FIRE & BIODIVERSITY GUIDELINES FOR THE AVON BASIN

PREPARED FOR THE

AVON CATCHMENT COUNCIL AND DEPARTMENT OF ENVIRONMENT AND CONSERVATION

BY

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Consultant Report to the Avon Catchment Council and the Department of Environment and Conservation, August 2007.

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PREFACE

While preparing these guidelines and in discussions with people involved with fire management, it became apparent that the scope of the project was very ambitious and that one person was unlikely to come up with a set of detailed guidelines that meet ecological fire requirements for the large area covered by the Avon Basin. This was in part due to the scarcity of scientific knowledge and fire management experience about different vegetation communities in this semi-arid region. Similar guidelines in other states have taken many years to develop with the full support of the scientific community. But it was also ambitious in that there was insufficient time for much wider debate within the Department of Environment and Conservation, and with Fire and Emergency Services, Local Government, environment groups and the community. This debate and community engagement is required before more comprehensive fire management guidelines can be developed to meet the needs of all concerned.

Given the short time frame for the project it was decided that these guidelines would aim to compile existing knowledge of the ecological fire requirements for a range of vegetation communities within the Avon Basin, and highlight areas where there are gaps in our knowledge and management experience. In this context, the guidelines should be regarded as a preliminary step in the development of fire management guidelines for the Avon Basin, and not be used in a prescriptive way. The guidelines should hopefully stimulate further debate and research that is targeted to meeting the gaps in our knowledge.

Due to the low level of fire management in the Avon Basin in the past, there was seen to be a need to develop a more strategic and regional approach to fire management. Much of the content of these guidelines is aimed at this level, rather than at the local district or prescriptive level. Regional management structures and processes are discussed along with a simple means to prioritise prescribed burns. This prioritization process is an important part of regional planning due to the limited resources available to complete the work, and can be developed further as a decision model within a spatial GIS framework or as a Master Burn database that can be used at all levels of fire management.

An ArcGIS project was developed during this project that has a number of layers of DEC corporate spatial data clipped to the four main IBRA sub-regional natural resource management boundaries in the Avon Basin. This tool can be used as a basis for strategic fire planning although it needs to be updated with other important layers such as fire history and reserve condition data as this information becomes available. A summary of the vegetation associations, vegetation classes, key fire response species and threatened flora for each of the four main IBRA sub-regions is presented as tables in the Appendices section. This information enables strategic fire planning to be managed at the IBRA sub-regional level should this be required in the future.

EXECUTIVE SUMMARY

This report has presented the current knowledge about biodiversity values in the Avon Wheatbelt and aspects of their conservation with respect to fire management. A basic structure and process has been recommended for implementing and monitoring fire management for biodiversity conservation as a priority. This is based on using Beard-Hopkins vegetation associations as the basic fire planning unit and developing a GIS spatial database to identify areas available for burning. Fire management for biodiversity conservation needs to be considered in association with priorities for community protection as detailed in the associated Wildfire Threat Analysis report.

In compiling this report several issues were outstanding in their need for urgent attention.

1. Many areas have not been burnt for at least 50 years and are in urgent need of regeneration before they lose their reproductive capacity. This is particularly so for many of the smaller reserves and areas of remnant vegetation.

2. High priority vegetation communities and species requiring sensitive fire management include:

- Granite rock and ironstone outcrops
- Heath communities
- Salmon gum woodlands
- Fresh and brackish wetlands
- Malleefowl habitat
- Hollow dependent fauna habitat (eg numbats, red tailed phascogales, cockatoos)

3. There is currently a serious lack of DEC resources to effectively coordinate and manage a large prescribed burning program in the wheatbelt region, equivalent to that in operation in the forested south-west region. It was expected that these guidelines cover all tenures including fire management on private property, but there will need to be a major allocation of resources at all levels of management to effectively implement such a program.

4. Collation of fire history data should be a high priority for DEC, both at the local district level from recent burns and wildfire events, and from satellite imagery over the last 30 years. This information is vital for strategic fire planning and spatial analysis and should include data on fire season and intensity or patchiness of burn.

