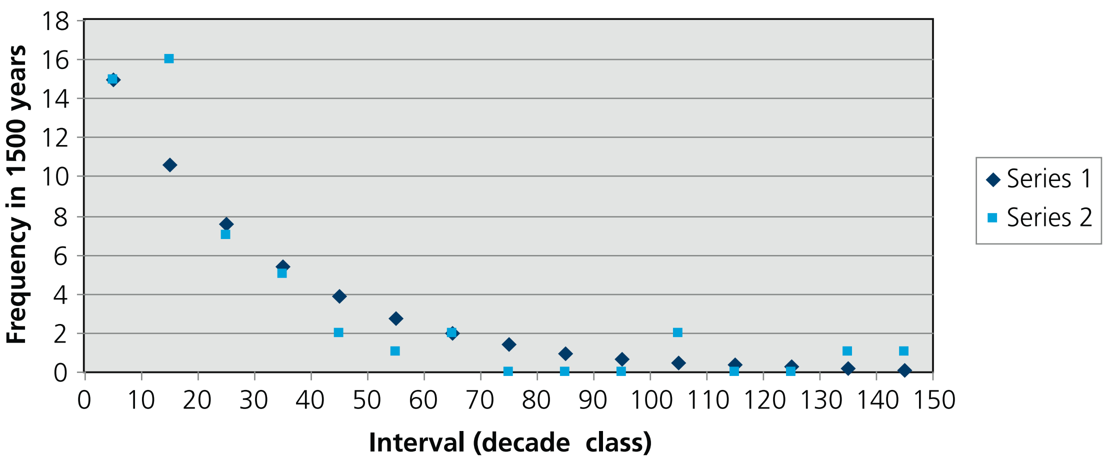


Figure 6.1 Negative exponential distribution of decadal fire intervals (or times since fire) from theory (Series 1) and from random selection over a period of 1500 years (Series 2). Points on the graph are at the middle of each decadal interval class.

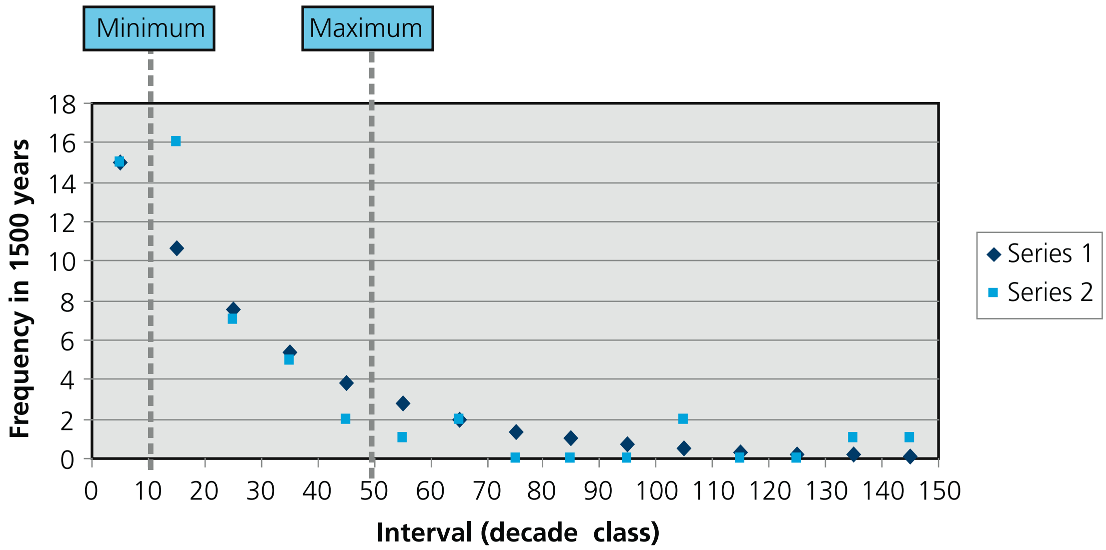


Figure 6.2. As for Figure 6.1 except with maximum and minimum tolerable limits as assessed for many Victorian plant communities (Wouters *et al*. 2002). Notice the proportion of intervals outside the tolerable range on both sides.

If the minimum tolerable fire interval represents a desired fire-free period to avoid extinction without relying on dispersal, say 10 years, while the mean interval of 30 years stays the same (the frequency of intervals is given in Chapter 5): a zero frequency for (M-1) years (equals nine in this case); and a mean interval as for (C-M+1) years for the remaining years (mean interval is 30-10+1=21 years). The results of this scenario are shown in Figure 6.3.

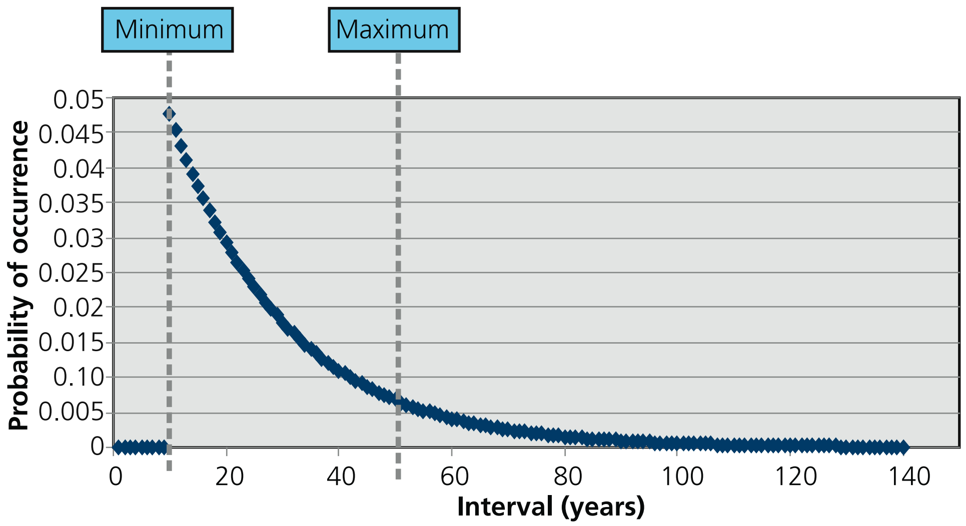


Figure 6.3. Theoretical fire interval distribution when there is a 10-year fire-free period and a 30-year mean interval. The interval scale is in years rather than decades. The minimum and maximum tolerable limits are shown.

If the minimum and maximum tolerable limits are to be kept but the distribution of intervals in Figure 6.2 exists, then there will be a chance of local extinction from intervals that are shorter than the minimum or longer than the maximum. If the minimum limit is taken into account and the maximum tolerable limit kept, then Figure 6.3 shows that there will still be a low probability of occurrence of intervals that are beyond the maximum (14%). It also shows that the species will become locally extinct as a result of such occurrences, perhaps only temporarily. Remember that especially the upper tolerable limits are somewhat vague. As a comparison, the probability distribution for a mean interval of 30 years, without a fire-free period greater than one time step, but using the Olson Model of McCarthy *et al*. (2001*a*) is shown with the tolerable limits of Wouters *et al*. (2002) (Figure 6.4).

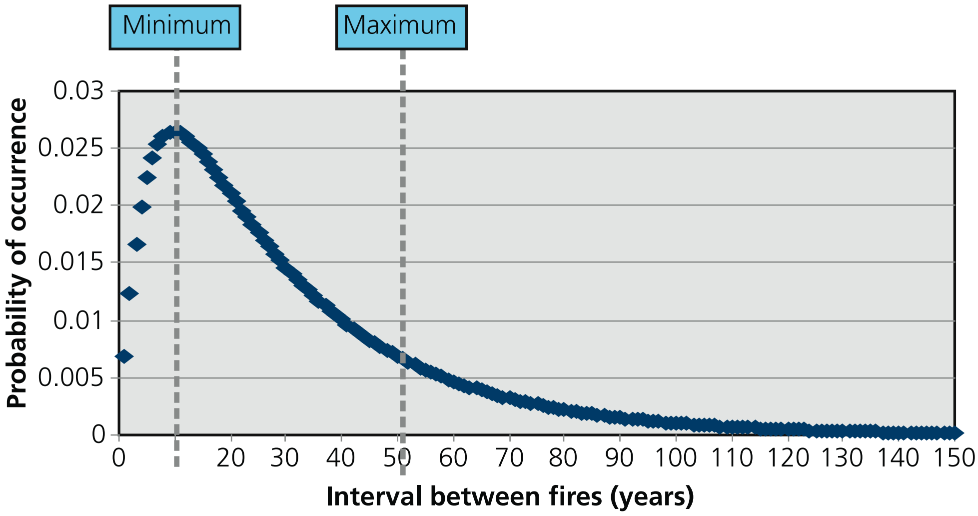


Figure 6.4 The minimum and maximum tolerable levels of Wouters *et al*. (2002) for the Olson Model of McCarthy *et al*. (2001*a*). This model is based on the concept that rate of spread of fire is proportional to fuel load, so the chance of a point in the landscape being burnt will be strongly influenced by fuel load.

If the juvenile period (or time to maturity) and longevity of woody seeder species are somehow mathematically related, then minimum and maximum levels may be set mathematically. There are more data available on matters of age to maturity and longevity in North America and South Africa than in Australia (for latter see Gill and McCarthy 1998). Plots of the scattered data available for Pinus in North America (from Fowells 1965), irrespective of response type, show that longevity is about 44 times the time to first seeding or sporulation (Figure 6.5). It is emphasised that these data are drawn from many sources and cannot be considered to have been collected in a uniform manner; nor are they all certain. This type of data may vary from place to place. Relying on such a relationship in practice, as opposed to theoretically in modelling, is unwise because of the scatter of values. In South Africa, age at first reproduction was statistically related to approximate life span for seeder shrubs (Frost 1984). The longevities of the North American trees far exceeded those of the South African shrubs, with the same age at first reproduction. These formulas should not be considered to apply to other tree species nor to the same species grown as ornamentals or plantations. It is likely that age to first reproduction is a function of growth conditions and that longevity is a function of temporal site variables (e.g. droughts and windstorms), as well as genetic predisposition and site quality.

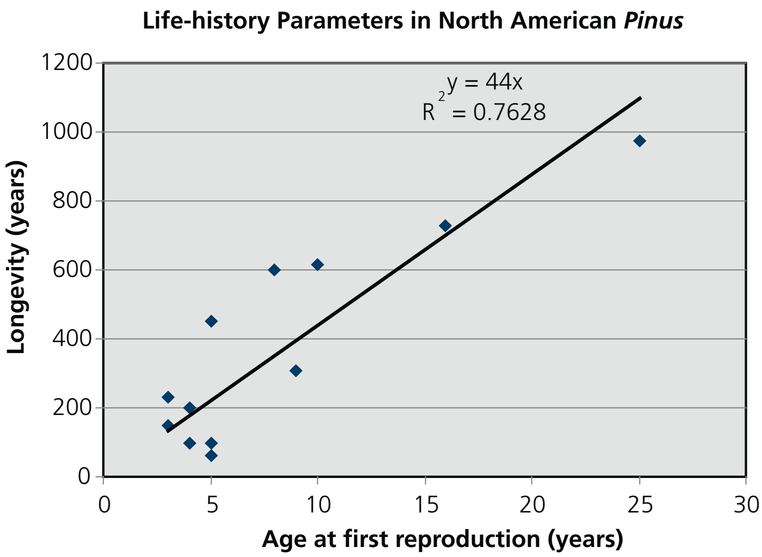


Figure 6.5 Longevity and time to first reproduction are apparently correlated in North American Pine trees (Fowells 1965). The data has been interpreted to some extent and should not be considered precise.

As there is difficulty in using uncertain life-history markers to determine prescribed-burning intervals, examining expected outcomes against actual outcomes is important (Tolhurst and Friend 2005). Let us assume that the mean interval is set to be the average between the minimum and maximum tolerable limits and these, in turn, are equal to the primary juvenile period (minimum interval) and longevity (maximum interval) (Tolhurst and Friend 2005). If two fires are set to occur at an interval equal to half plant longevity (i.e. close to the mean because the juvenile period is short in relation to longevity) and all obligate canopy seeder plants (in which scorched individuals die but seeds are stored on canopies) are too tall to be killed in the first fire, the second fire could occur when all the plants have just died of old age and so, presumably, have become locally extinct. While this may be unlikely, it is a possibility and the antithesis of what is desired.

Mean interval as a function of longest plant juvenile period

Gill and Nicholls (1989) suggested that seed production be considered adequate for replenishment of the population when two times the juvenile period had passed since the last fire, for the most vulnerable species – those that depend on seed stored in the canopy for regeneration after fire and have the longest juvenile period. Perhaps this period could be seen as a mean fire interval, or a desirable fire-free period (Burrows and Friend 1998). If this rule of thumb was used to establish mean interval, the less conservative option – a mean interval between fires of about 40 years – would be seen as a guide in *E. regnans* forest (Plate 5.1), which is not far from the estimated 37–75 years of McCarthy *et al*. (1999*b*). In the heaths of western Victoria, dominated by *Banksia ornata*, a mean of about 10 years between fires would apply (Gill and McMahon 1986). Longevity of the dominant *B. ornata* is about 50 years and regeneration occurs only after fire. Local extinction is likely if the interval is less than the juvenile period. If these figures are used with the negative-exponential model, they will show, as above, considerable probabilities of local extinction. In nature, without artificial boundaries, such as roads and farms, seed may be appropriately dispersed and the interval models may not be as simple as Figures 6.1 and 6.2 but more like Figure 6.4 or something else (M.A. McCarthy *et al.* 2001*a*).

In Figure 6.6, a distribution with a minimum fire interval equal to the primary juvenile period of five years and a mean interval of 10 years is calculated to see what proportion lies beyond 50 years. There are a negligible proportion of intervals greater than 50 years but note that there are considerable probabilities of fire in the period 5 to 10 years, when total restoration of dominant cover would not be expected. As with all these estimates, close checking of assumptions in the field, for fires that are prescribed or occur naturally, is wise so that revisions can be made accordingly. A case in point is when the species has produced seed in the presumed time. The situation described here is very similar to that of *B. ericifolia* around Sydney (Bradstock and O'Connell 1988), in which the possibility has been raised that competition by this dominant species may affect biodiversity (Keith and Bradstock 1994; Tozer and Bradstock 2002).

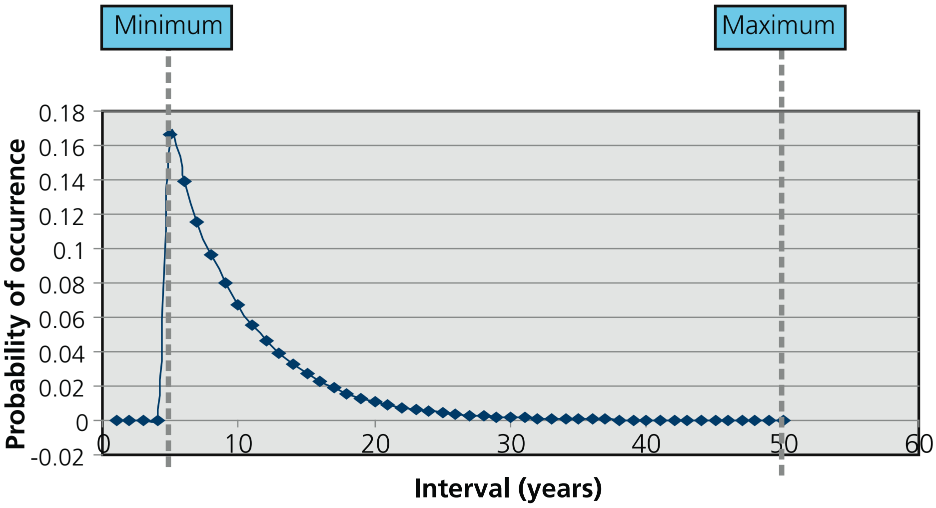


Figure 6.6 Heathland fire-interval distribution based on the dominant seeder species, *Banksia ornata*, with the minimum interval equal to the juvenile period, and the mean interval as the stand age ideal for replenishment of the population from seed (i.e. two times juvenile period). Longevity is believed to be about 50 years.

The significance of Figure 6.6 is that it implies no extinction occurs. The method has achieved, at least theoretically, variation within the domain of safety (i.e. no extinction). The way this variation has been achieved gives the minimum fuel levels possible while maintaining the species' presence and variation in intervals. Biodiversity managers might find that there is too great a chance of burning near the minimum limit and choose to use a more complex function, such as the Olson function (McCarthy *et al*. 2001*a*; Figure 6.4), with a minimum interval like that in Figure 6.6 as a guide.

A refinement to Figure 6.6 would be the use of time to first mature seed, rather than primary juvenile period. Similarly, the use of the mean fire interval determined by a doubling of this period could alter the graph. However, because of the likely variation in the juvenile period, the present graph may be taken to also represent the circumstance where seed set has also occurred by five years. The actual values need to be determined in the field.

That continual checking is desirable, and the use of the behaviour of the species in the field preferable as a guide to when to burn, is perhaps underlined by the likely variation in the juvenile period and the possibility of excessive dominance (Keith and Bradstock 1994). Checking that the method suggested is appropriate to all the species of the community, including sprouters that may be shaded out, would be worthwhile: Tozer and Bradstock's (2002) results suggest that *Xanthorrhoea resinifera*, a sprouter, would be a useful indicator in their study area.

Another method based on species' characteristics is Stochastic Dynamic Programming (McCarthy *et al.* 2001*b*) – a method also devised using plant characteristics, in this case *B. ornata*, a seeder shrub. It is designed to assist a manager in knowing when to burn, in light of the possibility of unplanned fire and the state of the plant community.

Animal species as indicators

Functional groups: fauna

If the flora is used as a guide to the fire regime needed for biodiversity conservation, does this jeopardise any faunal species? Are there other aspects of the environment that would be affected in consequence? There is no easy answer to these questions but the possibilities in relation to fauna are briefly examined here.

Faunal studies in relation to fire, such as plant studies, have often used time since fire as the major variable (e.g. Catling and Newsome 1981; Woinarski 1999; Catling *et al.* 2001; Keith *et al.* 2002), rather than between-fire interval, fire severity or a series of different fire regimes. A series of peaks in abundance of various animals often follows time since fire. However, the question remains as to whether all the species were present all of the time during the research study. Animals may be harder to trap when numbers are small, and they may be found in limited core areas where species are less sensitive to changes in time since fire than in the area as a whole (P Catling, pers. comm., in Gill and Bradstock 1995*a*; Brown *et al*. 1998, p. 34). In other words, there may be a spatially explicit component to survival in a fire event.

In interpreting faunal-population data, the central aim of management needs to be kept in mind. Is the aim to maximise the population size or simply seek species' persistence? The former may be important for a rare or threatened species, while the latter may be suitable in general terms. The overall aim in a conservation reserve is to avoid extinction of the local species' complement (Bradstock *et al.* 1995).

With respect to fires, the most obvious variable to examine in relation to the success or otherwise of animal populations is habitat, and this can be the basis for defining functional groups (e.g. Corbett *et al.* 2003 in Australian savannas; Bradstock *et al*. 2005). How do fires affect habitat and how do those effects influence populations of various animal species? The example of the tree-hollow dependent possums in tall forests in Central Victoria is a well-studied example (Lindenmayer 1996; Mackey *et al*. 2002). This small possum lives in hollows in older trees of Mountain Ash, *E. regnans*, in Victoria. Without these hollows the species perishes. The hollows are only formed in older trees that are themselves the result of specific fire histories (Mackey *et al*. 2002). Trees of *E. regnans* are readily killed by crown scorching, so it would appear that hollows in that case are not the result of fires. In the case of resprouting trees, fires may assist in the production of hollows but also affect their demise (see Gill and Catling 2002).

With ground-dwelling mammals, habitat has been measured using a habitat complexity score (HCS; see Chapter 3) (Newsome and Catling 1979). HCS is an additive score (of zero to three) for five sub scores in density of tree cover; density of shrub cover; density of ground herbage; cover of logs, rocks, debris (litter, Catling *et al*. 2001] etc on the ground; and moisture regime from dry to waterlogged. HCS is independent of plant species' composition. Numbers of animal species trapped were related to the score (Newsome and Catling 1979). This does not mean that a manager should strive to have most of the area in a state where the HCS is at a maximum; some species may prefer one end or the other or indeed the middle (Friend and Wayne 2003; Catling *et al.* 2001). As such, having a variety of system states (times since fire is the crudest measure) in a conservation area, seems worthwhile if all species are to be conserved.