5. There is an urgent need for a forum on how to achieve the various scales of mosaic burning required, particularly for the protection of fauna habitat in mallee heath and shrubland communities. There are strongly divergent views within DEC and these need to be debated openly and outcomes implemented in an adaptive management framework.

6. Greater emphasis needs to be placed on managing other interactive factors, such as grazing pressure, exotic weeds and feral animals, before and after prescribed burning, to maximise the opportunities for regeneration and conservation of native flora and fauna. In particular, kangaroo and wallaby total grazing pressure needs to be actively managed before burning in smaller reserves to increase the reproductive capacity of adult plants and allow greater seedling survival after fire.

7. There is an urgent need for an effective process for fire response and fire behaviour monitoring. At present there is no coordinated monitoring system in place, either at the species, vegetation community or landscape scale. Dedicated DEC Science input and collaboration with district and regional staff is required to ensure effective design of monitoring systems and trials, collation and analysis of data and interpretation of results.

8. Every burn and wildfire event should be viewed as an opportunity to learn about fire responses and behaviours under different conditions. This body of information needs to be formally collated and communicated widely within DEC, FESA, the volunteer bush fire brigades and landholders and it is strongly recommended that an annual fire forum be held for this purpose.

9. The system of prioritization suggested in this report needs to be fully and openly debated as there are differing views within the DEC as to what extent fire management should cater to the needs of individual threatened and fire regime sensitive species. Most published information on fire responses is about individual species, rather than on the response of vegetation communities, which appears to be driving current fire planning and can lead to very complex burn prescriptions.

10. Members of volunteer bush fire brigades and FESA should be required to attend a training day each year to become more informed about fire management for biodiversity conservation. They openly acknowledge a lack of understanding in this area and a desire to learn. It is the responsibility of the DEC to ensure that all personnel engaged in fire operations be adequately briefed about the priorities, planning and strategies used to achieve these biodiversity conservation outcomes.

PART ONE

GENERAL FIRE AND BIODIVERSITY GUIDELINES

1. INTRODUCTION

1.1 Description of the Avon Basin

The Avon River Basin in the south-west of Western Australia is defined by the catchments of the Avon-Mortlock, Yilgarn and Lockhart river systems, on an ancient, dissected plateau of low relief. Most of the Yilgarn and Lockhart systems in the eastern part drain internally to salt lake chains, and only in very wet years will they flow into the Avon River. The western portion of the basin, past the Meckering fault line, is a zone of rejuvenated drainage with continuous stream flows in most years, flowing into the Swan River and down the Darling Scarp to the Indian Ocean. Mean annual rainfall across the Avon Basin varies from 250mm in the north-east to 500mm in the west (Fig. 1.1). However, most surface and groundwater resources are unsuitable for domestic or farm use due to high salinity (Avon Catchment Council, 2005).

The Avon Catchment covers 11.8 million hectares, of which about 70% (8.3 million ha) has been released for agriculture and is mostly (83.6%) cleared of native vegetation. The only vegetation left in the western agricultural area is on areas of shallow soils, rock outcrops and acid sand-plain soils, and in conservation reserves. There are approximately 491,000 ha of privately owned remnant vegetation, and 870,000 ha of conservation reserves in the agricultural region. These areas of native vegetation are typically small and highly fragmented, with about 50,000 privately owned remnants (average area 10ha) and 463 conservation reserves in a matrix of agricultural farmland (Fig.1.2). In the more arid eastern portion, there are another 3.5 million ha of mostly uncleared unallocated crown land, vacant crown land, conservation reserves, pastoral leases and mining leases (Avon Catchment Council, 2005).

Native vegetation in this region is recognised internationally for its high biodiversity value, particularly for its species richness and high level of endemism. There are over 4,000 vascular plant species of which 60% are endemic to the region. It lies in the South West Botanical Province which is recognised as one of 25 biodiversity hotspots in the world under significant threat (Myers *et al*., 2000). Clearing for agriculture and fragmentation of remaining vegetation is clearly the major threat to biodiversity in the agricultural area, followed by dryland salinity and altered hydrology as a result of rising saline groundwater (Keighery, 2004). Other threats include grazing by domestic animals, impacts of exotic weeds, feral animals and altered fire regimes (Beecham, 2003). In the largely uncleared eastern area, the main threats to biodiversity are feral animals, exotic weeds, mining and inappropriate fire regimes (Cowan *et al*., 2003).

Along with the large scale loss of native vegetation in the Avon Basin is the loss and significant threat to many flora and fauna species and ecological communities (Avon Catchment Council, 2005). There are over 120 Declared Rare Flora species and over

230 Priority Flora species in the region, in approximately 1840 known populations, as well as 450 flora species at risk from rising saline groundwater. Two mammal species are now extinct and 12 are lost from the region, with a further 5 threatened and 7 priority mammal species. Several bird species are endangered and many have suffered significant contraction in their occurrence. Fresh and saline wetland systems and fringing vegetation are being impacted by altered hydrology and increasing salinity. Other special value ecosystems such as granite outcrops are particularly sensitive to inappropriate fire regime.

Figure 1.1 The Avon Basin boundary used in these Guidelines is shown with local government boundaries and rainfall isohyets. Portions of the four main IBRA sub-regions included are Avon Wheatbelt P1 (AW1), Avon Wheatbelt P2 (AW2), Western Mallee (MAL2) and Southern Cross (COO2).

1.2 The need to conserve biodiversity

Biodiversity is the variety of life forms - it includes all the different plants, animals, invertebrates, micro-organisms, their genes and the ecosystems in which they exist. Biodiversity provides the ecological functions and processes on which we depend. It also has scientific, educational, aesthetic and spiritual values. Biodiversity has productive uses and provides opportunity for recreation and tourism. The natural biodiversity in this region is important to conserve for our children for them to value and appreciate. At a global scale, the south west WA flora is the only region in Australia recognised as a biodiversity hotspot which is unusually rich in endemic species and under considerable threat of species loss (Hopper, 2003).

There are numerous international, national and state agreements, laws, strategies and policies governing the conservation of biodiversity. The WA government has obligations under international conventions for the protection of wetlands, migratory birds and world cultural and natural heritage. At the national level, the last ten years has seen the development of the National Objectives and Targets for Biodiversity Conservation 2001-2005, nationally agreed Framework for the establishment of a Comprehensive, Adequate and Representative (CAR) reserve system for forest in Australia (1997), the Environment Protection and Biodiversity Conservation Act (1999), Wetlands Policy (1997) and the National Framework for Management and Monitoring of Australia's Native Vegetation (2001).

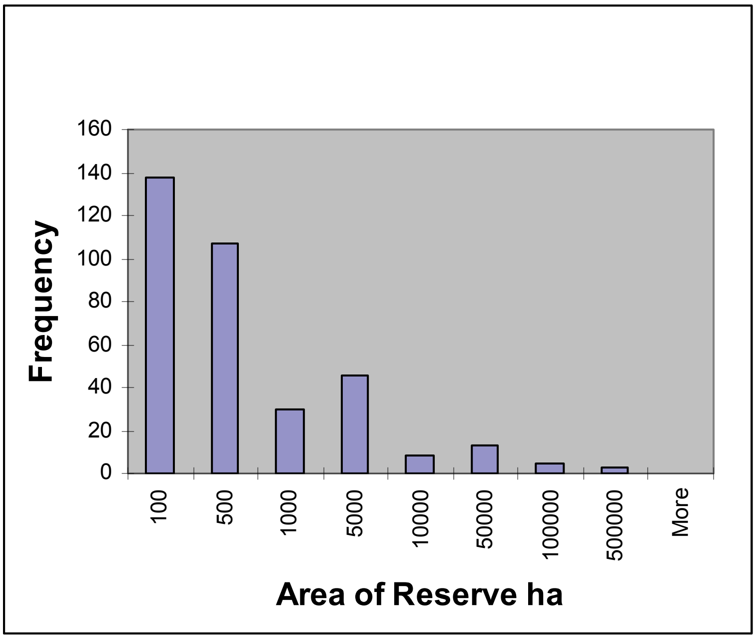


Figure 1.2 Histogram showing the frequency distribution of DEC reserves in the Avon Basin by area. These were calculated using the total area of each reserve, often consisting of several separate areas or polygons.