In a 20-year study in south-eastern Australia (Catling *et al*. 2001), the abundances of ten relatively large ground-dwelling mammals were statistically related to:

- HCS alone (Bandicoot spp., Brushtail Possum – *Trichosurus vulpecula* and Ringtail Possum – *Pseudocheirus perigrinus*)
- Both HCS and time since fire (Potoroo – *Potorous tridactylus*, Wombat – *Vombatus ursinus*, Wallaby spp. and introduced Cat – *Felis catus*)
- Time since fire alone (Kangaroo – *Macropus giganteus* and Dingo – *Canis lupus*)
- Neither (introduced Fox – *Vulpes vulpes*).

Animal abundances in this study increased (e.g. Potoroos), decreased (Wallaby) or showed no trend (introduced Fox) with time since fire. HCS can rise with time after fire as shrubs, tree canopies and litter cover are re-established (Catling *et al*. 2001), while in the longer term, understoreys can diminish in cover and stature after a peak.

Among native birds there are species that occupy open habitats (e.g. the Australian Magpie – *Gymnorhina tibicen*) while others inhabit forest (e.g. the Lyrebird – *Menura novaehollandiae*), so it may be expected that there will be relationships between bird species and habitat structure as affected by fires (see Gill and Catling 2002). Among invertebrates, there is a range of responses to fires; the structure of the litter in forests, affected by fire regimes, is a case in point (York 1999). As the above examples indicate, habitat structure may be a general determinant of species' composition among all fauna, vertebrate and invertebrate. In turn, fire regimes affect both habitat and plant species' composition.

A further consideration is for animal species that are killed by fires with certain characteristics. The Koala, *Phascolarctos cinereus,* which browses or rests in certain species of eucalypt trees is largely exposed to any fire beneath. If the leaves are completely scorched, it may be expected that most koalas will die or be severely injured, the latter making them more prone to predation. If Koalas become locally extinct, then unaffected breeding populations can begin migrating back into the area. The fire regime suited to Koalas will depend on the chance of their extinction and of immigration (see Gill and Bradstock 1995*a*). The situation is similar to that of mistletoes (Gill 1996). Note that fires do not necessarily eliminate Koalas from an area either temporarily or permanently. The fires have to be intense enough to elevate temperature to tissue-killing levels (say 60 degrees Celsius) at all heights in the trees where Koalas are present.

Assessing plant species that are the most likely to become extinct locally from fire regimes, perhaps temporarily – including seasonality of fires and fire properties, not just interval – and that are critical elements of habitat, and using them as indicators, is worthwhile. Habitat in this case may be structural or related to food. An example of a critical food species is given by species of Mistletoe in Australia: certain animal species, such as the Mistletoe Bird (*Dicaeum hirundinaceum*), appear to be largely dependent on these species (see Gill 1996).

Three ways in which the local extinction of species or depletion of animal populations occurs have been identified:

1. Effect on habitat independent of plant species (tree hollows, HCS and other features not detailed here, such as foliage-height diversity)
2. Effect on plant species with special significance for certain animals (fruit, seeding time and flowering time)
3. Perhaps rarely, a strong effect on the animals themselves (e.g. Koala) – direct mortality of vertebrates from fire events is probably overemphasised, the predation in post-fire conditions perhaps being more important; many animals survive both fire and predation to rebuild post-fire populations when habitat is once again favourable (Recher 1981).

The assumption often made for practical reasons is that if plant species persist in a fire-prone environment then animal species will also do so. However, this is not necessarily true. Take hollows, for example, which may be formed decades after trees have established (see Gill and Catling 2002 and Mackey *et al*. 2002, p. 38); stands of young trees may be free of hollows whereas old growth may have abundant hollows. A fire regime is possible that will eliminate trees with hollows while maintaining the host-plant species. A further example is that of the Koala, mentioned earlier. If the feed trees of the Koala are repeatedly scorched over a sufficient area then the animals cannot re-establish in parts of the area, at least, so that the plants persist and the animals do not. The possibilities of discrimination between animal and plant species by fire regimes needs further research.

Using contemporary fire histories

Contemporary interval distributions

As a result of fire mapping, there is a certain amount of data available to discern a frequency distribution of intervals between fires or times since fire. A simple example indicates that the latter are more likely to be available than the former. In parks of the Sydney region, fire maps cover about 71% of the jurisdiction area and so times-since-fire data are available for this portion of the total area only (de Ligt 2005). It is most unlikely that fires are confined to this proportion of the target area, so the limitation is one of time of record rather than proneness. Less area is available for calculating intervals from these maps than time-since-fire data as two fires in any one place are necessary for an interval to be measured, whereas only one is needed for establishing time since the last fire.

Fires in conservation reserves may be unplanned or prescribed. For the determination of fire intervals in a reserve, the following points may need to be considered:

- Prescribed fires usually burn a smaller proportion of an area (because unburnt patches are more common) than unplanned fires. If only the perimeter of fires is mapped then there will be bias towards a shorter mean interval being recorded.
- Maps of prescribed burning are often of target area (blocks), not the areas actually burned. This has the same effect as above, shortening the mean interval.
- Scale is important for measuring all fires, unplanned or prescribed; maps have the potential to average the mean fire intervals present across a landscape (Chapter 5).
- Having a limited history of fire maps available is usual. If the record is short, the fire interval measured may be artificially long because large fires, not yet recorded, are a major contributor to the average proportion burnt per year and thus to mean fire interval (see Gill and Allan *in press*).

There is no suitable substitute for a long historical record comprised of accurate maps of all the fires that have occurred in the area.

Contemporary fire histories are likely to be most useful as a guide to managers when landscapes have a full complement of indigenous species and in which exotic species are rare or do not substantially alter fuels. Proliferations of exotic species can markedly affect fire regimes. In the USA, invasion of the exotic Cheat Grass (*Bromus tectorum*) into the shrub steppe of Idaho and Utah has changed the fire regime – formerly a fire burned once every 60–110 years but now occurs once every three to five years (Pimental *et al*. 2007). The growth of the introduced Gamba Grass (*Andropogon gayensis*) in the Darwin area of Northern Territory has dramatically increased fire intensity, while prescribed burning to minimise biomass in the late dry season means that it can be burnt twice a year every year in this monsoon–tropical area (Rossiter *et al*. 2003). Abundant growth of the semi-succulent shrub *Chrysanthemoides* sp. in coastal New South Wales is likely to cause decreased fire proneness there.

Times-since-fire frequency distribution

A hypothetical times-since-fire frequency distribution, as detailed in Chapter 5, may be seen as ideal and used as the basis for management (Tolhurst 1999; Tolhust and Friend 2005). The aim would be to intervene in the landscape so that the field reality conformed to a presumed ideal, such as a negative-exponential distribution. There are a number of difficulties here:

- Times-since-fire distributions, like other statistical distributions, have values that vary substantially from the ideal, depending on sample size. A smooth distribution is not to be expected, especially in small areas with short records.
- If the area is incompletely mapped for fires, the distribution is unlikely to reflect the correct mean interval.
- Times-since-fire distributions may appear to be similar to the 'ideal', yet arise from quite different circumstances; interval distributions may be quite different while times-since-fire distributions are similar (McCarthy *et al.* 2001*a*). Thus the times-since-fire distribution may look 'right' but the interval distribution – the fundamental variable – may be 'incorrect'.
- The 'right' answer might be made to occur by manipulating the most fire-prone areas, distorting the assumption of randomness.
- Unrestricted randomness raises the possibility that fires may have infinite intervals – the tail of the frequency distribution. Statistically, the real distribution would be expected to be truncated but where the tail should be truncated is unknown. If it is truncated, the probability distribution is compromised, in strict terms. Randomness seems to be part of nature but the nature of randomness is not fully understood. This is a problem with interval distributions as well.
- Maps provide a mean interval for the target area and fail to reflect the nuances of landscape and fire variation.
- A simple times-since-fire distribution, such as the negative exponential, provides a mean interval and a variation about the mean. However, in a fragmented landscape where the biodiversity of conservation reserves is isolated, managers may seek to guarantee survival of their species by declaring a fire-free period based on plant juvenile periods so that fires will not occur too frequently and cause local extinction (Tolhurst 1999; Chapter 5). If this is attempted while simultaneously aiming to attain an idealised times-since-fire distribution (as above), an inconsistency arises as the presence of a fire-free period compromises the distribution. The way this occurs has been detailed earlier in the chapter.
- If a conservation reserve is isolated from fires that previously started outside the reserve then spread into it (by changing land use and effective suppression), yet fires inside the reserve continue as before, the proportion of area burnt per year at the edges of the reserve and some distance into it will be reduced. Thus the intervals between fires at edges will have increased. If fire spread across the edge in a large area was equally in and out before establishment, then the between-fire interval would have been halved by its isolation. This is provided that the fires previously moving out of the reserve have been unaffected by new practices within it, or by the size of the reserve being less than the area of the largest fire. The contemporary regime at the edge may not be the historical regime.

Text Box 6.1. **Conservation of *Athrotaxis selaginoides*, an endemic Tasmanian conifer: a case history**

King Billy Pine (*Athrotaxis selaginoides*), an endemic conifer species predominantly of subalpine, high rainfall areas of Tasmania (Brown 1988), has populations highly sensitive to fire events (Brown 1996). Nearly one-third of the estate of the species has been estimated to have burned in the last 100 years (Brown 1988), suggesting an average annual rate of 0.33% per year or proportionally 0.0033 yr^{-1}. The inverse of this proportion suggests a mean interval between fires of about 300 years. The tolerable intervals of the species (Wouters *et al*. 2002) are estimated to be between about 100 years (time to first seed production) and 1000 years (estimated longevity) (see Gill 1996). So the appropriate between-fire interval to maintain it would be about 550 years, using the method of Wouters *et al*. (2002) and 200 years using the method of Gill and Nicholls (1987). Thus the estimated actual value falls between these two.

The mean intervals between fires recommended for the conservation of *Athrotaxis selaginoides* on the basis of life-history markers, above, are not necessarily appropriate, despite their general agreement with the current extent of burning:

- Fire sizes of the last 100 years may have been larger than previously. The species has synchronous, non-serotinous yet episodic seed production (Calais and Kirkpatrick 1983; Cullen and Kirkpatrick 1988), and dispersal distances may be short in relation to distances needed to recolonise burnt ground. Thus seed may not always be present for colonisation soon after fire, or the distances presented by large burnt patches may be too great to be reached through dispersal.
- The pattern of burning may not be random (Brown 1996), an assumption of the model, and overlapping fire areas may underestimate the total proportion actually burned.
- The simple calculations used above are based on uncertain values for the time to first seed production and species' longevity.
- Other factors may have come into play since white settlement, such as changed grazing patterns and seedling survival (M Brown, pers. comm., 2007).

The policy of active suppression of fires near *Athrotaxis* stands, as suggested by Brown (1996), perhaps in association with prescribed burning, would appear to offer long-lived species a safeguard for the potential adverse effects of any decreases in fire interval (i.e. increases in frequency) as a result of global warming.

Discussion and conclusion

This chapter explored the choice of fire intervals but it is important to emphasise that interval is just one of the variables involved in determining the effects of fires on biodiversity. Choices are difficult but the use of species' characteristics has obvious promise in a program of species' conservation. The method allows for the estimation of fire interval in small areas of vegetation, as well as large – an advantage over the use of fire maps where large areas may be needed because of the usual absence of depth in the fire record.

Uncertainty about responses to fire regimes remains an obstacle to the full understanding of how plants and animals will react to management. However, seeder species are useful indicators for creating hypotheses about between-fire intervals, especially when the only local seed available is found on the canopies of the plants. For an individual species, the upper and lower critical limits can be estimated. The longest time to first seed production of the suite of species present in the community can then be used to set the critical short-interval limit; the shortest longevities amongst these species – when there is seedling establishment only after fire – sets the upper critical limit.

Variation within critical limits of between-fire intervals for the survival of species has been a theme of this chapter and of Chapter 5. The nature of that variation and the critical limits are open to question but the use of frequency distributions of intervals within limits is a way forward. An alternative approach has been that of Bradstock *et al.* (1995) who made suggestions for intervals within the limits for the conservation of coastal heathlands near Sydney, based on their knowledge of the different responses of herbs and shrubs and their seedbanks. In the example of *B.ornata*, above, setting the minimum interval equal to the time to first seed set (juvenile period in Figure 6.6) – the mean interval to twice the juvenile period and variation based on a negative exponential distribution – gave a result within the estimated longevity (Figure 6.6), while other methods left room for a considerable probability of local extinction.

Consideration of variation within limits may be given to fire intensity and season of fire occurrence as well as interval. What this variation might be is a matter for examination in the species found in a particular reserve. Interactions, such as those between interval and season for a mallee eucalypt species (see Noble 1997, pp. 51–53), may complicate matters but critical limits may be identified.

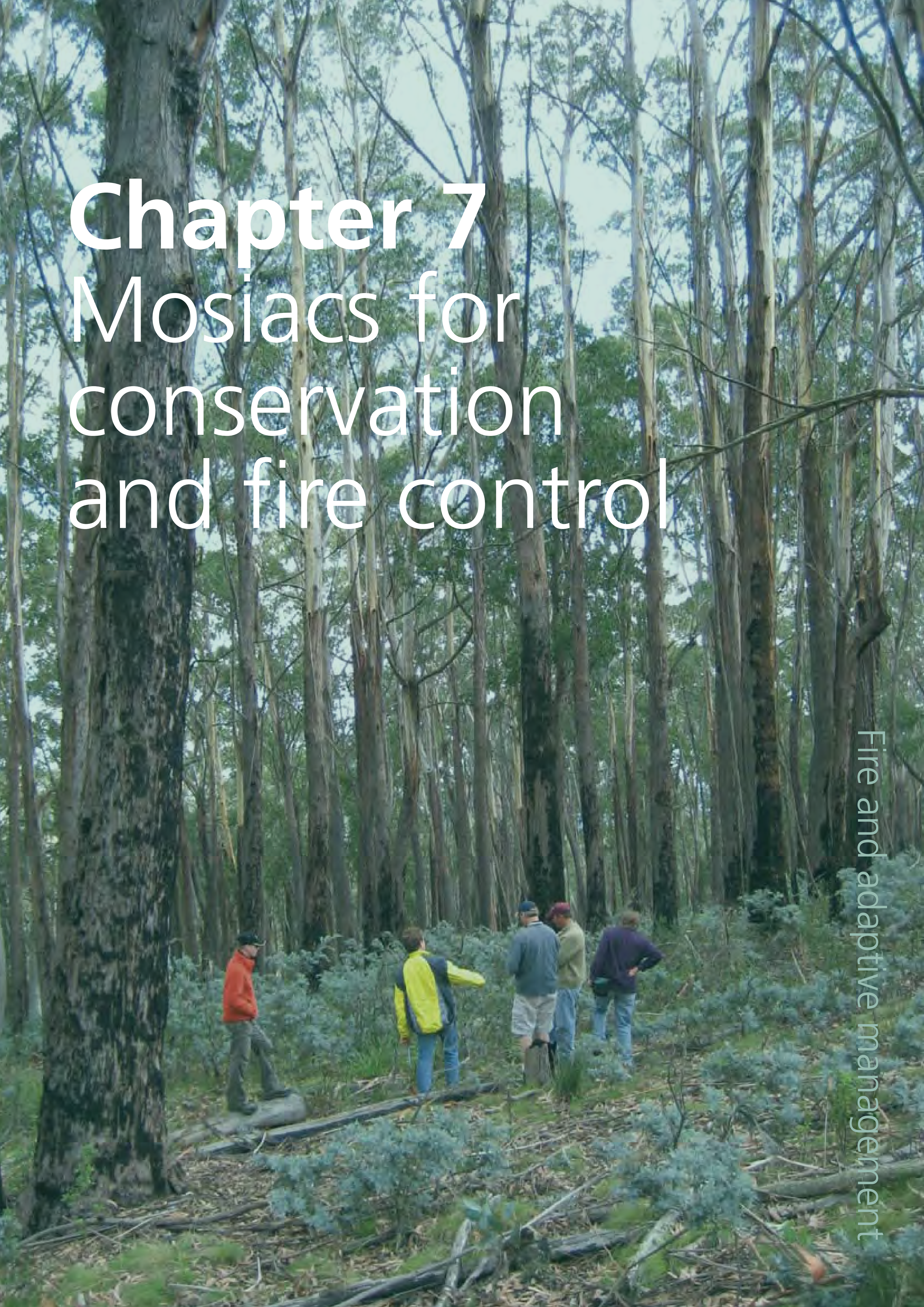
Variation within limits will enhance the performance of a species at one time and disadvantage it at another but without causing local extinction. Variation appropriately present will not allow a plant species to be so dominant that it is detrimental to other species and disadvantages them in the long-term (Tozer and Bradstock 2002). A rare, threatened or endangered species, however, may be targeted so as to improve its performance and thereby its chances of remaining extant.

In fragmented landscapes, more concern than otherwise may be expressed by the conservation manager when fires burn outside critical limits, because opportunities for re-establishment and replenishment of populations over time may be few. Compounding effects, such as predation of indigenous animal species by foxes, may also be likely.

Where fuel minimisation is used as the single criterion for interval setting, species may be lost locally or intervals set that put certain species near their shorter-interval limit. In those parts of large reserves where fuel levels may be less critical, the choice may be made to maintain intervals near the upper, longer, critical limit as a hedge against any decrease in interval length as a result of global warming. However, a number of fire-management strategies may be devised for dealing with the uncertain effects of global warming and these cannot be dealt with here.

There is much to be learnt. This brief review provides an introduction to the effects of fire regimes on ecosystems and landscapes in the face of change, including variation of their components. This section has been particularly concerned with effects of variation in time. In Chapter 6, some aspects of variation in space are considered.

Chapter 7 Mosiacs for conservation and fire control



Chapter 7 Mosaics for conservation and fire control

Introduction

Fire mosaics (Plates 7.1 and 7.2) have assumed paradigm status in many management circles (Gill 1986; Bradstock *et al*. 1995; Gill and Bradstock 1995*b*; Parr and Brockett 1999; Bradstock *et al.* 2005). Burnt patches of varying age form an interlocking mosaic that has been considered to inhibit (Griffin 1984) or stop the spread of unplanned fires, and is thought by some to be ideal for conservation. Brockett *et al*. (2001) expressed the idea of the perceived benefits of the ideal fire mosaic in southern African savannas, 'This should mimic the prehistoric or historic fire patterns, [thereby conserving biodiversity] reduce fire hazard and the costs of prescribed burning, and the costs of managing wildfires.' In this chapter, the use of mosaics for conservation is briefly examined, and their use for fire mitigation explored. Text Box 7.1 provides a set of terms relevant to mosaics.

Plate 7.1 Satellite imagery of visible fire footprints of different ages (e.g. 1999 fire at the far left, 2003 fire in the centre with the long corridor shaped 2004 prescribed burn to the south) across a dune field in the Big Desert National Park (LandsatTM, Feb 2007).

Text Box 7.1. **The nomenclature of patchiness of fires** (consistent with Gill *et al*. 2003)

- A *fire event* occurs when a fire arises from an ignition source (primary) or when a number of fires merge; it may be expressed on the ground as one or a number of burnt patches with the multiple patches having formed as a result of spot fires (secondary ignitions)
- A *fire's area* is the area enclosed by a fire's final perimeter(s) – the geographic extent to which a fire has spread
- *Unburnt patches* occur within a fire's area
- A *burnt patch* is a contiguous burnt area within a fire's area
- *Within-fire heterogeneity* can be expressed as patterns of varying fire severity (Plate 7.2) – due, at least in part, to various fire intensities – or as the pattern of unburnt patches; it can be affected by rockiness (Price *et al.* 2003), marginal intensities for fire spread and discontinuities in fuel
- *Mosaic burning* is the process of creation of a spatial pattern of burned and unburned patches from a single- or multiple-ignition prescribed fire
- A *burn mosaic* or *fire-created mosaic* (Plate 7.1) is the map of burned patches due to prescribed and/or unplanned fires of different seasons and ages. At any one time, a pattern of patches with different times since fire is present across the landscape – a *time-since-fire mosaic*. Similarly, patches of the latest between-fire intervals can be identified as a *between-fire interval mosaic*. The fire-created mosaic can be due to both unplanned and prescribed fires and vary temporally and spatially. In a broad sense, 'all fires are mosaic fires' (Horton 1982), so a description of patch sizes, shapes and arrangements is necessary to distinguish how the user differentiates these fires from any others (Williams *et al.* 1994; Gill *et al*. 2003).

Patchiness can also be seen in terms of the vertical structure of the plant community with various heights of charring, scorching or leaf survival – degrees of severity.