At the state level, the main instruments are the Wildlife Conservation Act (1950), the Conservation and Land Management Act (1984), Environmental Weeds Strategy and the Wetlands Conservation Policy for WA (1997). The WA government is currently preparing a new Biodiversity Conservation Act to replace the outdated Wildlife Conservation Act (1950) which will greatly strengthen protection for threatened species and communities with enhanced enforcement mechanisms (Department of Conservation and Land Management, 2004). The Department of Environment and Conservation (DEC), formally the Department of Conservation and Land Management (CALM), has primary responsibility for enacting the state legislation for conserving biodiversity and meeting national and international obligations.

In addition, there are various local government biodiversity plans and regional Natural Resource Management strategies that recognise the importance of conserving biodiversity across all land tenures with management strategies to prioritise and implement management actions (Avon Catchment Council, 2005). At the property level, some land managers have entered into voluntary agreements to manage their remnant vegetation to conserve biodiversity, such as for Land for Wildlife, World Wildlife Fund for Nature, Australian Nature Conservancy, Woodland Watch and the Remnant Vegetation Protection Scheme.

1.3 Fire regimes

Fire regime refers to the particular combination of season, intensity, frequency and scale of fire in a given area over a period of time. For many plant species in this semi-arid region, fire is a cue or stimulus for regeneration, while other species have evolved ways of avoiding fire. Inappropriate fire regimes may result in local extinctions of plants and animals, reduction in species richness or structural and spatial diversity of vegetation, or loss of fauna habitat. No one fire regime is optimal for all species, and some fire regimes can threaten biodiversity (Burrows and Armstrong, 2003).

1.4 Fire in the Avon Basin

Fire has been a part of the landscape in the south-west of Western Australia for probably 250 million years and with modern vegetation types and a drier Mediterranean climate for at least 30 million years (Hopper, 2003). Aboriginal people employed fire to manage their environment sometime during the last 60 thousand years, which probably involved quite different fire regimes to that which occurred previously (Hallam, 1975; Abbott 2003; Burrows and Wardell-Johnson, 2003). Aboriginal fire regimes were more frequent, smaller scale burns, and often used during the summer months to improve access to food resources, to maintain travel routes and for communication (Abbott, 2003). The considerable diversity of flora and fauna in the region has evolved during this prolonged period of changing fire regimes, with a wide variety of responses to different fire regimes being evident. Then more recently, Europeans utilised fire extensively to assist in clearing the land for agriculture, particularly between 1945 and 1970, and for burning crop stubble and managing the fire hazard of native vegetation by conducting fuel reduction burns. There have been considerably fewer wildfires since the 1970's following the reduction in land clearing (Department of CALM, 1995). Lightning is still responsible for a significant proportion of fires (Fig. 1.3). In more settled areas many wildfires are now deliberately lit or escape from accidents involving machinery or transport, for example during crop harvesting and stubble burning. This rapid change in fire regimes during the last 150 years, in a landscape where plants have evolved over millions of years, may have had significant consequences in terms of biodiversity conservation.

Resilience of plants and animals to a particular fire regime may affect their distribution and abundance in the landscape yet our knowledge of their responses to different regimes is extremely limited. There have been numerous studies on fire responses at the species and population level and to single fire events in the wheatbelt, but few at the vegetation community or landscape scale or to multiple fire events. Our knowledge of fire history in the Avon Basin is also limited, being anecdotal or with little detail about exact area burnt or the intensity or season of burn. More recent records for some areas have been derived from aerial photographs and Landsat satellite imagery interpretation. Detailed and continuous records of wildfire or prescribed burns have not been maintained in all areas by the Department of Environment and Conservation (DEC), Fire and Emergency Services (FESA), local government or bush fire brigades. Fire management has been largely based on responding to wildfire events and meeting community fire protection objectives. As a consequence, fire regimes in conservation reserves in the wheatbelt have been highly variable, often with long intervals between fires. Many fragments may not have been burnt since the original clearing fires over 50 years ago. As a result, many areas of vegetation are in decline as species reach their senescence stage.

Figure 1.3 Lightning strikes during summer are common in agricultural areas and fires can travel rapidly in open cereal and stubble paddocks. Fire fighting resources are usually able to attend these fires quickly and minimise the areas burnt.