Plate 7.2 Differences in fire severity (seen as different colours) within and between vegetation types in a bog (foreground) and subalpine vegetation in the ACT, following the widespread 2003 fires (Gill 2003).

Fire-created mosaics and the conservation of biodiversity

Mosaics

The idea that particular mosaics are needed as a conservation measure has been expressed in different ways by a variety authors, but the theme is the same:

- '… it seems essential that prescribed burning should result in a mosaic effect if it is to simulate the controlled fires of the period before colonisation by European man' (Christensen and Kimber 1975)
- 'A patch mosaic system of burning is based on the premise that fire pattern is a surrogate for diversity' (Parr and Brockett 1999, in Africa), or 'pyrodiversity promotes biodiversity' (Martin and Sapsis 1991)
- The aim is a 'mosaic … at appropriate temporal and spatial scales' to 'protect and promote species and structural diversity of plant communities in forest ecosystems' (Burrows and Wardell-Johnson 2003, in south-western Australia)
- 'The aim of fire management should be the establishment of a mosaic of burnt and unburnt grassland during the dry season so that there is always some regenerating grass available for grazing [by native herbivores]' (Ettringham 1979, in Africa)
- '... it is generally acknowledged that fire-induced heterogeneity is important for maintaining diverse species assemblages in northern Australian savannas' (Price *et al.* 2005)
- A range of fire regimes across a landscape may enhance the coexistence of 'life history strategies' (Clark *et al.* 2002).

What is the evidence for the value of different fire-created mosaics for conservation of biodiversity? Can these statements be tested? Is any of the evidence based on the performance of one species, for example, in terms of population size, biomass or cover, or is it based on the chance of species' persistence or extinction?

It might appear obvious that mosaics of patches of different ages since fire are important to biodiversity, because there are obvious changes with time, such as a decrease in vascular plant species' richness[1] (Gill 1999*a*), or an increase in the richness of fungal species as shown by their fruiting bodies (Robinson and Bougher 2003; Robinson 2006). Thus if there is a diversity of patches at different times since fire, then the overall apparent diversity of a landscape will be higher.

It is possible that the total diversity of plant species in some areas is quite stable but the apparent diversity shows large changes with time since fire – as in parts of the arid zone (Gill 1999*a*). Species, such as the mistletoes, have no long-lived seed storage so they have no hidden form. (See also 'Plants', below.)

Animals

Burrows *et al*. (2006) drew attention to the association between the change from the pattern of burning of Australian arid-zone landscapes by Aboriginal people, to the contemporary patterns of burning and the decline and extinction of vertebrate animals within the critical weight range of 35 g to 5,500 g. They also noted that feral predators and competition from feral herbivores are other possible explanations raised in the literature. Bolton and Latz (1978), working in the central Australian arid zone, studied three population dynamics of a small wallaby, *Lagorchestes hirsutus.* These animals 'owe their existence to a tight mosaic of vegetation in various stages of fire succession' (Bolton and Latz 1978), because they depend on cover for protection from predators, but need newly burned areas for feed. A similar interaction occurs in wet forests in Tasmania, where small burned forest clear-cuts were the focus of native browsers that apparently decimated establishing seedlings of native trees. From a forestry point of view, the answer was to increase the size of clearings (Mount 1969). The Partridge Pigeon (*Geophaps smithii*) in the savannas of the Northern Territory is another example where a pattern of burnt and unburnt patches within the home ranges of the animals enhances abundance (Fraser *et al.* 2003). Cover is important for protection from predators, so the role of predators needs to be determined. These examples can be termed *shelter-feeder* examples, and are based on the population sizes of particular species of animals.

1 *Richness* is a count of species present that ignores species' identity. Richness could stay the same while the plant-community composition could change. If plants are not to be seen in vegetative form, they could be present as seed or other storage organs in the soil (see Gill 1999*a*).

Short and Turner (1994) examined the effects of a mechanically created mosaic on three species of small mammals. They found that the 'scale of mosaic proved to have no significant effect on … any of the three species.' Letnic (2003) examined three small mammals' responses to experimental burning of one-hectare patches in the Simpson Desert, 'Factors indicative of temporal and spatial variation in rainfall, time and site had a greater effect on the abundance of small mammals than the fire treatment.' Therefore, considering the studies above, it may be important to understand when, where and why certain animal species are affected and others not by various mosaics.

If a species is an old-growth specialist habitat, patches of that habitat are likely to be relatively small and dispersed quite widely compared with young growth. This is the arena of Hanski and Gilpin's (1997) *metapopulation dynamics* in which species may become extinct in some patches, but recolonise other small patches. Dispersal is important in such cases. If the patch is in some way the result of fire, then the distribution of patches in a certain format can become important to overall persistence. Leadbeater's Possum (*Gymnobelideus leadbeateri*) from the forests of Victoria is an example (Lindenmayer and Possingham 1996).

Two simulation studies illustrate some of the complexity of the situation. In the first of these, Bradstock *et al.* (2005) were concerned about the persistence of the Mallee Fowl, *Leipoa ocellata*, in a fire-prone dunefield. They examined the effect of a range of fire regimes using the demography, territory area and dispersal of this ground-dwelling bird. Prescribed and unplanned fires were simulated in relation to their effects on critical elements of the tree flora. They found that spatial aspects of fire regimes and aspects of fire management were important to the persistence of the bird.

In the second of the simulation studies, Barclay *et al.* (2005) looked at the effects of between-fire intervals and patch metrics on the presence of Mountain Pine Beetle (*Dendroctonus ponderosae*) in a landscape of its fire-susceptible host Lodgepole Pine (*Pinus contorta* var. *latifolia*) in North America. The beetle causes death to the host, especially when at harvestable age. In the simulation, large fire patches were not traversable by the beetle, while small ones were. Long fire cycles gave an age structure highly susceptible to attack because there were large areas of continuous habitat. Thus if timber was valued, beetles would be minimised at an appropriate patch density and scale. The original, natural intermix of beetles and pines seems to be unknown. In such simulations, patches are considered to be homogeneous.

The fourth tier of fire effects recognised by Gill (1998) is that animals may recognise heterogeneity greater than that encompassed by one vegetation type or ecosystem. Gill cites the cases of the Regent Parrot (*Polytelis anthopeplus*) and the Orange-bellied Parrot (*Neophema chrysogaster*) of south-eastern Australia, which nest in one type and feed in another.

Heterogeneity occurs both within and between fires at various spatial scales, causing various effects on elements of habitat for various animals. Some examples are:

- *Overlap between fire patches and home-range areas*. As the home ranges of animals varies by orders of magnitude, any effects of home-range overlap with fire areas (Gill 1998; Woinarski *et al*. 2005), even those of modest dimensions, will favour the populations of some species in some areas, but are unlikely to favour all the populations of all species in all areas unless patchiness within the fire is extreme.
- *A time-since-fire distribution of species' richness*. If the number or composition of animal species changes with time since fire, then there will be a greater diversity of animals in the area than there would be without the range of times since fire.
- *Heterogeneity within a time-since-fire patch*. Catling *et al.* (2001) demonstrate how their habitat complexity score varies within an area burned at the one time. This heterogeneity is likely to be due to variations in fire intensity and severity, and also variations in the vegetation itself.
- *Heterogeneity and time since fire*. In the study of Catling *et al.* (2001), their habitat complexity score varied from relatively low values soon after fire to relatively high values later on.
- *Heterogeneity through time* (*the fire regime*). This has been discussed in previous chapters.

From a conservation point of view, the persistence of a species as a population above a particular critical threshold may be more important than its population size above this threshold.

Plants

The effects of fire patchiness on the plant world are:

- **A time-since-fire distribution of species' richness**
  Gill (1999*a*) reviewed the Australian literature on this subject and suggested five types of trajectories with time for plant species' richness. If the common increase in richness soon after fire is ignored, these types become no change, a decrease to a lower richness, fluctuations in richness, increase in richness after a period of no change and decrease in richness after a period of no change. The last two could be regarded as subsets of the first pattern of no change. No change is assumed to be no change in species' composition or richness. When there is a range of times since fire in an area, then there will be a range of species' richness associated with this patterning, unless there is no change in the richness pattern with time since fire.
- **An interaction between patchiness in a fire event and seeder plant species with no in situ seed storage**
  There are certain woody species that are eliminated from a fire area when adult plants are killed. The latter group of plants has no in situ seed storage in the soil or on the plant. In North America, species with this habit include White Spruce (*Picea glauca*), Balsam Fir (*Abies balsamea*) and Larch (*Larix laricina*). If these species are to persist, seeds must disperse from surviving trees in unburnt areas into a burned area (Greene and Johnson 2000; see Chapter 6). Many Australian mistletoes (e.g. *Amyema miquellii*) that are killed with 100% leaf scorch need to migrate back from unaffected sources if the species is to have a presence (Gill 1996). Koalas (*Phascolarctos cinereus*) can have a similar dynamic (Gill and Bradstock 1995*a*). Thus the distance from unburnt populations becomes important in relation to the future presence of the species. The unburnt source of mistletoe may be above the scorch line of a tree with green foliage or from hosts outside the burnt area. In each of these cases, the presence of the species in a patch will depend on the fire regime (interval between plant-killing fires), the distance of seed dispersal and establishment of seedlings. There are similarities between these examples and that of the Mountain Pine Beetle simulation. Greene and Johnson (2000) found that even in large fires there appeared to be sufficient seed dispersal to reach all burnt ground from unburnt sources.
- **An interaction between patchiness in a fire event and seeder plant species with in situ canopy seed storage**
  A similar situation to (ii), above, occurs when plant populations are readily killed by fire, but afterward rely on seeds stored in the canopy for replenishment of populations. Such populations are readily eliminated by two fires at too short an interval. Patches depleted of these species may be restored from seeds of mature individuals affected by a subsequent fire. However, seed dispersal distances can be less than 50 m for shrub species (as found by Hammill *et al*. (1998) and Groenveld *et al.* (2002)), but over 1.6 km for a *Banksia* using molecular techniques (He *et al*. 2004). For eucalypts, seed dispersal distance is usually related to tree height as in the case of Mountain Ash (Mackey *et al.* 2002, p.35–36). For some species of North American seeder trees, dispersal distances from burnt edges were modelled as being up to 2 km (Greene and Johnson 2000).
- **A burning–grazing interaction**
  This occurs when herbivores concentrate on regrowth in burnt patches within their home ranges to the detriment of plant performance (cf. shelter-feeding habit for animals that may be to their benefit – refer to earlier section).

General

Mosaic effects add an extra level of complexity to the understanding of the affects of fire regimes on the biota. Thus it is not surprising that while 'it is generally acknowledged that fire-induced heterogeneity is important for maintaining diverse species assemblages … scant relevant data are currently available to examine this proposition' (Price *et al*. 2005); and that Bradstock *et al.* (2005) stressed the exploratory rather than predictive nature of their simulation.

The extent to which a given species is dependant on a particular mosaic remains to be seen. The problem is a difficult one to resolve because mosaics have many different patch sizes, patch shapes and patch dispersion patterns spread across a backdrop of various vegetation types. However, the fire regime with components of type, interval, intensity and season remains important for species' persistence, and each of these components may have their own mosaics at various scales.

Time-since-fire mosaics and fire-spread mitigation

Instead of stopping unplanned fires with a track network or some other system of linear fuel breaks of limited capacity (see Chapter 2), an area of land could be burnt in patches so that when a fire started in one patch it would soon run into one with little fuel and be automatically extinguished. Therefore a mosaic of burnt patches would be self containing, wouldn't it? Fire size would be limited and savings would be made on fire suppression. Apparently, this was the way Aboriginal people burnt the landscape. According to this view, 'Before Piranpa [white men] came, there weren't big bushfires because country had always been looked after by Aṉangu and such fires did not occur' (Muṯitjulu Community and Baker 1996, for central Australia). A patch burn strategy was introduced in Central Australia to 'act as a partial barrier – breaking up the front of any wildfire sweeping through the area' (Saxon 1984). This is a popular idea that seems to solve an important problem. Unfortunately, reality is not always as simple as this intuitive 'solution' suggests. In this section the idea of mosaics as a means of fire control is explored.

The formal expression of the idea of mosaic or patch burning was put forward in Australia in the context of the arid zone's Uluru Kata Tjuta National Park (Saxon 1984), the location of the Mutitjulu settlement mentioned above. In North America, Minnich and Chou (1997) contrasted the fire situations of the shrublands of southern California, USA with those of Baja California, Mexico:

> Land managers of Mediterranean wildlands should critically examine the 'well-managed' status of chaparral in Baja California as a 'showcase' for ecosystems functioning under natural disturbance. The spontaneity at which fine-grained patchiness develops without fire control in chaparral demonstrates that mosaics can be managed at long intervals (70yr) and at minimal cost because prospective fires are influenced by pre-existing patch structure.

These observations have been strongly contested by Keeley and Fotheringham (2001) who quote a number of authors when concluding that:

> Management plans that call for widespread prescription burning to 'recreate' a landscape mosaic of different age classes of vegetation will not stop large catastrophic fires.

Moritz *et al*. (2004) reached the same conclusion after examining fire maps to determine the probability of fire occurrence at a point as a function of time since fire. In most cases it proved to be near a constant rate of 2.7% per year. A blocking effect would require that early periods after fire would have a very low chance of burning, compared with later periods.

Let us consider what is needed for a perfect system (see also Gill and Bradstock 1998; see Chapter 2):

- **A pattern of burnt (no fuel or low fuel) areas is needed that will impede the spread of fire and contain it**
  One way of looking at this is topic to see whether a fire that started on one side of a mosaic of patches of different times since fire – in our conservation reserve – could or could not reach the other. This is the realm of *percolation theory* (Plotnick and Gardner 1993), and a simple model consisting of a square lattice (like a chess board) is used here by way of introduction. Let's say that the black squares on a chess board represent fuel and the white squares represent no fuel i.e. burnt areas. Now that fuel is present, a rule for the spread of fires in fuel cells is needed. One such rule is that fire can only spread from a black square (where fuel is present) to another black square through the sides of the squares.

  In the chessboard model of alternate black and white squares, the fire cannot spread across the board because any fire that starts will be confined to one black cell – it has no side in common with another black square. On the other hand, if the rule is that fire can spread through the corners of the cells, as well as through the sides, then it can travel right across the board and cannot be confined in the way envisaged. Thus the rules of the model system are important.

  The most common rule used for fire spread in percolation models is that fire spread only takes place through the sides of the square cells of the lattice. A random pattern of fuel presence is usually adopted for these models. Filled cells form clusters or groups of varying size (equivalent to unburnt patches in the field). It turns out that, for a large randomly filled lattice in which 59.275% of cells contain fuel, there is a continuous path for fire across the whole grid 50% of the time (Plotnick and Gardner 1993). So if a lesser proportion of cells are filled, then the fire will not be able to travel across the whole grid some of the time, perhaps even ever if the proportion filled is low enough – the ideal for our model mosaic.

  A related model to the percolation model also has fuel cells placed at random on a grid but has random ignitions as well. This is the *forest-fire model* (Turcotte 1999).

- **Patch sizes would be appropriate to the scale of fire spread mechanisms**
  Areas of unplanned fires (the burnt patches) vary widely according to fuel type, fuel continuity and weather, not to mention the efficacy of suppression. In a totally controlled system a series of burning blocks in which prescribed burning takes place might be used. These blocks are implicitly equivalent to the cells in the model lattice, as practitioners sometimes consider them to be barriers to fire spread. The issue of appropriate block size in the field is equivalent to that of the appropriate widths of fuel breaks discussed earlier (see Chapter 2). If a fire brand can travel 30 km and light a fire at the landing point (Luke and McArthur 1978, p. 102), then burnt breaks greater than this width may be considered necessary to stop any fire that occurs, even in the absence of active fire suppression. This suggests that the minimum sized patch would have to be 30 km square (90,000 ha), an area bigger than most reserves. Such a spotting distance is an extreme that has been noted only once, but spotting distances of the order of one kilometre are common. If the potential fire intensity was reduced throughout the mosaic, then the chance of fire brand production and the scale or grain of the mosaic could be reduced. Short-distance spotting may be of the order of '200 metres or so' (Tolhurst and MacAuley 2003). If the area outside the reserves was untreated, then long-distance spotting might apply.

- **Patch shape would not compromise patch size**
  If patches were aligned in one direction, but unplanned fires spread at right angles to this, then larger patches might be needed to attain suitable linear dimensions to prevent fire spread.

- **To be effective as a self-containment pattern, many cells are needed**
  One cell or patch is obviously not enough for a self-containment pattern. A fire in the surrounding area would just bypass the treated cell. A lattice of 3 × 3 cells or blocks is hardly enough either, because the edge cells dominate. An area of 10 × 10 cells would seem to be a minimum in a model system, and in reality a larger grid, by cell count, might be appropriate because in the field there is more diversity. Whatever system is used, a perimeter break may be needed.
- **Within-patch scales and pattern need to be sufficient**
  Especially within a burnt patch created by prescribed fire, there are always unburnt areas that could carry a fire at some time, producing fire brands. Therefore the pattern and extent of unburnt areas within a patch or burning block needs to be considered. Pattern within the block can be thought of in the same way as pattern between patches, but the pattern of burning may not be random. Managers may aim to burn around 70% of an area when undertaking prescribed burning (see Chapter 3 for a range of values). This extent is not always achieved. Thus even within a formal prescribed-burning block, the extent of burning is incomplete and may allow the passage of fire under less-than-extreme conditions. This is especially so in rugged terrain where edges of the block, ridges and exposed slopes may be burnt, but sheltered slopes and valleys may remain unburnt – a non-random pattern with respect to topography.

  A more complete burn may be achieved by successive bouts of burning carried out as the block dries out. Costs increase with this process, of course, and there may be biodiversity implications. If conditions suited to burning-brand production were minimised, then theoretically patch size can be smaller. Potential fire intensity can be taken as a surrogate for burning-brand production, even though this has limitations. As fire intensity will be maximised under extreme weather conditions, any reduction in the expression of factors, such as heavy fuels, stringy-bark or other sources of fire brands occurring on long, steep slopes aligned with strong winds, will assist. Fuel loads, fuel type (with respect to fire-brand production), weather and terrain are all implicated in any successful system of mosaic burning. A rate of unplanned ignition is also implied.
- **Appropriate pattern and scale needs to be sustainable**
  If the necessary pattern is not achieved at some stage, then the system can become ineffective. Retaining pattern means being able to keep the level of burnt areas – strictly speaking, areas that do not allow fire spread – at a suitable level. In the previous model percolation system, the non-fire-spread area would have to be kept to about 50% or more at all times; this could be made up of blocks with various times since fire up until the threshold for spread is reached.

  In reality, fire weather and terrain, not just fuel, are important variables. Thus low fuel loads can sometimes support fire spread during extreme fire weather so patches in the mosaic will vary temporally in effectiveness. Wetter areas associated with drainages or sheltered aspects can be effective in preventing spread when fire intensities are low but become dry, like other parts of the landscape, during droughts and then cease to have blocking capacity.
- **Aims of management would not be compromised in an ideal system**
  Criteria for assessing a mosaic-burning system, as for any other system, ultimately depend on whether or not land-management objectives are achieved.

The efficacy of a patch in passively stopping a fire is not easily defined. The fuel load (or other fuel property) sufficient to just carry an unplanned fire may be a positive but small load. In grasslands it may be less than in forest litter. The minimum could be associated with heterogeneity in fuel distribution, rather than in average load. Fuels in discrete patches, as in Spinifex grasslands (see Plate 7.3), need flames long enough to carry from one grass hummock to the next, and winds strong enough and suitably aligned to adequately tilt the flames to reach the next hummock (e.g. Bradstock and Gill 1993). If the gaps between Spinifex patches are from time to time filled with ephemeral species(e.g. due to unusually heavy rain), then spread is facilitated and will take place at a lower wind speed.

Plate 7.3 Discrete fuel array in Spinifex grassland, Central Australia. After an unusually wet period, the interstices can fill with short-lived grasses that form a continuous fuel bed when dry (Marsden-Smedley 2005).

Forest litter fuels in southern Australia may be burned during severe weather conditions, within a year or two of treatment (as with the 2003 fires in south-eastern Australia; see Chapter 5). Prescribed burning may not be possible so soon after a fire as it will not carry in the mild conditions needed for safety. Prescribed fires may not be desirable at such a short interval for environmental reasons. Thus there is the possibility that even with the most concentrated burning program, under extreme conditions some fires will spread across the fuel-reduced areas, preventing passive containment of the fire. However, it may assist in the suppression of fire when fire crews are present.

Three time periods of importance, defined by threshold conditions, can be identified in relation to the efficacy of patches for fire mitigation. They are:

1. The time of the last fire to the time of the first fire potentially able to burn at the same place i.e. during extreme fire weather
2. The time from the last fire to the time of the first *possible* prescribed fire
3. The time from any previous fire until the time of first *desirable* prescribed fire.

Operationally, patch burning, whether in formal blocks or in riskier unbounded (i.e. no fuel breaks) situations, is no panacea. Unbounded burning has a less-than-perfect record (e.g. Esplin *et al.* 2003, p. 89), as fire may escape the target area and become unplanned.

Discussion and conclusion

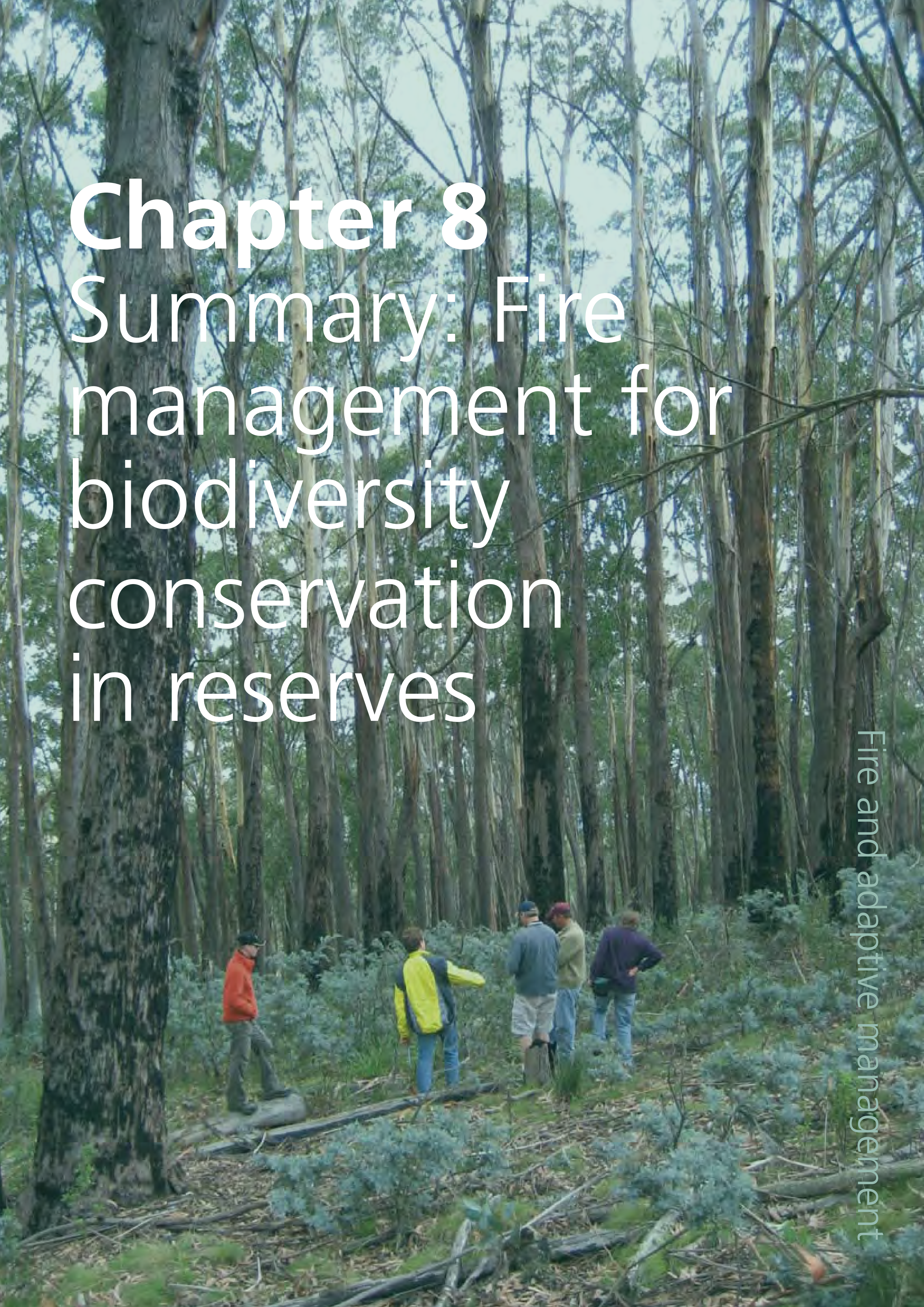
Fire-created mosaics consist of an interlocking set of patches of different sizes, shapes and arrangements. They may be described in different ways, consisting of burnt and unburnt patches in an unburnt matrix, patches created at different times since the last fire, or patches with different between-fire intervals. How the mosaic is seen will depend on the aims of management and operational considerations.

In a conservation reserve the apparent plant species' richness varies with time since fire (Gill 1999*a*), so a mosaic of times-since-fire patches can maximise the expression of plant richness at a landscape level. Similarly, animal species' richness can be maximised but the extent of interactions at patch edges, or between patches generally, is largely unknown.

The use of fuel-modification treatments for landscape fire management has benefits and problems that vary according to circumstances, such as terrain, fuel type and the aims of management. While fires may not necessarily be stopped by a particular arrangement of times-since-fire patches, they may assist suppression if the spread-inhibiting patches are large enough. The most likely patch-burning system to achieve passive fire control is one with a large number of blocks, with discontinuous fuels and a slow rate of re-establishment of fuels after fire (see Table 5.1).

'Sound, well established methods to design the spatial patterns of treatment application are still missing, and optimisation of the spatial arrangement of prescribed fire clearly needs further research' (Fernandes and Botelho 2003).

Effectiveness, cost and safe implementation of particular fire-created mosaics in an interventionist management regime are all important issues. As may be considered usual in such circumstances, the way forward is to carefully define aims, sensibly apply treatments and measure and review results so that each operation provides a learning opportunity. Monitoring should not just be for the effectiveness of the operation in itself but also for the achievement of the aims of management.



Chapter 8
Summary: Fire management for biodiversity conservation in reserves

Chapter 8 Summary: Fire management for biodiversity conservation in reserves

Ideally, management is carried out as an adaptive process in which a succession of procedures is continually revisited and revised so that management is evolutionary (Chapter 1). In this way, a management aim, such as avoiding the extinction of indigenous species in a fire-prone environment, while eliminating naturalised species, is increasingly realised. Monitoring of critical processes, as well as outcomes, allows the current understanding of the system to be enhanced, and allows continual adjustments to be made to management practices. Adoption of an adaptive management system in reserves is particularly important during the current period of rapid climate change, increasing human populations and further species' naturalisation.

Knowledge gained on the impact of management processes on biodiversity applies to the various circumstances and species of a reserve, but is not necessarily applicable to every species of every reserve. For example, the season of burning may affect one species in one place, but the season of burning could be irrelevant to all species found in another area. However, knowing that seasonality can be important at times, and that it may interact with between-fire interval, increases an observer's vigilance, and thereby leads to a program of adaptive management with respect to this variable.

In a world where increasing control over fire regimes is being sought through fire lighting and firefighting in reserves within 'fragmented' landscapes, a network of access tracks and roads is often necessary. While such a system may enhance visitor access to conservation reserves, there are many potentially adverse ecological effects of the road and track network (Chapter 2). Can aerial suppression replace the need for tracks? It seems not as trackless reserves appear unattainable where fire suppression and prescribed burning are essential. However, the extent of the advantages of track networks for fire management need to be assessed in the light of the extent of the adverse effects they can have on biodiversity conservation.

Tracks form fuel breaks that have value in mitigating fire spread up to a certain limit set by fire properties; broad-scale fuel management can modify relevant fire properties such as flame length, fire intensity and spot-fire potential. In south-eastern Australia, a common method for moving in this direction is the use of prescribed burning (Chapter 3). Its efficacy can be evaluated in relation to its success in fuel reduction, or its effect on fire spread and ease of fire control. But ultimately its effects need to be evaluated in light of the overall aims of management, such as biodiversity conservation.

The effects of prescribed fires on biodiversity can be considered in the context of the effects of both planned and unplanned (wild) fires. Fire's effect on species depends on fire regimes, that is, the history of seasons, intensities, intervals and types of fire. If species have adapted to *particular* fire regimes, they have therefore not adapted to others. In changing circumstances, as at present, some effects of regimes may appear soon after the change is introduced, but others may take many generations of fires to be fully revealed.

As an alternative to prescribed burning for fuel management in grassy ecosystems, grazing using domestic livestock has been introduced, or continued, in some places, even though a change of land use to a conservation reserve has occurred. Chapter 4 examines this contentious issue using the framework of grazing regimes, a parallel concept to that of fire regimes. While the effects of grazing on grassy fuels would appear obvious – fuel reduction – this is not necessarily the case where dominant fuel species are unpalatable and stocking rate is low. Complexity is apparent at every turn: native animals can overgraze, as can livestock; plant diversity may decline – a common historic experience with livestock grazing – or even be improved in some cases, while grazing regimes may interact with burning regimes in determining ecosystem outcomes.

Livestock grazing involves more than just herbivory. In practice, it involves an infrastructure of tracks, fences, watering points and yards. These can all have an effect on biodiversity, especially during installation and maintenance. Perhaps more importantly, grazing effects involve many aspects of an animal's presence, such as trampling, defaecating, urinating, carriage of seeds, rubbing of tree bark, creating drainage channels, camping and breakage of woody plants.

To many conservation managers, determining the fire regime that will achieve their biodiversity objectives is a major concern. Will the knowledge of past regimes in their area serve as an ideal template for the future? Can this regime be determined? Can this regime be maintained? Focus is often primarily on the between-fire interval and Chapter 5 explores and illustrates some of the theory relating to this – the underpinning of practice. As variation about a mean interval is recognised as important to some animal and plant species, and as irregular occurrence of fire at a point in the landscape seems natural, the nature of randomness and regularity for fire intervals is described. While some form of randomness is necessary for the conservation of some organisms, regular burning may be a feature of fuel management for the protection of economic assets.

The *manner* in which the landscape is burned affects the range of intervals that result, even if the mean interval remains the same (Chapter 5). The proportions of the landscape having different times since fire and different between-fire intervals, yet the same mean interval, varies with the degree of randomness of burning and the length of the fire-free interval after a previous fire. Formulae are developed for these variables and circumstances. What the optimum interval and its variation may be depends on the purpose of management and the nature of the system being managed.

An important message from Chapter 5 is that working with interval is more significant than working with time since fire, because the former is what a species may respond to in the long term. Furthermore, interval distributions can vary widely, even though their mathematically related time-since-fire counterparts can look quite similar. This creates difficulties from a fire-management point of view, because more fire-history information is necessary to provide intervals than time since fire, and such information is often incomplete.

In Chapter 6, various historical, biological, fuel and time-since-fire methods for determining the fire intervals that might be suited to conservation reserves were considered. The importance of a comprehensive, long-running program of accurate fire mapping to determine contemporary intervals was emphasised.

The inclusion of a fire-free period after the previous fire, as a means of allowing plants to attain reproductive maturity, creates a shift in what has been considered previously to be the ideal times-since-fire frequency distribution, the negative exponential distribution. The use of a range of methods to determine the shape and mean of a desirable fire-interval distribution, rather than a times-since-fire distribution, is recommended, along with a program of monitoring its effects on biodiversity.

Consideration of the effect on species' conservation of the variation in fire interval about a mean value (chapters 5 and 6) elicits the idea of the critical domain of intervals for species' persistence – a set defined by upper and lower bounds. Such bounds can be set according to particular plant species' attributes (functional groups), and the times of significant events (such as age at first reproduction, age at death or last seed production). Variation within critical limits is seen to be better than having ever-constant levels of variables of importance, such as between-fire interval. The concept of *appropriate variation within a critical domain* for biodiversity conservation is important. Our information on species' attributes is growing, but not complete. Assessing the status of key species in the field as a guide to the timeliness of prescribed fire or the need for protection from unplanned fires is recommended.

A dichotomy of approaches to management of fire for biodiversity can be drawn. On the one hand there are managers who desire a calendar prescription that is to be followed slavishly, while on the other the ideal is to assess the condition of the system in the landscape at appropriate times and react in light of current knowledge of fire-regime effects. While observation of pertinent species' characteristics and the flowering and fruiting calendars of plants are perhaps obvious, examination of even relatively robust species (with respect to fire regimes) in highly competitive situations could be considered.

Spatial variation in prescribed fires, within and between prescribed burning blocks, has been strongly advocated at times as a means of achieving conservation or fire-protection objectives. Spatial variation can also be caused by a succession of fires creating a landscape of patches with different times since fire, or more abstractly, with different between-fire intervals.

Spatial variation in fire-created patchiness can affect plant and animal responses and a number of examples are given in Chapter 7. Spatial patterning can affect fire spread, but its efficacy depends on the nature of the fires of concern and the nature of the fuel pattern. Patch size, shape and arrangement may all have a place in determining outcomes for fire suppression and conservation, but further details and case histories are needed.

Ecosystems are complex and knowledge of them is incomplete. Reserves for biodiversity conservation are isolated in seas of farmland and forestry enterprises, and even towns and cities. Climates are changing, human populations are burgeoning, suppression capacities are growing and ignition rates and numbers of naturalised exotic species are increasing. Funding, resources and staff can be limited. The management challenge is formidable, but the extent of gains made in the last 30 years in terms of knowledge of ecosystems and how they work, management planning and the implementation of field operations is also formidable.

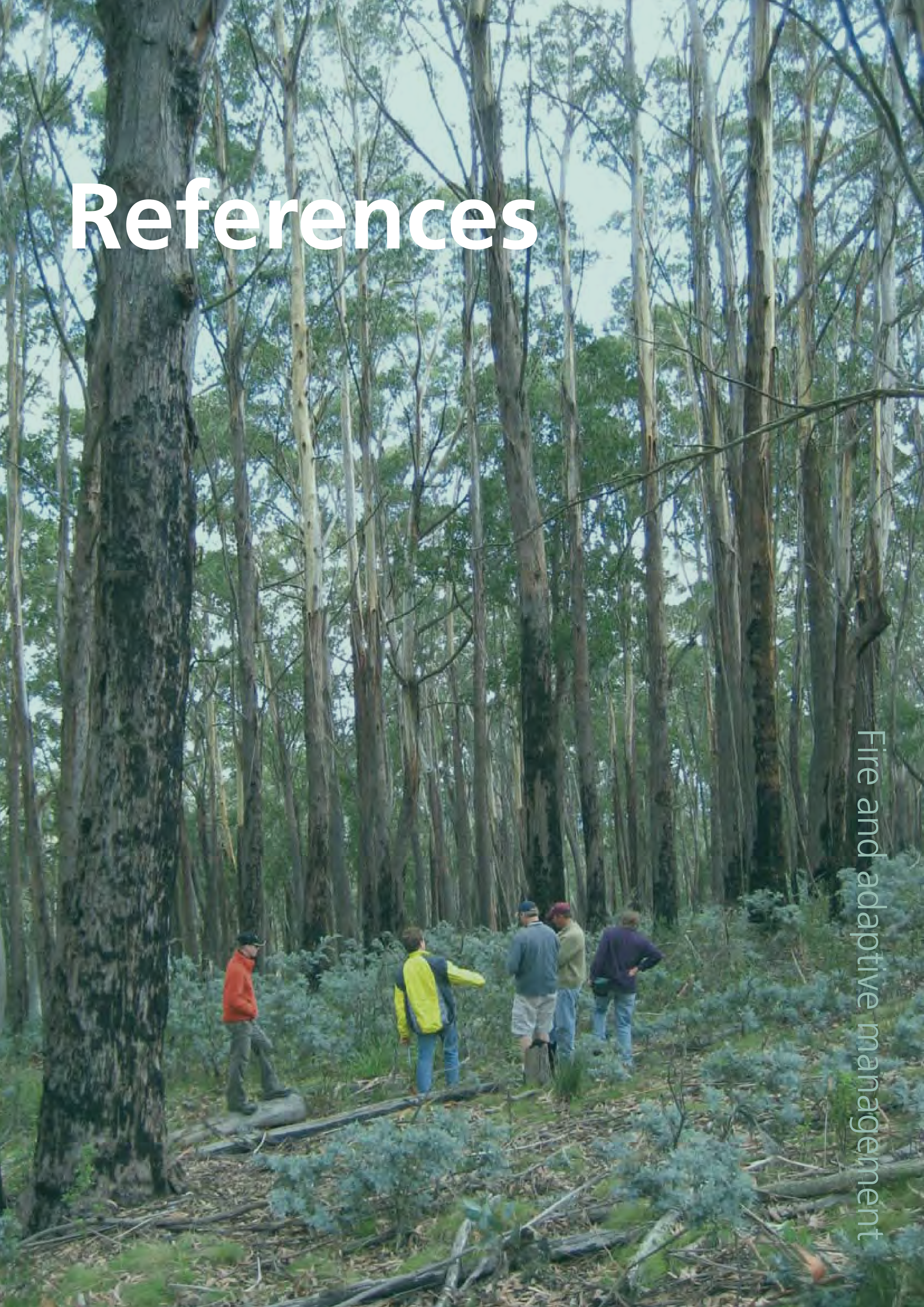
Much of our knowledge is to be found in the literature, but local knowledge especially is also invested in local people (Gill 1977). While scientific and agency literature raises awareness of issues, local knowledge is necessary to ascertain the applicability of scientific findings.

Seeking to understand appropriate domains in which to work to achieve management aims is a desirable goal. There are limits to the suitable extent of track networks, widths of tracks, lengths of fire intervals, intensity of grazing and areas of fire patches, for example. Knowing what these limits are in different circumstances and why is a challenge for researchers and managers alike.

The development of more sophisticated computer simulations to explore specific management options is anticipated. The value of these can be enhanced by what may seem by astute observers to be everyday observations. There are important roles for field naturalists, professional land managers, firefighters and scientists to play in the continual process of seeking better outcomes for fire-prone reserves set aside for the conservation of biodiversity.

This publication sets out to stimulate thought and action aimed at the effective conservation of Australia's unique and fire-prone biodiversity, a natural heritage. Reserves for nature conservation are rarely set aside for one purpose only. They may have the additional aims of protecting human life and property, providing a clean water supply to neighbouring cities and towns, preserving cultural history and promoting visitor use. With a multiplicity of aims in a climate of change, the need for careful thought, effective management systems and a knowledge network becomes particularly important if we are to achieve sought-after outcomes.

References



References

Abbott, I. (2003). Aboriginal fire regimes in south-west Western Australia: evidence from historical documents. In 'Fire in Ecosystems of South-west Western Australia: Impacts and Management.' (Eds I. Abbott and N. Burrows.) pp. 119–146. (Backhuys Publishers: Leiden.)

Abbott, I. and Burrows, N. (Eds) (2003). 'Fire in Ecosystems of South-west Western Australia: Impacts and Management.' (Backhuys Publishers: Leiden.)

ACT Bushfire Service (undated). Operations Handbook. [Accessed winter 2004].

Adam, P. (1995). Urbanization and transport. In 'Conserving Biodiversity: Threats and Solutions.' (Eds R. A. Bradstock, T. D. Auld, D. A. Keith, R. Kingsford, D. Lunney and D. Sivertsen.) pp. 55–75. (Surrey Beatty and Sons: Sydney.)

Adams, R. and Simmons, D. (1999). Ecological effects of firefighting foams and retardants: a summary. *Australian Forestry* **62**, 307–314.

Alexiou, P. N. (1983). Effect of feral pigs (*Sus scrofa*) on subalpine vegetation at Smokers Gap, ACT. *Proceedings of the Ecological Society of Australia* **12**, 135–142.

Allan, G. E. and Southgate, R. I. (2002). Fire regimes in the spinifex landscapes of Australia. In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent. (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 145–176. (Cambridge University Press: U.K.)

Allcock, K. G. and Hik, D. S. (2004). Survival, growth, and escape from herbivory are determined by habitat and herbivore species for three Australian woodland plants. *Oecologia* **138**, 231–241.

Anderson, A. N., Cook, G. D. and Williams, R. J. (Eds) (2003). Fire in Tropical Savannas. The Kapalga Experiment. *Ecological Studies* **169**. Springer, New York.

Andrews, A. (1990). Fragmentation of habitat by roads and utility corridors: a review. *Australian Zoologist* **26**, 130–141.

Archibald, S., Bond, W. J., Stock, W. D. and Fairbanks, D. H. K. (2005). Shaping the landscape: fire-grazer interactions in an African savanna. *Ecological Applications* **15**, 96–109.