Management of fire specifically for biodiversity conservation in the Avon Basin has been a recent consideration. A fire management strategy has been developed for the eastern reserves in the Katanning district of DEC and has dual goals of minimising risk of species loss and minimising risk to life and property from wildfire or inappropriate fire regimes (McClusky *et al*., 2003). This strategy is based on a 30 year minimum fire interval and maintaining at least 20% of the major vegetation communities in each reserve in each of the early, mid or late fire successional age class or seral stages. Another fire management plan, the Western Goldfields Wildfire Threat Analysis, was developed for a large area which includes a significant proportion of the eastern uncleared portion of the Avon Basin (Daniel, 2003). This analysis is based on assessing fire management risks and valuing assets, including declared rare and priority flora. It also aims to maintain a range of seral stages in each of six major vegetation types across the Western Goldfields region using known fire responses of the main structural species. Currently there are no guidelines for managing fire for biodiversity conservation or for community protection across the Avon Basin or wheatbelt region.

1.5 Scope and objectives for guidelines

The Avon Catchment Council developed the Avon Natural Resource Management Strategy in 2005 to provide direction and priority for protecting and conserving the natural resources in the Avon Basin (Avon Catchment Council, 2005). One of the regionally based projects under the Avon Investment Plan is the ND006 Fire Management and Biodiversity Project. This project was developed to address the complexities of fire management in a highly fragmented landscape with high biodiversity conservation value. The Fire and Biodiversity Guidelines are expected to guide fire management for all reserves and remnants of native vegetation in the Avon Basin, regardless of tenure and management responsibility, including that on privately owned land. They are based on the four major IBRA sub-regions contained largely within the Avon Basin ie Avon Wheatbelt 1, Avon Wheatbelt 2, Mallee 2 and Coolgardie 2. The guidelines will be used to underpin the development of fire management plans for each IBRA sub-region and area management plans at a later date.

The Wildfire Threat Analysis for the Avon Basin is being developed concurrently by Ecoscape (Mueller, in prep.) as part of the ND006 project. It will consider the risk of ignition, likely fire behaviour, suppression response and values at risk, including environmental and community values. This analysis will designate areas that require a low to high level of protection and assist with the planning of scarce resources.

The Fire and Biodiversity Guidelines are to consider the following:

- Identify key fire sensitive flora and fauna species for developing appropriate fire regimes for different vegetation types
- The maintenance of a range of seral stages in small to large remnants
- The impact of fire in small often isolated remnants where there are other threatening processes contributing to species loss
- Develop a process to group vegetation based on size and shape of remnant and vegetation type
- The impact of fire on ecosystem processes and population viability in a fragmented landscape
- Interactions between fire and weeds, feral animals and salinity in disturbed communities
- Impact of fire regime on flora and fauna communities with differing life attributes and adaptations to fire
- The influence of climate, soil type, hydrography and vegetation type in the different IBRA sub-regions on fire regime suitability

The guidelines will also address the following:

- Develop a monitoring strategy to assess and refine the fire management guidelines
- Assess the ecological impact of pre-burn management actions to guide best management practice
- Consult with local government authorities, FESA and landholders to identify other factors relevant for fire planning
- Identify gaps in knowledge of fire management in a fragmented landscape
- Identify gaps in biodiversity conservation information
- Encourage annual review of knowledge and the use of adaptive management
- Provide a basis for developing fire management plans for each IBRA sub-region
- Provide information for an education program for landholders and local government authorities

The key objectives of these Guidelines are therefore to:

- Outline the need for fire management for biodiversity conservation in the Avon Basin
- Define the principles for managing fire in a fragmented landscape
- Review the ecological knowledge about fire and biodiversity to improve the understanding for fire management by practitioners and those in the wider community
- Develop processes for managing fire and prescribed burning across all tenures, areas and vegetation types to enhance biodiversity outcomes

These Fire and Biodiversity Guidelines are in three parts. Part One is a general review of literature and background to fire planning, priorities and processes. This approach uses fire responses of flora and fauna to develop appropriate fire regimes. Part Two contains descriptions of nine major vegetation classes in the Avon Basin and recommendations for fire management based on existing knowledge and key indicator species. Part Three consists of five appendices with detailed tables of vegetation associations, vegetation classes, key fire response species and rare and priority flora for each IBRA sub-region. An ArcGIS project has also been developed using the IBRA sub-regional boundaries and based on DEC corporate databases to assist with fire management planning at the landscape and reserve level. This database will continue to be developed as more fire management information becomes available.