Ashton, D. H. and Williams, R. J. (1989). Dynamics of sub-alpine vegetation in the Victorian region. In 'The Scientific Significance of the Australian Alps.' (Ed. R. B. Good.) pp.143–168. (Australian Alps National Parks Liaison Committee and Australian Academy of Science: Canberra.)

Auditor General, Victoria (2003). 'Fire Prevention and Preparedness.' (Government Printer: Melbourne.)

Australian Capital Territory (2004). 'Strategic Bushfire Management Plan for the ACT Version 1.' (ACT Government: Canberra.)

Baker, W. L. (1989). Effect of scale and spatial heterogeneity on fire-interval distributions. *Canadian Journal of Forest Research* **19**, 700–706.

Banks, J. C. G. (1989). A history of forest fire in the Australian Alps. In 'The Scientific Significance of the Australian Alps.' (Ed. R. Good.) pp. 265–280. (Australian Alps National Parks Liaison Committee and Australian Academy of Science: Canberra.)

Barclay, H. J., Li, C., Benson, L., Taylor, S. and Shore, T. (2005). Effects of fire return rates on traversability of lodgepole pine forests for mountain pine beetle (Coleoptera: Scolytidae) and the use of patch metrics to estimate traversability. *Canadian Entomologist* **137**, 566–583.

Beer, T., Gill, A. M. and Moore, P. H. R. (1988). Australian bushfire danger under changing climatic regimes. In 'Greenhouse. Planning for Climatic Change.' (Ed. G. Pearman.) pp.421–427. (CSIRO: Melbourne.)

Bell, T., Tolhurst, K. and Wouters, M. (2005). Effects of the fire retardant Phos-check on vegetation in eastern Australian heathlands. *International Journal of Wildland Fire* **14**, 199–211.

Bennett, A. F. (1991). Roads, roadsides and wildlife conservation: a review. In 'Nature Conservation 2: The Role of Corridors.' (Eds D. A. Saunders and R. J. Hobbs.) pp. 99–118. (Surrey Beatty and Sons: Chipping Norton, NSW.)

Benson, D. H. (1985). Maturation periods for fire-sensitive shrub species in Hawkesbury sandstone vegetation. *Cunninghamia* **1**, 339–349.

Bolton, B. L. and Latz, P. K. (1978). The Western Hare Wallaby *Lagorchestes hirsutus* (Gould) (Macropodidae) in the Tanami Desert. *Australian Wildlife Research* **5**, 285–293.

Bond, W. J. and Van Wilgen, B. W. (1996). 'Fire and Plants.' (Chapman and Hall: London.)

Bonneh, O., Ginsburg, P. and Woodcock, J. (2004). Integrated forest fire management in Israel. A 15 year review (1987–2002). *International Forest Fire News* **30**, 72–88.

Bradstock, R. A. and Gill, A. M. (1993). Fire in semi-arid mallee shrublands: size of flames from discrete fuel arrays and their role in the spread of fire. *International Journal of Wildland Fire* **3**, 3–12.

Bradstock, R. A. and Kenny, B. J. (2003). An application of plant functional types to fire management in a conservation reserve in southeastern Australia. *Journal of Vegetation Science* **14**, 345–354.

Bradstock, R. A. and O'Connell, M. A. (1988). Demography of woody plants in relation to fire: *Banksia ericifolia* L.f. and *Petrophile pulchella* (Shrad.) R. Br. *Australian Journal of Ecology* **13**, 508–518.

Bradstock, R. A., Keith, D. A. and Auld, T. D. (1995). Fire and conservation: imperatives and constraints on managing for diversity. In 'Conserving Biodiversity: Threats and Solutions.' (Eds R. A. Bradstock, T. D. Auld, D. A. Keith, R. Kingsford, D. Lunney, and D. Sivertsen.) pp. 323–333. (Surrey Beatty and Sons: Sydney.)

Bradstock, R. A., Tozer, M. G. and Keith, D. A. (1997). Effects of high frequency fire on floristic composition and abundance in a fire-prone heathland near Sydney. *Australian Journal of Botany* **45**, 641–655.

Bradstock, R. A., Williams, J. E. and Gill, A. M. (Eds) (2002). 'Flammable Australia: The Fire Regimes and Biodiversity of a Continent.' (Cambridge University Press: Cambridge, U.K.)

Bradstock, R. A., Bedward, M., Gill, A. M. and Cohn, J. S. (2005). Which mosaic? A landscape ecological approach for evaluating interactions between fire regimes, habitat and animals. *Wildlife Research* **32**, 409–423.

Bradstock, R. A., Bedward, M., Kenny, B. J. and Scott, J. (1998*a*). Spatially explicit simulation of the effect of prescribed burning on fire regimes and plant extinctions in shrub lands typical of south-eastern Australia. *Biological Conservation* **86**, 83–95.

Bradstock, R. A., Gill, A. M., Kenny, B. J. and Scott, J. (1998*b*). Bushfire risk at the urban interface estimated from historical weather records: consequences for the use of prescribed fire in the Sydney region of south-eastern Australia. *Journal of Environmental Management* **52**, 259–271.

Brazil, M., Rubinstein, J. H., Thomas, D. A., Weng, J. F. and Wormald, N. C. (1997). Minimal Steiner trees for rectangular arrays of lattice points. *Journal of Combinatorial Theory Series A* **79**, 181–208.

Brockett, B. H., Biggs, H. C. and Van Wilgen, B. W. (2001). A patch burning system for conservation areas in southern African savannas. *International Journal of Wildland Fire* **10**, 169–183.

Brown, J. K. and Smith, J. K. (Eds). Wildland Fire in Ecosystems: Effects of Fire on Flora. USDA Forest Service General Technical Report RMRS-GTR-42-volume 2. Ogden, Utah, USA.

Brown, K. L., Gadd, L. S., Norton, T. W., Williams, J. E. and Klomp, N. I. (1998). The Effects of Fire on Fauna in the Australian Alps National Parks: A Database. The Johnstone Centre, Charles Sturt University, Report 118. Albury, NSW.

Brown, M. J. (1988). Distribution and Conservation of King Billy Pine. Tasmanian Department of Lands, Parks and Wildlife. Hobart, Tasmania.

Brown, M. J. (1996). Benign neglect and active management in Tasmania's forests: a dynamic balance or ecological collapse? *Forest Ecology and Management* **85**, 279–289.

Brown, P. L. (1963). (Ed.) 'Clyde Company Papers V. 1851–53.' (Oxford University Press: London.)

Bunnell, D. L. (1995). Prescribed natural fire planning considerations: negotiating conflicting goals. USDA Forest Service General Technical Report INT-GTR-320. pp. 55–62. Ogden, Utah, USA.

Burrows, N. D. and Friend, G. (1998). Biological indicators of appropriate fire regimes in southwest Australian ecosystems. *Tall Timbers Fire Ecology Conference Proceedings* 20, 413–421.

Burrows, N. D. and Wardell-Johnson, G. (2003). Fire and plant interactions in forested ecosystems of south-west Western Australia. In 'Fire in Ecosystems of South-west Western Australia.' (Eds I. Abbott and N. Burrows.) pp. 225–268. (Backhuys Publishers: Leiden, The Netherlands.)

Burrows, N. D., Burbidge, A. A., Fuller, P. J. and Behn, G. (2006). Evidence of altered fire regimes in the Western Desert Region of Australia. *Conservation Science Western Australia* 5, 272–284.

Bush Fire Coordinating Committee (NSW) (2005). Bush Fire Risk Management Planning Package. Facilitator's Guidelines. Policy Document, Sydney.

Bush Fire Coordinating Committee (NSW) (2006). Management of Bush Fire Operations. Policy Document, Sydney.

Bushfire Council of New South Wales (1970s). Huge firebreak system planned for western N.S.W. *SITREP* extract.

Byram, G. M. (1959). Combustion of forest fuels. In 'Forest Fires: Control and Use.' (Ed. K. P. Davis.) pp. 61–89. (McGraw Hill: New York.)

Calais, S. S. and Kirkpatrick, J. B. (1983). Tree species regeneration after logging in temperate rainforest, Tasmania. *Papers and Proceedings of the Royal Society of Tasmania* **117**, 77–83.

Caling, T. M. and Adams, R. (1999). Ecological impacts of fire suppression operations in a small vegetation remnant. In 'Australian Bushfire Conference Bushfire '99 Proceedings.' pp. 69–75. (Charles Sturt University: Albury.)

Carr, S. G. M. (1962). The role of shrubs in some communities of the Bogong High Plains. In 'Proceedings of the Royal Society of Victoria.' **75**, 301–310.

Cary, G. J. (2002). Importance of a changing climate for fire regimes in Australia. In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 26–46. (Cambridge University Press: U.K.)

Cary, G. J. and Bradstock, R. A. (2003). Sensitivity of fire regimes to management. In 'Australia Burning. Fire Ecology, Policy and Management Issues. (Eds G. Cary, D. Lindenmayer and S. Dovers.). pp. 65–81. (CSIRO Publishing: Melbourne.)

Cary, G. J. and Golding, C. K. (2002). 'Fuel Modelling for Fire Management. A Review for the Northern Directorate of the New South Wales National Parks and Wildlife Service.' Unpublished.

Cary, G. J. and Morrison, D. A. (1995). Effects of fire frequency on plant species composition of sandstone communities in the Sydney region: combinations of inter-fire intervals. *Australian Journal of Ecology* **20**, 418–426.

Catchpole, W. (2002). Fire properties and burn patterns in heterogeneous landscapes. In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 49–75. (Cambridge University Press: Cambridge, U.K.)

Catling, P. C. (1991). Ecological effects of prescribed burning practices on the mammals of south-eastern Australia. In 'Conservation of Australia's Forest Fauna.' (Ed. D. Lunney. pp. 353–363. (Royal Zoological Society of NSW: Mosman.)

Catling, P. C. and Newsome, A. E. (1981). Responses of the Australian vertebrate fauna to fire: an evolutionary approach. In 'Fire and the Australian Biota.' (Eds A. M. Gill, R. H. Groves and I. R. Noble.) pp. 273–310. (Australian Academy of Science: Canberra.)

Catling, P. C., Coops, N. C. and Burt, R. J. (2001). The distribution and abundance of ground-dwelling mammals in relation to time since wildfire and vegetation structure in south-eastern Australia. *Wildlife Research* **28**, 555–564.

Cheal, P. D., Day, J. C. and Meredith, C. W. (1979). 'Fire in the National Parks of North-West Victoria.' Victorian National Parks Service, Melbourne.

Cheney, N. P. (1979). Bushfire disasters in Australia 1945–1975. In 'Natural Hazards in Australia. Proceedings of the Symposium of the Australian Academy of Science, Canberra. 1976'. (Eds R. L. Heathcote and B. G. Thom.) pp. 72–93. (Australian Academy of Science: Canberra.)

Cheney, N. P. (1994). Bushfirefighting and occupational health and safety. A national perspective. In 'Proceedings of the Bushfire Fighting and Occupational Health and Safety Conference, Sydney, 3 June 1994'. (Worksafe Australia: Sydney.) 10 pp.

Cheney, N. P. and Bary, G. A. V. (1969). The propagation of mass conflagrations in a standing eucalypt forest by the spotting process. In 'Mass Fire Symposium, Canberra, 10–12 February 1969. Collected Papers', paper 46.

Cheney, N. P. and Gould, J. S. (1995). Fire growth in grassland fuels. *International Journal of Wildland Fire* **5**, 237–247.

Cheney, N. P. and Sullivan, A. (1997). 'Grassfires: Fuel, Weather and Fire Behaviour.' (CSIRO Australia: Melbourne.)

Cheney, N. P., Gould, J. S. and Catchpole, W. R. (1998). Prediction of fire spread in grasslands. *International Journal of Wildland Fire* **8**, 1–13.

Cheney, N. P., Gould, J. S. and Knight, I. (1992). A Prescribed Burning Guide for Young Regrowth Forests of Silvertop Ash. Research Paper 16, Forestry Commission of New South Wales, Sydney.

Christensen, P. E. and Kimber, P. C. (1975). Effects of prescribed burning on the flora and fauna of south-west Australian forests. In 'Proceedings of the Ecological Society of Australia.' **9**, 85–106.

Claridge, A. W. (1998). Use of tracks and trails by introduced predators: an important consideration in the study of native ground-dwelling mammals. *The Victorian Naturalist* **115**, 88–93.

Clark, E. (1992). Alpine grazing, a tale of two states. *The Australian Alps*. *Revue de Geographie Alpine.'* (Eds P. Grenier and R. Good.) **80** (2–3), 129–155.

Clark, J. S. (1991). Disturbance and population structure on the shifting mosaic landscape. *Ecology* **72**, 1119–1137.

Clark, J. S. (1996). Testing disturbance theory with long-term data: alternative life-history solutions to the distribution of events. *The American Naturalist* **148**, 976–996.

Clark, J. S., Gill, A. M. and Kershaw, P. (2002). Spatial stability in fire regimes: its effects on recent and past vegetation. In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 125–141. (Cambridge University Press: U.K.)

Cochrane, M. A. and Laurance, W. F. (2002). Fire as a large-scale edge effect in Amazonian forests. *Journal of Tropical Ecology* **18**, 311–325.

Cogger, H., Cameron, E., Sadlier, R. and Eggler, P. (1993). 'The Action Plan for Australian Reptiles.' (Australian Nature Conservation Agency: Canberra.)

Corbett, L. K. (2004). Swamp buffalo. *Bubalus bubalis* (Linnaeus, 1758). In 'The Mammals of Australia.' (Ed. R. Strahan.) pp. 723–725. (Reed New Holland: Sydney.)

Corbett, L. K., Andersen, A. N. and Muller, W. J. (2003). Terrestrial vertebrates. *Ecological Studies* **169**, 126–152.

Cornish, P. (2001). The effects of roading, harvesting and forest regeneration on streamwater turbidity levels in a moist eucalypt forest. *Forest Ecology and Management* **152**, 293–312.

Costin, A. B. (1954). 'A Study of the Ecosystems of the Monaro Region of New South Wales with Special Reference to Soil Erosion.' (Soil Conservation Service of New South Wales: Sydney.)

Costin, A. B. (1981). Alpine and sub-alpine vegetation. In 'Australian Vegetation.' (Ed. R. H. Groves.) pp. 361–376. (Cambridge University Press: Cambridge, U.K.)

Costin, A. B. (1983). Mountain lands in the Australian region: some principles of use and management. *Proceedings of the Ecological Society of Australia* **12**, 1–13.

Coyne, P. (2001). 'Protecting the Natural Treasures of the Australian Alps.' (A Report to the Natural Heritage Working Group of the Australian Alps Liaison Committee. Australian Alps National Parks.)

Craig, A. B. (1999). Fire management of rangelands in the Kimberley low rainfall zone: a review. *Rangeland Journal* **21**, 39–70.

Croke, J. (2004). Forest harvesting activities and water quality: a new approach to an old problem. In '*Forest Management Workshop Proceedings*. *Canberra 23–25 March 2004*.' (Eds J. Croke, S. Mockler and I Takken.) (Cooperative Research Centre for Catchment Hydrology: Canberra.)

Crowley, G., Garnett, S. and Shephard, S. (2003). 'Management Guidelines for Golden-Shouldered Parrot Conservation.' (Queensland Parks and Wildlife Service: Brisbane.)

CSIRO (1997). 'CSIRO Grassland Fire Spread Meter.' (Styrox Australia: Warriewood, NSW.)

Cullen, P. J. and Kirkpatrick, J. B. (1988). The ecology of *Athrotaxis* D. Don. (Taxodiaceae). I Stand structure and regeration of *A. cupressoides*. *Australian Journal of Botany* **36**, 547–560.

Cunningham, G. M. (1981). Desert vegetation. In '*Australian Vegetation*.' (Ed. R. H. Groves.) pp. 377–389. (Cambridge University Press: Cambridge, U.K.)

Daly, R. L. and Hodgkinson, K. C. (1996). Relationships between grass, shrub and tree cover on four landforms in semi-arid eastern Australia, and prospects for change by burning. *Rangelands Journal* **18**, 104–117.

Davidson, S. (2005). Cultural burning revives a Kakadu wetland. *Ecos* **125**, 14–16.

De Ligt, R. (2005). Patterns in the Probability of Burning with Times-Since-Fire in the Greater Sydney Region. B.Sc. Thesis, The Australian National University, Canberra.

Dickinson, M. B. and Johnson, E. A. (2001). Fire effects on trees. In 'Forest Fires. Behavior and Ecological Effects.' (Eds E. A. Johnson and K. Miyanishi.) pp. 477–525. (Academic Press: San Diego.)

Dorges, B. and Heucke, J. (2004). One-humped camel. *Camelus dromadarius*. In 'The Mammals of Australia.' (Ed. R. Strahan.) pp. 719–721. (Reed New Holland: Sydney.)

Dorrough, J. and Moxham, C. (2005). Eucalypt establishment in agricultural landscapes and implications for landscape-scale restoration. *Biological Conservation* **123**, 55–66.

Dorrough, J., Yen, A., Turner, V., Clark, S. G., Crosthwaite, J. and Hirth, J. R. (2004). Livestock grazing management and biodiversity conservation in Australian temperate grassy landscapes. *Australian Journal of Agricultural Research* **55**, 279–295.

Driessen, M. M., Hocking, G. J. and Beukers, P. (1990). 'Habitat, Conservation Status and Management of the Tasmanian Bettong *Bettongia gaimardi*.' Tasmanian Department of Parks, Wildlife and Heritage, Hobart.

Eldridge, D. J. and Rath, D. (2002). Hip holes: kangaroo (*Macropus* spp.) resting sites modify the physical and chemical environment of woodland soils. *Austral Ecology* **27**, 527–536.

Eldridge, D. J. and Bradstock, R. A. (1994). The effect of time since fire on the cover and composition of cryptogamic soil crusts on a eucalypt shrubland soil. *Cunninghamia* **3**, 521–527.

Ellis, S, Kanowski, P. and Whelan, R. (2004). 'Council of Australian Governments National Inquiry on Bushfire Mitigation and Management.' (Commonwealth of Australia: Canberra.)

Enright, N. J., Lamont, B. B. and Miller, B. P. (2005). Anomalies in grasstree fire history reconstructions for south-western Australian vegetation. *Austral Ecology* **30**, 668–673.

Esplin, B., Gill, A. M. and Enright, N. (2003). Report of the Inquiry into the 2002–2003 Victorian Bushfires. State Government of Victoria, Melbourne.

Ettringham, S. K. (1979). 'The Ecology and Conservation of Large African Animals.' (Macmillan Press: London.)

Evans, M. C. and Jarman, P. J. (1999). Diets and feeding selectivities of bridled nailtail wallabies and black-striped wallabies. *Wildlife Research* **26**, 1–19.

Everist, S. L. (1981). 'Poisonous plants of Australia.' (Angus and Robertson: London.)

Fehmi, J. S., Karn, J. S., Ries, R. E., Hendricksen, J. R. and Hanson, J. D. (2002). Cattle grazing behaviour with season long free-choice access to four forage types. *Applied Animal Behaviour Science* **78**, 29–42.

Fensham, R. J. (1998). The grassy vegetation of the Darling Downs, south-eastern Queensland, Australia. Floristics and grazing effects. *Biological Conservation* **84**, 301–310.

Fernandes, P. M. and Botelho, H. S. (2003). A review of prescribed burning effectiveness in fire hazard reduction. *International Journal of Wildland Fire* **12**, 117–128.

Finlayson, H. H. (1945). 'The Red Centre: Man and Beast in the Heart of Australia.' (Angus and Robertson: Sydney.)

Finney, M. A. (1995). The missing tail and other considerations for the use of fire history models. *International Journal of Wildland Fire* **4**, 197–202.

Fire Ecology Working Group (2004). Guidelines and Procedures for Ecological Burning on Public Land in Victoria 2004. Victorian Department of Sustainability and Environment, Melbourne.

Fowells, H. A. (1965). 'Silvics of Forest Trees of the United States.' (USDA Forest Service: Washington, D.C.)

Fraser, F., Lawson, V., Morrison, S., Christophersen, P., McGregor, S. and Rawlinson, M. (2003). Fire management experiment for the declining Partridge Pigeon, Kakadu National Park. *Ecological Management and Restoration* **4**, 94–102.

Friedel, M. H. and James, C. (1995). How does grazing of native pastures affect their biodiversity? In 'Conserving Biodiversity: Threats and Solutions.' (Eds R. A. Bradstock, T. D. Auld, D. A. Keith, R. Kingsford, D. Lunney and D. Sivertsen.) pp. 249–259. (Surrey Beatty and Sons: Sydney.)

Friend, G. and Wayne, A. (2003). Relationships between mammals and fire in south-west Western Australian ecosystems: what we know and what we need to know. In 'Fire in Ecosystems of South-west Western Australia.' (Eds I. Abbott and N. Burrows.) pp. 363–380. (Backhuys Publishers: Leiden, The Netherlands.)

Frost, P. G. H. (1984). The responses and survival of organisms in fire-prone environments. *Ecological Studies* **48**, 273–309.

Garkaklis, M. J., Caalver, M. C., Wilson, B. A. and St.J. Hardy, G. E. (2004). Habitat alteration caused by an introduced plant disease, *Phytophthora cinnamomi*: a potential threat to the conservation of Australian forest fauna. In 'Conservation of Australia's Forest Fauna.' 2nd edn. (Ed. D. Lunney.) pp. 899–913. (Royal Zoological Society of New South Wales: Sydney.)

Garnett, S. (1992). Threatened and Extinct Birds of Australia. Royal Australian Ornithologists Union Report No. 82. 212 pp.

Garnett, S.T. and Crowley, G. M. (2000). 'The Action Plan for Australian Birds.' (Environment Australia: Canberra.)

Gelbard, J. L. and Belnap, J. (2003). Roads as conduits for exotic plant invasions in a semiarid landscape. *Conservation Biology* **17**, 420–432.

Gill, A. M. (1975). Fire and the Australian flora: a review. *Australian Forestry* **38**, 4–25.

Gill, A. M. (1977). Management of fire-prone vegetation for plant species conservation in Australia. *Search* **8**, 20–26.

Gill, A. M. (1981). Adaptive responses of Australian vascular plant species to fires. In 'Fire and the Australian Biota.' (Eds A. M. Gill, R. H. Groves and I. R. Noble.) pp. 243–272. (Australian Academy of Science: Canberra.)

Gill, A. M. (1986). Research for the Fire Management of Western Australian State Forests and Conservation Reserves. Western Australian Department of Conservation and Land Management, Technical Report No. 12. 82 pp.

Gill, A. M. (1995). Stems and fires. In 'Plant Stems. Physiology and Functional Morphology.' (Ed. B. Gartner.) pp. 323–342. (Academic Press: San Diego.)

Gill, A. M. (1996). How fires affect biodiversity. In 'Biodiversity and Fire – the Effects and Effectiveness of Fire Management.' pp. 47–55. (Department of Environment, Sports and Territories: Canberra.)

Gill, A. M. (1998). An hierarchy of fire effects: impact of fire regimes on landscapes. In '3rd International Conference on Forest Fire Research and 14th Conference on Fire and Forest Meteorology Proceedings, Luso, Portugal, November 1998. Volume 1'. pp. 129–144.

Gill, A. M. (1999*a*). Bushfires and biodiversity: an Australia-wide perspective on plant-species changes after a fire event. *Environment Australia Biodiversity Technical Paper* No. 1. pp. 9–53. Available at www.deh.gov.au/biodiversity/publications/technical/fire/index.html [Verified April 2008].

Gill, A. M. (1999*b*). Facets of fire ecology, biodiversity conservation and landscape patch dynamics. In 'Management of Fire for the Conservation of Biodiversity. Workshop Proceedings.' (Eds G. Friend, M. Leonard, A. Maclean, I. Siegler.) pp. 18–21. (Victorian Department of Natural Resources and Environment and Parks Victoria: Melbourne.)

Gill, A. M. (2000). Fire-pulses in the heart of Australia: fire regimes and fire management in Central Australia. Report to Environment Australia, August 2000.

Gill, A. M. (2003). The wheel of management: biodiversity conservation as restoration. In 'Bushfires: Managing the Risk 2002.' (Eds A. Baker, B. Diekman and M. Sparks.) pp. 119–123. (Nature Conservation Council of New South Wales: Sydney.)

Gill, A. M. (2005). Landscape fires as social disasters: an overview of 'the bushfire problem' *Global Climate Change B*. *Environmental Hazards* **6**, 65–80.

Gill, A. M. (2006). Dating Fires from Balga Stems: The Controversy over Fire Histories Determined from south-western Australian *Xanthorrhoea*. Report to Western Australian Department of Environment and Conservation, Perth. Available at http://www.naturebase.net/content/view/2531/482 [Verified April 2008].

Gill, A. M. and Allan, G. (in press). Large fires, fire effects and the concept of the fire regime. *International Journal of Wildland Fire*

Gill, A. M. and Bradstock, R. A. (1992). A national register for the fire responses of plant species. *Cunninghamia* **2**, 653–660.

Gill, A. M. and Bradstock, R. A. (1995*a*). Extinctions of biota by fires. In 'Conserving Biodiversity: Threats and Solutions.' (Eds R. A. Bradstock, T. D. Auld, D. A. Keith, R. Kingsford, D. Lunney and D. Sivertsen.), pp. 309–322. (Surrey Beatty and Sons: Sydney.)

Gill, A. M. and Bradstock, R. A. (1995*b*). Mosaic burning: paradigm or dogma? In 'Bushfire '95 Presented Papers.' [Abstract only]. (Forestry Tasmania: Hobart.)

Gill, A. M. and Bradstock, R. A. (1998). Prescribed burning: Patterns and Strategies. Proceedings of the 13th International Conference on Fire and Forest Meteorology, Lorne, Australia, October 1996. pp. 3–6. (International Association of Wildland Fire: Moran, Wyoming, U.S.A.)

Gill, A. M. and Catling, P. C. (2002). Fire regimes and biodiversity of forested landscapes of southern Australia. In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 351–369. (Cambridge University Press: Cambridge, U.K.)

Gill, A. M. and McCarthy, M. C. (1998) Intervals between prescribed fires in Australia: what intrinsic variation should apply? *Biological Conservation* **85**, 161–169.

Gill, A. M. and McMahon, A. (1986). A post-fire chronosequence of cone, follicle and seed production in *Banksia ornata*. *Australian Journal of Botany* **34**, 425–433.

Gill, A. M. and Moore, P. H. R. (1990). Fire intensities in eucalypt forests of south-eastern Australia. *International Conference on Forest Fire Research, Coimbra, Proceedings* **B**.24, pp. 1–12.

Gill, A. M. and Moore, P. H. R. (1997). Contemporary fire regimes in the forests of southwestern Australia. Contract Report, Environment Australia. 115 pp.

Gill, A. M. and Nicholls, A. O. (1989). Monitoring fire-prone flora in reserves for nature conservation. In 'Fire Management on Nature Conservation Lands. Western Australia, Department of Conservation and Land Management Occasional Paper1/89.' (Eds N. Burrows , L. McCaw and G. Friend.) pp. 137–151.

Gill, A. M. and Zylstra, P. (2005). Flammability of Australian forests. *Australian Forestry* **68**, 88–94.

Gill, A. M., Allan, G. and Yates, C. (2003). Fire-created patchiness in Australian savannas. *International Journal of Wildland Fire* **12**, 323–331.

Gill, A. M., Burrows, N. D. and Bradstock, R. A. (1995). Fire spread in hummock grasslands. *CALM Science Supplement* **4**, 29–34.

Gill, A. M., Pook, E. W. and Moore, P. H. R.(1987*a*). Measuring Fire Properties for Ecological Effects at Kakadu National Park. Unpublished Report to the Australian National Parks and Wildlife Service.

Gill, A. M., Christian, K. R., Moore, P. H. R. and Forrester, R. I. (1987*b*). Bushfire incidence, fire hazard and fuel reduction burning. *Australian Journal of Ecology* **12**, 299–306.

Gill, A. M., Ryan, P. G., Moore, P. H. R. and Gibson, M. (2000). Fire regimes of World Heritage Kakadu National Park, Australia. *Austral Ecology* **25**, 616–625.

Gill, A. M., Worthy, M., Wade, A., Mueller, N. and White, I. (2008). Fire and water: framing fire and fuel management in Canberra's catchments. In 'Proceedings of Water Down Under 2008, 14–17 April 2008'. Compact Disc. (Government of South Australia: Adelaide.)

Goldammer, J. G. and Furyaev, V. V. (1996). 'Fire in Ecosystems of Boreal Eurasia.' (Kluwer Academic Publishers: Dordrecht, The Netherlands.)

Good, R. B. (1996). Fuel dynamics, PREPLAN and future research needs. In 'Fire and Biodiversity. The Effects and Effectiveness of Fire Management. Proceedings of the Conference held 8–9 October 1994, Footscray, Melbourne' pp. 253–261. (Department of the Environment, Sport and Territories, Commonwealth of Australia: Canberra.)

Goosem, M. (2004). Linear infrastructure in the tropical rainforests of far north Queensland: mitigating impacts on fauna of roads and powerline clearings. In 'Conservation of Australia's Forest Fauna.' 2nd edn. (Ed. D. Lunney.) pp. 418–434. (Royal Zoological Society of New South Wales: Sydney.)

Gould, J. S. (2004). Fire management – an integral part of forest management. In 'Forest Management Workshop Proceedings. Canberra 23–25 March 2004.' (Eds J. Croke, S. Mockler and I. Takken.) (CRC for Catchment Hydrology: Canberra.)

Gould, J. S., McCaw, W. L., Cheney, N. P., Ellis, P. F., Knight, I. K. and Sullivan, A. L. (2007). 'Project Vesta – Fire in Dry Eucalypt Forest: Fuel Structure, Fuel Dynamics and Fire Behaviour.' (Ensis-CSIRO: Canberra, ACT. Department of Environment and Conservation: Perth WA.)

Grant, S. R. and Wouters, M. A. (1993). The effect of fuel reduction burning on the suppression of four wildfires in Western Victoria. *Victorian Department of Conservation and Natural Resources Research Report* No. 41.

Greene, D. F. and Johnson, E. A. (2000). Tree recruitment from burn edges. *Canadian Journal of Forest Research* **30**, 1264–1274.

Griffin, G. F. (1984). Climate, vegetation, man and fire. In 'Anticipating the Inevitable: A Patch-burn Strategy for Fire Management at Uluru (Ayers Rock-Mt Olga) National Park.' (Ed. E. C. Saxon.) pp. 7–12. (CSIRO Australia: Melbourne.)

Griffin, G. F., Price, N. F. and Portlock, H. F. (1983). Wildfires in the central Australian rangelands, 1970–1980. *Journal of Environmental Management* **17**, 311–323.

Grgurinovic, C. (2000). Fungi subprogram. *Biologue* **23**, 4–5.

Groneveld, J., Enright, N. J., Lamont, B. B. and Wissel, C. (2002). A spatial model of co-existence among three *Banksia* species along a topographic gradient in fire-prone shrublands. *Journal of Ecology* **90**, 762–774.

Groves, R. H. and Williams, O. B. (1981). Natural grasslands. In 'Australian Vegetation.' (Ed. R. H. Groves.) pp. 293–316. (Cambridge University Press: Cambridge, U.K.)

Groves, R. H., Hosking, J. R., Batianoff, G. N., Cooke, D. A., Cowie, I. D., Johnson, R. W., Kieghery, G. J., Lepschi, B. J., Mitcell, A. A., Moerkerk, M., Randall, R. P., Rozefelds, A. C., Walsh, N. G. and Waterhouse, B. M. (2003). 'Weed categories for natural and agricultural ecosystem management.' (Bureau of Rural Sciences: Canberra.)

Hammill, K. A., Bradstock, R. A. and Alloway, W. G. (1998). Post-fire seed dispersal and species re-establishment. *Australian Journal of Botany* **46**, 407–419.

Hanski, I. and Gilpin, M. E. (1997). 'Metapopulation Biology: Ecology, Genetics, and Evolution.' (Academic Press: San Diego.)

He, T., Krauss, S. L., Lamont, B. B., Miller, B. P. and Enright, N. J. (2004). Long-distance seed dispersal in a metapopulation of *Banksia hookeriana* inferred from a population allocation analysis of amplified fragment length polymorphism data. *Molecular Ecology* **13**, 1099–1109.

Heddle, E. M. and Specht, R. L. (1975). Dark Island Heath (Ninety-Mile Plain, South Australia). VIII. The effect of fertilizers on composition and growth 1950–1972. *Australian Journal of Botany* **23**, 151–164.

Heemstra S. (2006). Bush fire patchiness. In 'Life in a Fire-Prone Environment: Translating Science into Practice – Proceedings of the Bushfire 2006 Conference, Brisbane, June 6–9, 2006. Paper No. 55.' (Ed. C. Tran.)

Henderson, M. K. and Keith, D. A. (2002). Correlation of burning and grazing indicators with composition of woody understorey flora of dells in a temperate forest. *Austral Ecology* **27**, 121–131.

Helms, R. (1893). Report on the grazing leases of the Kosciusko Plateau. *Agricultural Gazette of NSW* **4**, 530–531.

Hennessy, K., Lucas, C., Nicholls, N., Bathols, J., Suppiah, R. and Ricketts, J. (2005). 'Climate Change Impacts on Fire-weather in South-east Australia.' (CSIRO: Australia.)

Henzell, R. (2004). Goat. *Capra hircus*. In 'The Mammals of Australia.' (Ed. R. Strahan.) pp. 728–730. (Reed New Holland: Sydney.)

Hobbs, R. J. (2001). Synergisms among habitat fragmentation, livestock grazing, and biotic invasions in south-western Australia. *Conservation Biology* **15**, 1522–1528.

Hodgkinson, K. C. (2002). Fire regimes in *Acacia* wooded landscapes: effects on functional processes and biological diversity. In 'Flammable Australia: The Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 259–277. (Cambridge University Press: Cambridge, U.K.)

Holling, C. S. (1978). (Ed.) 'Adaptive Environmental Assessment and Management.' (Wiley and Sons: London.)

Hopkins, A. J. M. (1987). Management theory and optimum feedback. In 'Nature Conservation and the Role of Remnants of Native Vegetation.' (Eds D. A. Saunders, G. W. Arnold, A. A. Burbidge and A. J. M. Hopkins.) pp. 385–386. (Surrey Beatty and Sons: Sydney.)

Horton, D. H. (1982). The burning question: Aborigines, fires and Australian ecosystems. *Mankind* 13, 237–251.

Jackson, W. D. (1968). Fire, air, water and earth – an elemental ecology of Tasmania. *Proceedings of the Ecological Society of Australia* **3**, 9–16.

Janssen, M. A., Anderies, J. M. and Walker, B. H. (2004). Robust strategies for managing rangelands with multiple stable attractors. *Journal of Environmental Economics and Management* **47**, 140–162.

Johnson, C. (2006). 'Australia's mammal extinctions. A 50 000 year history. (Cambridge University Press: UK.)

Johnson, E. A. and Gutsell, S. L. (1994). Fire frequency models, methods and interpretations. *Advances in Ecological Research* **25**, 239–287.

Johnson, E. A. and Miyanishi, K. (2001) (Eds). 'Forest Fires: Behavior and Ecological Effects.' (Academic Press: San Diego.)

Jolaosho, A. O., Olanite, J. A., Onifade, O. S. and Oke, A. O. (2006). Seed in the faeces of ruminant animals grazing native pastures under semi-intensive management in Nigeria. *Tropical Grasslands* **40**, 79–83.

Keeley, J. E. and Fotheringham, C. J. (2001). Historic fire regimes in southern California shrublands. *Conservation Biology* **15**, 1536–1548.

Keetch, J. J. and Byram, G. M. (1968). A drought index for forest fire control. USDA Forest Service Research Paper SE-38.

Keith, D. (1996). Fire-driven extinction of plant populations: a synthesis of theory and review of evidence from Australian vegetation. *Proceedings of the Linnean Society of New South Wales* **116**, 37–78.

Keith, D. A. and Bradstock, R. A. (1994). Fire and competition in Australian heath: a conceptual model and field investigations. *Journal of Vegetation Science* **5**, 347–354.

Keith, D. A., McCaw, W. L. and Whelan, R. J. (2002). Fire regimes in Australian heathlands and their effects on plants and animals. In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 199–237. (Cambridge University Press: U.K.)

Kemp, D. R., King, W. M., Gilmour, A. R., Lodge, G. M., Murphy, S. R., Quigley, P. E., Sanford, P. and Andrew, M. H. (2003). SGS Biodiversity Theme: Impact of plant biodiversity on the productivity and stability of grazing systems across southern Australia. *Australian Journal of Experimental Agriculture* **43**, 961–975.

Kenny, B., Sutherland, E., Tasker, E. and Bradstock, R. (2004). 'Guidelines for Ecologically Sustainable Fire Measurement. NSW Biodiversity Strategy.' (New South Wales Government: Sydney.)

Kershaw, P., Clark, J. S., Gill, A. M. and D'Costa, D. M. (2002). A history of fire in Australia. In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 3–25. (Cambridge University Press: U.K.)

Kimber, R. (1983). Black lightning: Aborigines and fire in Central Australia and the Western Desert. *Archaeology in Oceania* **18**, 38–45.

King, K. J. (2004). Simulating the Effects of Anthropogenic Burning on Patterns of Biodiversity. PhD Thesis, The Australian National University, Canberra.

King, K. J., Cary, G. J., Bradstock, R. A., Chapman, J., Pyrke, A. and Marsden-Smedley, J. B. (2006). Simulation of prescribed burning strategies in south-west Tasmania, Australia: effects on unplanned fires, fire regimes, and ecological management values. *International Journal of Wildland Fire* **15**, 527–540.

Kirkpatrick, J. B., Gilfedder, L., Bridle, K. and Zacharek, A. (2005). The positive and negative conservation impacts of sheep grazing and other disturbances on the vascular plant species and vegetation of lowland subhumid Tasmania. *Ecological Management and Restoration* **6**, 51–60.

Knight, T. M. and Holt, R. D. (2005). Fire generates spatial gradients in herbivory: an example from a Florida sandhill ecosystem. *Ecology* **86**, 587–593.

Lamont, B. B., Swanborough, P. W. and Ward, D. (2000). Plant size and season of burn affect flowering and fruiting of the grasstree *Xanthorrhoea preissii. Austral Ecology* **25**, 168–272.

Lamont, B. B., Ward, D. J., Eldridge, J., Korczynskyj, D., Colangelo, W. L., Fordham, C., Clements, E. and Wittkuhn, R. (2003). Believing the Balga: a new method for gauging the fire history of vegetation using grasstrees. In 'Fire in Ecosystems of South-west Western Australia.' (Eds I. Abbott and N. Burrows.) pp. 147–169. (Backhuys Publishers: Leiden, The Netherlands.)

Landsberg, J., James, C. D., Morton, S. R., Hobbs, T. J., Stol, J., Drew, A. and Tongway, H. (1997).). The Effects of Artificial Sources of Water on Rangeland Biodiversity. Final Report to the Biodiversity Convention and Strategy Section of the Biodiversity Group, Environment Australia. CSIRO Wildlife and Ecology, Canberra.

Landsberg, J., James, C. D., Maconochie, J., Nicholls, A. O., T. J., Stol, J. and Tynan, R. (2002). Scale-related effects of grazing on native plant communities in an arid rangeland region of South Australia. *Journal of Applied Ecology* **39**, 427–444.

Lang, S. and Gill, A. M. (1997). Appendix 3. The use of micromaps for determining fire regimes. Summary of a thesis and some additional information. In 'Contemporary Fire Regimes in the Forests of Southwestern Australia.' (Eds A. M. Gill and P. H. R. Moore.) Contract Report, Environment Australia. 115 pp.

Langevelde, F. van, van der Vijver, C. A. D. M., Kumar, L., van der Koppel, J., de Ridder, N., van Andel, J., Skidmore, A. K., Hearne, J. W., Stroosnijder, L., Bond, W. J., Prins, H. H. T. and Rietkerk, M. (2003). Effects of fire and herbivory on the stability of savanna ecosystems. *Ecology* **84**, 337–350.

Leaver, B. (1994). Fire values. In 'An Assessment of the Values of Kosciuszko National Park.' (Eds. Kosciuszko Independent Scientific Committee.) pp. 117–128. (NSW National Parks and Wildlife Service: Sydney.)

Lee, A. K. (1995). 'The Action Plan for Australian Rodents.' (Australian Nature Conservation Agency: Canberra.)

Leigh, J. H. (1994). Chenopod shrub lands. In 'Australian Vegetation.' 2nd edn. (Ed. R. H. Groves.) pp. 345–367. (Cambridge University Press: Cambridge, U.K.)