2. FIRE MANAGEMENT PRINCIPLES AND OBJECTIVES

2.1 Principles for fire management

There are a number of general fire management principles developed for Western Australia that are based on current scientific knowledge and which may underpin the ecological management objectives to enhance biodiversity conservation (Burrows and Abbott, 2003; Burrows 2005).

Principle 1. Fire is an environmental factor that has influenced, and will continue to influence, the nature of the south-west landscapes and biodiversity.

Principle 2. Species and communities vary in their response to, and reliance on, fire. Knowledge of the life histories of organisms or communities and their relationship to fire should underpin the use of fire in natural ecosystems.

Principle 3. Following fire, environmental factors such as landform, topography and life histories of various species, and random climatic events, often drive ecosystems towards a new transient state with respect to species composition and structure. This may prevent scientists from identifying which changes are specifically attributable to fire.

Principle 4. Fire management is required for two primary reasons, which are not necessarily mutually exclusive:

a) to conserve biodiversity
b) to reduce the occurrence of large intense wildfires.

Fire management should consider both ecological and protection objectives in order to optimise outcomes.

Principle 5. The damage potential, suppression difficulty and biological impact (killing power) of a fire and rate of recovery following a fire are in direct proportion to the fires intensity and size.

Principle 6. Fire diversity promotes biodiversity. An interlocking mosaic of patches of vegetation – representing a range of biologically derived fire frequencies, intervals, seasons, intensities and scales – need to be incorporated into ecologically based fire regimes if they are to optimise the conservation of biodiversity at the landscape scale.

Principle 7. Avoid applying the same fire regime over large areas for long periods of time, and avoid extreme regimes, such as very frequent or very infrequent intervals over large areas.

Principle 8.
The scale of the fire-induced mosaic should:

a) enable dispersal of young native animals

b) optimise boundary habitat

c) optimise connectivity or the ability of animals to move through the landscape

Principle 9.
All available knowledge, including life histories, attributes of native plants and animals and knowledge of Nyoongar fire regimes should be used to develop ecologically based fire regimes for a landscape or a vegetation complex.

Principle 10. Fire history, vegetation complexes and landscape units should be used to develop known and ideal mix of time since last fire.

Principle 11. Wildfire can damage and destroy both conservation and societal values, so a systematic and structured approach must be used to identify and manage the consequences of such an event.

Principle 12. Fire management should adapt to changing community expectations and new knowledge gained through research, monitoring and experience.

These principles provide a theoretical framework within which fire planning can be modelled. While the principles of creating and maintaining spatial and temporal variability in fire regimes to produce a patchwork of fire age mosaics is advantageous in most circumstances, it may not be achievable in the many small fragmented reserves and remnants found in the Avon Basin. These guidelines provide some tools to assess the situations where these principles can best be implemented.

2.2 Objectives for ecological fire management

The key biodiversity conservation objective for ecological fire management can be stated as:

"to manage fire appropriately to minimise the risk of extinction of any species from its current geographic distribution".

Species are prone to many disturbances which can impact significantly on their ability to survive. Ecological disturbances include drought, floods, severe frost, storm damage, disease and wildfire, and are considered to be central to the structuring of biodiversity (Huston, 2003). Other human induced disturbances such as clearing of vegetation, soil erosion, dryland salinity, grazing and predation by exotic animals and deliberately lit fires can significantly modify and transform an ecosystem. Disturbance refers to a physical change which can have positive or negative effects on species and ecosystems, and operate at the microhabitat to the landscape scale. In unproductive environments of low annual rainfall and infertile soils, where populations grow and recover slowly, an increase in the frequency of disturbance generally produces a decrease in species diversity. In contrast, in productive environments, an increase in disturbance frequency often produces an increase in species diversity (Huston, 2003). Fire can therefore be either a significant threat or a stimulant to species and communities, depending on how it is applied.