Leigh, J. H. and Holgate, M. D. (1978). Effects of pasture availability on the composition and quality of the diet delected by sheep grazing native, degenerate and improved pasture in the Upper Shoalhaven Valley, New South Wales. *Australian Journal of Experimental Agriculture and Animal Husbandry* **18**, 381–390.

Leigh, J. H., Boden, R. and Briggs (1984). 'Extinct and Endangered Plants of Australia.' (Macmillan Company: South Melbourne.)

Leigh, J. H., Wood, D. H., Holgate, M. D., Slee, A. and Stanger, M. G. (1989). Effects of rabbit and kangaroo grazing on two semi-arid grassland communities in central-western New South Wales. *Australian Journal of Botany* **37**, 375–396.

Leigh, J. H., Wimbush, D. J., Wood, D. H., Holgate, M. D., Slee, A. V., Stanger, M. G. and Forrester, R. I. (1987). Effects of rabbit grazing and fire on a subalpine environment. I. Herbaceous and shrubby vegetation. *Australian Journal of Botany* **35**, 433–464.

Le-Ray-Meyer, N. (2003). A Report on the Aerial Fire Fighting Operations. In 'Report of the Inquiry into the 2002–2003 Victorian Bushfires.' (Eds B. Esplin, A. M. Gill and N. Enright.) (State Government of Victoria: Melbourne.)

Letnic, M. (2003). The effects of experimental patch burning and rainfall on small mammals in the Simpson Desert, Queensland. *Wildlife Research* **30**, 547–563.

Lindenmayer D. B. (1996). 'Wildlife and Woodchips. Leadbeater's Possum, A Test Case for Sustainable Forestry.' University of New South Wales Press, Sydney.

Lindenmayer D. B. and Possingham H. P. (1996). Modelling the inter-relationships between habitat patchiness, dispersal capability and metapopulation persistence of the endangered species, Leadbeater's possum, in south-eastern Australia. *Landscape Ecology* **11** (2): 79–105

Loane, I. T. and Gould, J. S. (1986). 'Aerial Suppression of Bushfires. Cost-benefit Study for Victoria.' (CSIRO Forest Research: Canberra.)

Lucas, C., Hennessy, K., Mills, G. and Bathols, J. (2007). 'Bushfire weather in southeast Australia: Recent trends and projected Climate Change Impacts'. Consultancy Report for The Climate Institute of Australia. CSIRO and the Bushfire Cooperative Research Centre, Melbourne.

Luke, R. H. and McArthur, A. G. (1978). 'Bushfires in Australia.' (Australian Government Publishing Service: Canberra.)

Lunney, D. (2001). Causes of the extinction of native mammals of the Western Division of New South Wales: an ecological interpretation of the nineteenth century historical record. *Rangeland Journal* **23**, 44–70.

Lunt, I. D. (2005). 'Effects of stock grazing on biodiversity values in temperate native grasslands and grassy woodlands in SE Australia: a literature review.' (Environment ACT, Canberra.) Available at http://www.tams.act.gov.au/live/environment/native_plants_and_animals/report_series [Verified April 2008].

Lunt, I. D. and Morgan, J. W. (2002). The role of fire regimes in temperate lowland grasslands of southeastern Australia. In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 177–196. (Cambridge University Press: U.K.)

Lunt, I. D., Eldridge, D. J., Morgan, J. W. and Witt, G. B. (2007). Turner Review No. 13. A framework to predict the effects of livestock grazing and grazing exclusion on conservation values in natural ecosystems in Australia. *Australian Journal of Botany* **55** (4) in press.

McArthur, A. G. (1967). Fire behaviour in eucalypt forests. Commonwealth of Australia Forestry and Timber Bureau Leaflet 107.

McCarthy, G. J. and Tolhurst, K. G. (1998). Effectiveness of firefighting first attack operations by the Department of Natural Resources and Environment from 1991/92–1994/95. Victorian Department of Natural Resources and Environment Research Report No. 45.

McCarthy, G. J. and Tolhurst, K. G. (2001). Effectiveness of broadscale fuel reduction burning in assisting with wildfire control in Parks and Forests in Victoria. Victorian Department of Natural Resources and Environment Research Report No. 51.

McCarthy, G. J., Tolhurst, K. G. and Chatto, K. (1999*a*). Overall Fuel Hazard Guide. Victorian Department of Natural Resources and Environment, Fire Management Research Report No. 47.

McCarthy, G. J., Wouters, M. and Tolhurst, K. G. (2003). Park and Forest Firefighting Resources Guide. Victorian Department of Sustainability and Environment, Melbourne.

McCarthy, M. A., Gill, A. M. and Bradstock, R. A. (2001*a*). Theoretical fire interval distributions. *International Journal of Wildland Fire* **10**, 73–77.

McCarthy, M. A., Gill, A. M. and Lindenmayer, D. B. (1999*b*). Fire regimes in mountain ash forest: evidence from forest age structure, extinction models and wildlife habitat. *Forest Ecology and Management* **124**, 193–203.

McCarthy, M. A., Possingham, H. P. and Gill, A. M. (2001*b*). Using stochastic dynamic programming to determine optimal fire management of *Banksia ornata. Journal of Applied Ecology* **38**, 585–592.

McCarthy, P. (2000). Lichen and bryophytes subprograms. *Biologue* **23**, 5.

McCaw, W. L. (1991). Measurement of fuel quantity and structure for bushfire research and management. In 'Proceedings of the Conference on Bushfire Modelling and Fire Danger Rating Systems, 11–12 July 1988.' (Eds N. P. Cheney and A. M. Gill.) pp. 147–155. (CSIRO Division of Forestry: Yarralumla, ACT.)

McCaw, W. L. (1997). Predicting Fire Spread in Western Australian Mallee–heath Shrubland. Ph.D. Thesis, University of New South Wales, Canberra.

McCaw, W. L., Maher, T. and Gillen, K. (1992). Wildfires in the Fitzgerald River National Park, Western Australia, December 1989. W. A. Department of Conservation and Land Management Technical Report 26.

McDonald, J. M. (1999). 'Gippsland Fire Protection Plan.' (Victorian Department of Natural Resources and Environment: Melbourne.)

McGregor, B. A. (2002). Comparative productivity and grazing behaviour of Huacaya alpacas and Peppin Merino sheep grazed on annual pastures. *Small Ruminant Research* **44**, 219–232.

Mackey, B., Lindenmayer, D. B., Gill, A. M., McCarthy, M. A. and Lindesay, J. (2002). 'Wildlife, Fire and Future Climate.' (CSIRO Publishing: Melbourne.)

McLeod, R. (2003). 'Inquiry into the Operational Response to the January 2003 Bushfires in the ACT.' (ACT Government: Canberra.)

Marsden-Smedley (1993). 'Fuel Characteristics and Fire Behaviour in Tasmanian Buttongrass Moorlands.' (Parks and Wildlife Service: Tasmania.)

Marsden-Smedley, J. B. and Catchpole, W. R. (1995). Fire behaviour modelling in Tasmanian buttongrass moorlands. II. Fire behaviour. *International Journal of Wildland Fire* **5**, 215–228.

Martin, R. E. and Sapsis, D. B. (1991). Fires as agents of biodiversity: pyrodiversity promotes biodiversity. In 'Proceedings of the Symposium on Biodiversity of Northwestern California, October 28–30, 1991, Santa Rosa, California.' pp. 150–157.

Maxwell, S., Burbidge, A. A. and Morris, K. (Eds) (1996). 'The 1996 Action Plan for Australian Marsupials and Monotremes.' (Environment Australia: Canberra.)

Mercer, G. N., Gill, A. M. and Weber, R. O. (1995). A flexible, non-deterministic, litter accumulation model. In 'Bushfire '95, Presented Papers of the Australian Bushfire Conference 27–30 September 1995, Hobart, Tasmania.' Forestry Tasmania, Tasmanian Parks and Wildlife Service, Tasmanian Fire Service. 10 pp.

Miller, G. H., Magee, J. W., Johnson, B. J., Fogel, M. L., Spooner, N. A., McCulloch, M. T. and Aycliffe, L. K. (1999). Pleistocene extinction of *Genyornis newtoni*: human impact on Australian megafauna. *Science* **283**, 205–208.

Minnich, R. A. and Chou, Y. H. (1997). Wildland fire patch dynamics in the chaparral of southern California and northern Baja California. *International Journal of Wildland Fire* **7**, 221–248.

Miyanishi, K. (2001). Duff consumption. In 'Forest Fires. Behavior and Ecological Effects.' (Eds E. A. Johnson and K. Miyanishi.) pp. 437–475. (Academic Press: San Diego.)

Mohr, F. (1989). Light-hand suppression tactics – a fire-management challenge. *United States Forest Service Fire Management Notes* **50**(1), 21–23.

Mohr, F. (1992–93). Wildfire suppressed and the wilderness still looks natural! *Fire Management Notes USFS* **53–54**(3), 3–6.

Mohr, F. (1994). M.I.S.T. – putting the concept into practice. *Wildfire December* **1994**, 4–15.

Mohr, F. and Curtiss, K. (1998). U.S. Army firefighters practice "No Trace Camping" on wilderness wildfires. *Fire Management Notes USFS* **58**(1), 4–8.

Moore, R. M. (1962). Effects of the sheep industry on Australian vegetation. In 'The Simple Fleece. Studies in the Australian Wool Industry.' (Ed. A. Barnard.) pp. 170–183. (Melbourne University Press: Melbourne.)

Moore, P. F. and Shields, B. (1996). The basis of fuel management on State Forest in NSW. *Proceedings of the Linnean Society of New South Wales* **116**, 115–126.

Morgan, J. W. (2004). Bryophyte composition in a native grassland community subjected to different long-term fire regimes. *Cunninghamia* **8**, 485–489.

Moritz, M. (2003). Spatiotemporal analysis of controls on shrubland fire regimes: age dependency and fire hazard. *Ecology* **84**, 351–361.

Moritz, M. A., Keeley, J. E., Johnson, E. A. and Schaffner, A. A. (2004). Testing a basic assumption of shrubland fire management: how important is fuel age? *Frontiers in Ecology and Environment* **2**(2), 67–72.

Morrison, D. A., Cary, G. J., Pengelly, S. M., Ross, D. G., Mullins, B. J., Thomas, C. R. and Anderson, T. S. (1995). Effects of fire frequency on plant species composition of sandstone communities in the Sydney region: inter-fire interval and time-since-fire. *Australian Journal of Ecology* **20**, 239–247.

Mount, A. B. (1969). Eucalypt ecology as related to fire. *Proceedings of the Annual Tall Timbers Fire Ecology Conference* **9**, 75–108.

Muṯitjulu Community and Baker (1996). 'Minkiri. A Natural History of Uluru by the Muṯitjulu Community.' (IAD Press: Alice Springs.)

Nairn, G. (2003). 'A Nation Charred: Inquiry into the Recent Australian Bushfires.' (Commonwealth of Australia: Canberra.)

Nelson, R. M. (1980). Flame characteristics for fires in southern fuels. USDA Forest Service Research Paper SE 205.

Neumann, F. G. (1992). Effects of low intensity prescribed fire on invertebrates in mixed eucalypt forest – Wombat State Forest. In 'Ecological Impacts of Fuel Reduction Burning in Dry Sclerophyll Forest: First Progress Report.' Victorian Department of Conservation and Environment Research Report 349, 4.1–4.19.' (Eds K. Tolhurst and D. Flinn.)

Newsome, A. E. and Catling, P. C. (1979). Habitat preferences of mammals inhabiting heathlands of warm temperate coastal, montane and alpine regions of southeastern Australia. In 'Ecosystems of the World 9A. Heathland and Related Shrub lands, Descriptive Studies.' (Ed. R. L. Specht.) pp. 301–316. (Elsevier: Amsterdam.)

Noble, I. R. and Slatyer, R. O. (1980). The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbance. *Vegetatio* **43**, 5–21.

Noble, I. R. and Slatyer, R. O. (1981). Concepts and models of succession in vascular plant communities subject to recurrent fire. In 'Fire and the Australian Biota.' (Eds A. M. Gill, R. H. Groves and I. R. Noble.) pp. 311–335. (Australian Academy of Science: Canberra.)

Noble, I. R., Bary, G. A. V. and Gill, A. M. (1980). McArthur's fire-danger meters expressed as equations. *Australian Journal of Ecology* **5**, 20l–203.

Noble, J. C. (1991). Behaviour of a very fast grassland wildfire on the Riverine Plain of southeastern Australia. *International Journal of Wildland Fire* **1**, 189–196.

Noble, J. C. (1997). 'The Delicate and Noxious Scrub.' (CSIRO Wildlife and Ecology: Canberra.)

Noble, J. C. and Grice, A. C. (2002). Fire regimes in semi-arid and tropical pastoral lands: managing biological diversity and ecosystem function. In 'Flammable Australia: The Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 373–400. (Cambridge University Press: Cambridge, U.K.)

NSW National Parks and Wildlife Service (2003). Fire Management Plan. Yathong, Nombinnie and Round Hill Nature Reserves.

O'Connell, A. M. (1987). Litter dynamics in Karri (*Eucalyptus diversicolor*) forests of south-western Australia. *Journal of Ecology* **75**, 781–796.

Olson, J. S. (1963). Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* **44**, 322–331.

Orchard, T. (2000). Vascular flora and algae subprograms. *Biologue* **23**, 5–6.

Osborne, W. (2001). 'Recovery Plan for the Southern Corroboree Frog (*Pseudophryne corroboree*).' (NSW National Parks and Wildlife Service: Sydney.)

Papachristou, T. G. (1997). Foraging behaviour of goats and sheep on Mediterranean kermes oak shrublands. *Small Ruminant Research* **24**, 85–93.

Parr, C. L. and Brockett, B. H. (1997). Patch-mosaic burning: a new paradigm for savanna fire management in protected areas? *Koedoe* **42**(2), 117–130.

Parsons, R. F. (1981). *Eucalyptus* scrubs and shrub lands. In 'Australian Vegetation.' (Ed. R. H. Groves.) pp. 227–252. (Cambridge University Press: Cambridge, U.K.)

Pausas, J. G., Bradstock, R. A., Keith, D. A., Keeley, J. E. and the GCTE Fire Network (2004). Plant functional traits in relation to fire in crown-fire ecosystems. *Ecology* **85**, 1085–1100.

Pearce, K., Holper, P., Hopkins, M., Bouma, W., Whetton, P., Hennessy, K. and Power, S. (Eds) (2007). 'Technical Report 2007. Climate Change in Australia'. CSIRO, Melbourne.

Pimental, D., Pimental, M. and Wilson, A. (2007). Plant, animal, and microbe invasive species in the United States and world. *Ecological Studies* **193**, 315–330.

Pitt, J. L. (1998). Grader Grass. *Agnote* No. F9, Northern Territory Government, Darwin.

Plotnick, R. E. and Gardner, R. H. (1993). Lattices and landscapes. *Lectures on Mathematics in the Life Sciences* **23**, 129–157.

Polakow, D. A. and Dunne, T. T. (1999). Modelling fire-return interval *T*: stochasticity and censoring in the two-parameter Weibull model. *Ecological Modelling* **121**, 79–102.

Price, O., Russell-Smith, J. and Edwards, A. (2003). Fine-scale patchiness of different fire intensities in sandstone heat vegetation in northern Australia. *International Journal of Wildland Fire* **12**, 227–236.

Price, O., Edwards, A., Connors, G., Woinarski, J., Ryan, G., Turner, A. and Russell-Smith, J. (2005). Fire heterogeneity in Kakadu National Park, 1980–2000. *Wildlife Research* **32**, 425–433.

Prober, S. M. and Thiele, K. R. (1995). Conservation of the grassy White Box woodlands: relative contributions of size and disturbance to floristic composition and diversity of remnants. *Australian Journal of Botany* **43**, 349–366.

Raison, R. J., Woods, P. V. and Khanna, P. K. (1983). Fuel dynamics in recurrently burnt eucalypt forests. In 'Proceedings of the Institute of Foresters of Australia 10th Triennial Conference'. pp. 59–64.

Recher, H. (1981). Death of an Australian myth: fire and its effects on wildlife. In 'Bushfires: Their Effect on Australian Life and Landscape.' (Ed. P. Stanbury.) pp. 39–47. (The Macleay Museum, The University of Sydney.)

Reh, W. and Seitz, A. (1990). The influence of land use on the genetic structure of populations of the common frog *Rana temporaria. Biological Conservation* **54**, 239–249.

Reseigh, J., Nalodny, C. and Reid, N. (2003). Effects of recent changes in grazing and fertiliser management on species richness of native ground-layer vegetation, Northern Tablelands, New South Wales. In 'Proceedings of the Conference on Rural Land Use Change. Victorian Department of Sustainability and Environment, Melbourne'. pp. 1–9.

Richards, R. M., Cary, G. J. and Bradstock, R. A. (2001). The sensitivity of snow gum to fire scarring in relation to Aboriginal landscape burning. In 'Bushfire 2001, Proceedings'. (Eds G. Pearce and L. Lester) pp. 285–292. (New Zealand Forest Research Institute, Rotorua, New Zealand.)

Roberts, R. G., Flannery, T. F., Aycliffe, L. K., Yoshida, H., Olley, J. M., Prideaux, G. J., Laslett, G. M., Baynes, A., Smith, M. A., Jones, R. and Smith, B. L. (2001). New ages for the last Australian megafauna: continent-wide extinction about 46,000 years ago. *Science* **292**, 1888–1892.

Robertson, A. I. (1997). Land-water linkages in floodplain river systems: the influence of domestic stock. In 'Frontiers in Ecology: Building the Links.' (Eds N. Klomp and I. Lunt.) pp. 207–218. (Elsevier: Oxford.)

Robinson, R. M. (2006). Varying fire regimes to promote fungal diversity across the landscape. *Australasian Plant Conservation* **14**(4), 16–17.

Robinson, R. M. and Bougher, N. L. (2003). The responses of fungi to fire in Jarrah (*Eucalyptus marginata*) and Karri (*Eucalyptus diversicolor*) forests of south-west Western Australia. In 'Fire in Ecosystems of south-west Western Australia: Impacts and Management.' (Eds I. Abbott and N. Burrows.) pp. 269–289. (Backhuys Publishers: Leiden.)

Robley, A. J., Short, J. and Bradley, S. (2001). Dietary overlap between the burrowing bettong (*Bettongia lesueur*) and the European rabbit (*Oryctolagus cuniculus*) in semi-arid coastal Western Australia. *Wildlife Research* **28**, 341–349.

Ross, G. (2000). Vertebrates subprogram. *Biologue* **23**, 6–7.

Rossiter, N. A., Setterfield, S. A., Douglas, M. M. and Hutley, L. B. (2003). Testing the grass-fire cycle: alien grass invasion in the tropical savannas of northern Australia. *Diversity and Distributions* **9**, 169–176.

Sandberg, D. V., Ottmar, R. D. and Cushon, G. H. (2001). Characterizing fuels in the 21st century. *International Journal of Wildland Fire* **10**, 381–387.

Sandberg, D. V., Ottmar, R. G., Peterson, J. L. and Core, J. (2002) (Eds). Wildland Fire in Ecosystems: Effects of Fire on Flora. USDA Forest Service General Technical Report RMRS-GTR-42-volume 5.

Sandell, P. (1995). Welcome. In 'Proceedings of the North West Ecological Management Workshop. pp. 1–3. (National Parks Service of Victoria: Melbourne.) G. J. Horner (Ed.)

Saxon, E. C. (1984) (Ed.). 'Anticipating the Inevitable: A Patch-burn Strategy for Fire Management at Uluru (Ayers Rock-Mt Olga) National Park.' (CSIRO Australia: Melbourne.)

Senge, P. (1990). 'The Fifth Discipline: the Art and Practice of the Learning Organization.' (Doubleday: New York.)

Short, J. and Turner, B. (1994). A test of the vegetation mosaic hypothesis: A hypothesis to explain the decline and extinction of Australian mammals. *Conservation Biology* **8**, 439–449.

Sneeuwjagt, R. J. and Peet, G. B. (1985). 'Forest Fire Behaviour Tables for Western Australia (3rd Edition).' Department of Conservation and Land Management, Perth.

Smith, J. K. (2000) (Ed.). Wildland Fire in Ecosystems: Effects of Fire on Fauna. USDA Forest Service General Technical Report RMRS-GTR-42-volume 1.

Specht, R. L. (1970). Vegetation. In 'The Australian Environment.' 4th Edn. (Ed. G. W. Leeper.) pp. 44–67. (Melbourne University Press: Melbourne.)

Stewart, I. (1997). 'The Magical Maze. Seeing the World through Mathematical Eyes.' (John Wiley and Sons: New York, N.Y.)

Stocker, G. C. and Sturz, J. (1966). The use of fire to establish Townsville lucerne in the Northern Territory. *Australian Journal of Experimental Agriculture and Animal Husbandry* **6**, 276–279.

Takken, I. and Croke, J. (2004). Road to stream connectivity in forestry catchments. In 'Forest Management Workshop Proceedings. Canberra 23–25 March 2004. CRC for Catchment Hydrology, Canberra.' (Eds J. Croke, S. Mockler and I Takken.)

Tasker, E. M. and Bradstock, R. A. (2006). Influence of cattle grazing practices on forest understorey structure in north-eastern New South Wales. *Austral Ecology* **31**, 490–502.

Taylor, B. D. and Goldingay, R. L. (2004). Wildlife road-kills on three major roads in north-eastern New South Wales. *Wildlife Research* **31**, 83–91.

Tolhurst, K. (1999). Towards the implementation of ecologically based fire regimes in the Grampians National Park. In 'Management of Fire for the Conservation of Biodiversity. Workshop Proceedings.' (Eds G. Friend, M. Leonard, A. Maclean and I. Siegler.) pp. 30–38. (Victorian Department of Natural Resources and Environment and Parks Victoria: Melbourne.)

Tolhurst, K. G. and Cheney, N. P. (1999). 'Synopsis of the Knowledge used in Prescribed Burning in Victoria.' (Victorian Department of Natural Resources and Environment: Melbourne.)

Tolhurst, K. and Flinn, D. (1992) (Eds). Ecological Impacts of Fuel Reduction Burning in Dry Sclerophyll Forest: First Progress Report. Victorian Department of Conservation and Environment Research Report 349.

Tolhurst, K. G. and Friend, G. R. (2005). An objective basis for ecological fire management in Victoria. In 'Variability in Fire Interval. How important is it and how do we manage it? Proceedings of a Workshop held 26 September 2003, Albany, Western Australia.' (Eds T. V. Walshe and J. E. Williams.) pp. 32–43. (University of Western Australia: Crawley.)

Tolhurst, K. J. and Howlett,K. A. (2003). House ignition likelihood index – an hazard assessment method for land managers in the wildland-urban interface. Conference Paper 89. In 'Proceedings of the 3rd International Wildland Fire Conference and Exhibition incorporating the 10th Annual Australasian Fire Authorities Council Conference, 3–6 October, Sydney.'

Tolhurst, K. G. and MacAulay, A. S. (2003). SAROS – A model for predicting the effect of mass spotting on rate of spread of forest fire. Conference Paper 88. In 'Proceedings of the 3rd International Wildland Fire Conference and Exhibition, incorporating the 10th Annual Australasian Fire Authorities Council Conference, 3–6 October 2003, Sydney.'

Tolhurst, K. J., Kellas, J. D. and Wilson, A. A. G. (1992*a*). Low intensity fire behaviour and fuel dynamics in dry sclerophyll forest, Wombat State Forest. In 'Ecological Impacts of Fuel Reduction Burning in Dry Sclerophyll Forest: First Progress Report. Victorian Department of Conservation and Environment Research Report 349.' (Eds K. Tolhurst and D. Flinn.) pp. 2.1–2.40.

Tolhurst, K. G. Flinn, D. W., Loyn, R. H., Wilson, A. A. G. and Foletta, I. (1992*b*). 'Ecological Effects of Fuel Reduction Burning in a Dry Sclerophyll Forest. A Summary of Principal Research Findings and their Management Implications.' (Victorian Department of Conservation and Environment: East Melbourne.)

Tozer, M. G. and Bradstock, R. A. (2002). Fire-mediated effects of overstorey on plant species diversity and abundance in an eastern Australian heath. *Plant Ecology* **164**, 213–223.

Tran, C. and Wild, C. (2000). 'A Review of Current Knowledge and Literature to Assist in Determining Ecologically Sustainable Fire Regimes for the Southeast Queensland Region.' (Griffith University and the Fire and Biodiversity Consortium: Brisbane.)

Trollope, W. S. W. (1996). Biomass burning in the savannas of southern Africa with particular reference to the Kruger National Park in South Africa. In 'Biomass burning and global change, Volume 1. Remote sensing, modeling and inventory development, and biomass burning in Africa.' (Ed. J. S. Levine.) pp. 260–269. (MIT Press: Cambridge, U.S.A.)

Trombulak, S. C. and Frissell, C. A. (2000). Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* **14**, 18–30.

Tscharke, M. (2001). Experiences managing Terrick Terrick National Park. In 'Our Valuable Native Grasslands, Better Pastures Naturally. Proceedings of the Second National Conference. Stipa Native Grasses Association, Gulgong NSW.'

Turcotte, D. L. (1999). Self-organized criticality. *Reports on Progress in Physics* **62**, 1377–1429.

Turner, J. S. (1962). Catchment erosion. In 'The Simple Fleece. Studies in the Australian Wool Industry.' (Ed. A. Barnard.) pp. 259–266. (Melbourne University Press: Melbourne, and The Australian National University: Canberra.)

Underwood, R. J., Sneeuwjagt, R. J. and Styles, H. G. (1985). The contribution of prescribed burning to forest fire control in Western Australia: Case studies. In 'Symposium on Fire Ecology and Management in Western Australian Ecosystems.' (Ed. J. R. Ford.) pp. 153–170. (Western Australian Institute of Technology: Perth.)

Vallentine, J. F. (2001). 'Grazing Management.' (Academic Press: San Diego.)

Van Wilgen, B. W., Biggs, H. C. and Potgieter, A. L. F. (1998). Fire management and research in the Kruger National Park, with suggestions on the detection of thresholds of potential concern. *Koedoe* **41**, 69–87.

Victorian Department of Sustainability and Environment (2006). 'Code of Practice for Fire Management on Public Land. Revision No 1.' (Victorian Government: Melbourne.)

Wace, N. M. (1977). Assessment of dispersal of plant species – the car-borne flora in Canberra. *Proceedings of the Ecological Society of Australia* **10**, 167–186.

Walker, J. (1981). Fuel dynamics in Australian vegetation. In 'Fire and the Australian Biota.' (Eds A. M. Gill, R. H. Groves and I. R. Noble.) pp. 101–127. (Australian Academy of Science: Canberra.)

Walker, J. and Penridge, L. K. (1987). 'FOL-PROF: a Fortran-77 package for the generation of foliage profiles. Part 1. User Manual. Technical Memorandum 87/9.' (CSIRO Division of Water Resources Research: Canberra.)

Walsh, D., Andrew, M. H., Sinclair, R. and Coleman, D. (2005). Evidence of palatability variation between the sex phenotypes of bladder saltbush from small-plot grazing trials. *The Rangeland Journal* **27**, 89–96.

Wareing, K. and Flinn, D. (2003). 'The Victorian Alpine Fires January–March 2003.' (Victorian Department of Sustainability and Environment: Melbourne.)

Wark, M. C. (1997). Regeneration of some forest and gully communities in the Angahook-Lorne State Park (North-eastern Otway Ranges) 1–10 years after the wildfire of February 1983. *Proceedings of the Royal Society of Victoria* **109**, 7–36.

Watson, J. A. L., Lendon, C. and Low, B. S. (1973). Termites in Mulga lands. *Tropical Grasslands* **7**, 121–126.

Wells, A. and Beesley, P. (2000). Invertebrates subprogram. *Biologue* **23**, 7

Whalley, R. D. B. and Lodge, G. M. (1986). The manipulation of species composition of pastures by grazing management. In 'Rangelands, a Resource under Siege. Proceedings of the 2nd International Rangeland Congress, Adelaide, Australia, 13–18 May 1984.' p. 252.

Whelan, R. J. (1995). 'The Ecology of Fire.' (Cambridge University Press: Cambridge, U.K.)

Whelan, R. J., Rodgerson, L., Dickman, C. R. and Sutherland, E. F. (2002). In 'Flammable Australia: Fire Regimes and Biodiversity of a Continent.' (Eds R. A. Bradstock, J. E. Williams and A. M. Gill.) pp. 125–141. (Cambridge University Press: Cambridge, U.K.)

Williams, R. J., Wahren, C-H., Bradstock, R. A. and Muller, W. J. (2006). Does alpine grazing reduce blazing? A landscape test of a widely-held hypothesis. *Austral Ecology* **31**, 925–936.

Williams, J. E., Whelan, R. J. and Gill, A. M. (1994). Fire and environmental heterogeneity in southern forest temperate ecosystems: implications for management. *Australian Journal of Botany* **42**, 125–137.

Wilson, A. A. G. (1988). Width of firebreak that is necessary to stop grass fires: some field experiments. *Canadian Journal of Forest Research* **18**, 682–687.

Wilson, A. A. G. (1992*a*). Assessing the Fire Hazard on Public Land in Victoria: Fire Management Needs, and Practical Research Objectives. Research Report 31, Fire Management Branch, Victorian Department of Conservation and Environment, Melbourne.

Wilson, A. A. G. (1992*b*). Eucalypt Bark Hazard Guide. Research Report 32, Fire Management Branch, Victorian Department of Conservation and Environment, Melbourne.

Wilson, A. A. G. (1993). Elevated Fuel Guide. Research Report 35. Fire Management Branch, Victorian Department of Conservation and Environment, Melbourne.

Wilson, A. D. and Harrington, G. N. (1984). Grazing ecology and animal production. In 'Management of Australia's Rangelands.' (Eds G. N. Harrington, A. D. Wilson and M. D. Young.) pp. 63–77. (CSIRO: Melbourne.)

Woinarski, J. C. Z. (1999). Fire and Australian birds. A review. Environment Australia Biodiversity Technical Paper No. 1. pp. 55–111. Available at www.deh.gov.au/biodiversity/publications/technical/fire/index.html [Verified April 2008].

Woinarski, J. C. Z. and Fisher, A. (2003). Conservation and maintenance of biodiversity in the ranglelands. *Rangelands Journal* **25**(2), 157–171.

Woinarski, J. C. Z, Williams, R. J., Price, O. and Rankmore, B. (2005). Landscapes without boundaries: wildlife and their environments in northern Australia. *Wildlife Research* **32**, 377–388.

Wong, N., Morgan, J. and Price, J. (2006). Conservation management options for native grasslands of northern Victoria. *Australasian Plant Conservation* **14**(4), 12–13.

Woods, D. (1996). 'NPWS fauna species-habitat matrix.' (NSW National Parks and Wildlife Services: Jindabyne.)

Wouters, M., Tolhurst, K. J., Troy, S. and Friend, G. (2002). 'Analysis of Disturbance by Fire on Public Land in Victoria.' (Victorian Department of Natural Resources and Environment: Melbourne.)

Wright, B. R. and Clarke, P. J. (2007). Fire regime (recency, interval and season) changes the composition of spinifex (*Triodia* spp.)-dominated desert dunes. *Australian Journal of Botany* **55**, 709–724.

York, A. (1999). Long-term effects of repeated prescribed burning on forest invertebrates: management implications for the conservation of biodiversity. Environment Australia Biodiversity Technical Paper No. 1. pp. 181–266. Available at www.deh.gov.au/biodiversity/publications/technical/fire/index.html [Verified April 2008].

List of reports in this series

1. 1977. A Study of the distribution of aerially applied fire retardant in softwood plantations. R. Rawson.
2. 1978. Low intensity prescribed burning in three *Pinus radiata* stand types. D. S. Thomson.
3. 1978. Fuel properties before and after thinning in young Radiata Pine plantations. D. F. Williams.
4. 1979. Using fire to reduce fuel accumulations after first thinning in Radiata Pine plantations. P. R. Billing.
5. 1979. Some of the effects of low intensity burning on Radiata Pine. P. R. Billing.
6. 1980. A low intensity prescribed burning operation in a thinned Radiata Pine plantation. P. R. Billing.
7. 1980. Some aspects of the behaviour of the Caroline Fire of February 1979. P. R. Billing.
8. 1981. Changes in understorey vegetation in Sherbrooke Forest following burning or slashing. R. Rawson and B. Rees.
9. 1981. Hazard reduction burning in the Big Desert. P. R. Billing.
10. 1981. The effectiveness of fuel-reduction burning: five case histories. P. Billing.
11. 1982. A fire tornado in the Sunset Country January 1981. P. Billing and R. Rawson.
12. 1982. A summary of forest fire statistics, 1972-73 to 1980-81. R. Rawson and B. Rees.
13. 1982. Fuel moisture changes under Radiata Pine. M. Woodman.
14. 1982. Fuel reduction burning in Radiata Pine plantations. M. Woodman and R. Rawson.
15. 1982. Project MAFFS/HERCULES: the Modular Airborne Fire Fighting System in Victoria. R. Rawson, B. Rees, E. Stuckey, D. Turner, C. Wood, and M. Woodman.
16. 1982. Using fire to reduce aerial fuels in first thinned Radiata Pine. P. R. Billing and J. V. Bywater.
17. 1982. Fuel properties before and after second thinning in Radiata Pine. M. Woodman.
18. 1983. Retardant distributions from six agricultural aircraft. B. Rees.
19. 1983. The Bright plantation fire: November, 1982. N. Watson, G. Morgan, and D. Rolland.

20. 1983. Otways Fire No 22 – 1982/83: Aspects of fire behaviour. P. Billing.
21. 1983. Otways Fire No 22 – 1982/83: A case study of plantation protection. P. Billing.
22. 1984. Forest Fire Statistics, 1974-75 to 1983-84. B. Rees.
23. 1985 The Avoca Fire, 14 January 1985. P. Billing.
24. 1985. Fuel management in Radiata Pine following heavy first thinning. P. Norman.
25. 1985. Effectiveness of Fuel Reduction Burning – 10 Case Studies. R. Rawson, P. Billing and B. Rees.
26. 1986. Operational aspects of the Infra-Red Line Scanner. P. Billing.
27. 1987. Heathcote fire: Bendigo Fire No.38 – 1986-87. P. Billing.
28. 1990. Fire behaviour and Fuel Reduction Burning – Bemm River. A. Buckley.
29. 1991. Fire hazard and prescribed burning of thinning slash in eucalypt regrowth forest. A. Buckley and N. Corkish.
30. 1987. Monitoring the ecological effects of fire. F. Hamilton (ed.)
31. 1992. Assessing fire hazard on public land in Victoria: fire management needs, and practical research objectives. A.A.G. Wilson.
32. 1992. Eucalypt bark hazard guide. A.A.G. Wilson.
33. 1992. Fuel reducing a stand of eucalypt regrowth in East Gippsland – a case study. A. J. Buckley.
34. 1992. Monitoring vegetation for fire effects. M.A. Wouters.
35. 1993. Elevated fuel guide. A.A.G. Wilson.
36. 1993. Wildfire behaviour in heath and other elevated fuels: a case study of the 1991 Heywood fire. M. A. Wouters.
37. 1993. The accumulation and structural development of the wiregrass (*Tetrarrhena juncea*) fuel type in East Gippsland. L.G. Fogarty.
38. 1993. A case study of wildfire management in the Byadlbo and Tringaringy Wilderness Areas. A.G. Bartlett.
39. 1993. Developing Fire Management Planning in Victoria: a case study from the Grampians. M. A. Wouters.
40. 1993. Fuel reducing regrowth forests with a wiregrass fuel type: fire behaviour guide and prescriptions. A.J. Buckley.
41. 1993. The effect of fuel reduction burning on the suppression of four wildfires in western Victoria. S.R. Grant and M.A. Wouters.
42. 1994. Fire behaviour and fire suppression in an elevated fuel type in East Gippsland: Patrol Track wildfire, February 1991. A.J. Buckley.
43. 1996. Fuel hazard levels in relation to site characteristics and fire history: Chiltern Regional Park case study. K. Chatto.
44. 2004. Surface fine fuel hazard rating – forest fuels in East Gippsland. G. J. McCarthy.
45. 1998. Effectiveness of firefighting first attack operations by the Department of Natural Resources and Environment from 1991/92–1994/95. G. J McCarthy and K.G. Tolhurst.
46. 1997. The development and testing of the Wiltronics T-H Fine Fuel Moisture meter. K. Chatto and K. Tolhurst.
47. 1998. Overall fuel hazard guide. G. J. McCarthy, K. Chatto and K. Tolhurst.
48. 1999. Development, behaviour, threat, and meteorological aspects of a plume-driven bushfire in west-central Victoria: Berringa Fire February 25–26, 1995. K. Chatto, K. Tolhurst, A. Leggett and A. Treloar.
49. 1997. Analysis of fire causes on or threatening public land in Victoria 1976/77 – 1995/96. C. Davies.
50. 2000. Assessment of the effectiveness and environmental risk of the use of retardants to assist in wildfire control in Victoria. CSIRO Forestry and Forest Products.
51. 2001. Effectiveness of broadscale fuel reduction burning in assisting with wildfire control in parks and forests in Victoria. G. J. McCarthy and K. Tolhurst.

52. 2003. Effectiveness of aircraft operations by the Department of Natural Resources and Environment and Country Fire Authority 1997-1998. G. J. McCarthy.
53. 2003. Modelling transport, dispersion and secondary pollutant formation of emissions from burning vegetation using air quality dispersion models. O. D. Valianatos, K. Tolhurst, S. Seims and N. Tapper.
54. 2003. Determination of sustainable fire regimes in the Victorian Alps using plant vital attributes. G. J. McCarthy, K. Tolhurst and K. Chatto.
55. 2004. Analysis of wildfire threat: issues and options. A. A. G. Wilson.
56. 2003. Prediction of firefighting resources for suppression operations in Victoria's parks and forests. G. J. McCarthy, K. Tolhurst, M. Wouters.
57. 2003. Ecological effects of repeated low-intensity fire in a mixed eucalypt foothill forest in south-eastern Australia. Summary report (1994-1999). Department of Sustainability and Environment.
58. 2003. Effects of repeated low-intensity fire on the understorey of a mixed eucalypt foothill forest in south-eastern Australia. K. Tolhurst.
59. 2003. Effects of a repeated low-intensity fire on fuel dynamics in a mixed eucalypt foothill forest in south-eastern Australia. K. Tolhurst and N. Kelly.
60. 2003. Effects of repeated low-intensity fire on carbon, nitrogen and phosphorus in the soils of a mixed eucalypt foothill forest in south eastern Australia. P. Hopmans.
61. 2003. Effects of repeated low-intensity fire on the invertebrates of a mixed eucalypt foothill forest in south-eastern Australia. N. Collett and F. Neumann.
62. 2003. Effects of repeated low-intensity fire on bird abundance in a mixed eucalypt foothill forest in south-eastern Australia. R. H. Loyn, R. B. Cunningham and C. Donnelly.
63. 2003. Effects of repeated low-intensity fire on terrestrial mammal populations of a mixed eucalypt foothill forest in south-eastern Australia. M. Irvin, M. Westbrooke, and M. Gibson.
64. 2003. Effects of repeated low-intensity fire on insectivorous bat populations of a mixed eucalypt foothill forest in south-eastern Australia. M. Irvin, P. Prevett, and M Gibson.
65. 2003. Effects of repeated low-intensity fire on reptile populations of a mixed eucalypt foothill forest in south-eastern Australia. M. Irvin, M. Westbrooke, and M. Gibson.
66. 2003. Effects of repeated low-intensity fire on tree growth and bark in a mixed eucalypt foothill forest in south-eastern Australia. K. Chatto, T. Bell and J. Kaellas.
67. 2003. A review of the relationship between fireline intensity and the ecological and economic effects of fire, and methods currently used to collect fire data. K. Chatto and K. Tolhurst.
68. 2003. Effects of fire retardant on vegetation in eastern Australian heathlands: a preliminary investigation. T. Bell.
69. 2003. Effects of fire retardant on heathland invertebrate communities in Victoria. N. Collett and C. Schoenborn.
70. 2003. Effects of fire retardant on soils of heathland in Victoria. P. Hopmans and R. Bickford.
71. 2004. An evaluation of the performance of the Simplex 304 helicopter belly-tank. H. Biggs.
72. 2004. Operational performance of the S-64F Airecrane Helitanker – 1997-98 fire season. H. Biggs.
73. 2008 Underpinnings of fire management for biodiversity conversation in reserves. M. Gill.
74. 2008. Flora monitoring protocols for planned burning: a user's guide. J. Cawson and A. Muir.
75. 2008. Flora monitoring protocols for planned burning: a rationale report. J. Cawson and A. Muir.

Supplementary report

1992. Ecological effects of fuel reduction burning in a Dry Sclerophyll Forest. K. Tolhurst et al.

www.dse.vic.gov.au