Species have evolved many adaptations to survive the effects of natural disturbances. These adaptations include the ability to resist the effects of disturbance, or to minimise their negative effects. Some species may also avoid some disturbances by being confined to refugia where the likelihood of harmful disturbances is minimal. Some of these adaptations allow plants and animals to survive different disturbance types, such as the ability of some plants to resprout after fire, grazing, severe drought or storm damage. However, species have not evolved to cope with the many recent human induced disturbances, especially where several of these may occur simultaneously or at frequent intervals. Biodiversity conservation in a highly fragmented human occupied landscape therefore requires considerable management effort to understand and minimise the risk of species loss from the many deleterious disturbances.

Fire as a disturbance regime may contribute to rarity and has the capacity to drive many rare species to extinction. It is these species that have highest priority for management. Species may be naturally rare because of their highly restricted geographic range, narrow habitat specificity and/or small population size. They may be the last remaining relictual elements from a previous climatic period, or recently evolved (Yates *et al*., 2003b). Rarity in the south-west WA flora is common, with high plant species turnover along habitat, environmental and geographical gradients. In a recent floristic survey of the wheatbelt, 60% of species were found in less than five quadrats (Gibson *et al*., 2004). Species may be rare due to large scale clearing of native vegetation and loss of preferred habitat, and the inability to disperse across the intervening agricultural matrix. Species may also become spatially or temporarily rare due to altered fire regimes (Yates *et al*., 2003b).

The mechanisms for fire-driven decline and extinction of plant populations include death of standing plants and seeds, failure of seed release and/or germination, failure of seedling establishment, interruption of maturation or developmental growth, and failure of seed production (Keith, 1996; Whelan *et al*., 2002). Both high and low frequency fire regimes may lead to extinctions. Non-sprouting plants are more sensitive to frequent fire regimes because they depend entirely on seeds for persistence and require a minimum fire free period to reach reproductive maturity (Yates *et al*., 2003b). A long absence of fire may also cause extinction if the adult population senesces and if seeds have low viability. Some animals are also vulnerable to inappropriate fire regimes, particularly those which rely on dense vegetation, have specific diets, poor mobility and dispersal, low reproductive rates or are susceptible to predation.

Ecological fire management focuses on identifying broad categories of plant and animal population responses to fire regimes using life history attributes which influence rates of mortality, recolonisation, post-fire survival, growth, reproduction and population increase (Yates *et al*., 2003b). In some circumstances it will be necessary to manage areas to ensure the survival of highly threatened and fire sensitive species, but generally speaking, managing fire specifically for one species will result in less optimal conditions for a range of other species. While it is not possible to prescribe fire regimes to optimise all known (and unknown) ecological requirements, the “do nothing” management scenario may result in serious decline and extinction of species.

2.3 Information

The key to developing and implementing the Fire and Biodiversity Guidelines is the availability of comprehensive, reliable and up to date information. The capacity to make recommendations for ecological fire management depends entirely on knowing the location of species and communities, whether they are at risk, what are their responses to different fire regimes, previous fire history and likely fire behaviour. This information is mostly available for the forested south-west regions of Western Australia, where the forest resource has been actively managed for fire over the last 50 years, but is less comprehensive for the wheatbelt region, particularly for fire history and fire behaviour.

Some spatial data is available at the local scale, such as the location and condition of threatened flora populations and weed mapping, but most data has been captured at the regional scale, with lower levels of accuracy. In addition, there is much relevant information on fire responses in the literature, but this is not available in GIS format. Information on some fire-related traits are known (eg whether species are resprouters or obligate seeders), but there is limited information on many other critical traits for describing fire response (eg juvenile period, years to senescence and seed longevity). Information on fire responses has been provided for these Guidelines for a range of species based on literature reviews and field observations (N. Burrows unpublished data; C.Yates unpublished data). The results of a large scale biodiversity survey of the WA agricultural zone (Keighery *et al*., 2004) with 682 terrestrial quadrats have been databased and could be available in GIS format. Many detailed flora and fauna surveys have been conducted in reserves in the Avon Basin (eg Muir, 1978; Muir *et al*., 1978; Dell *et al.,* 1979; Coates, 1990), with detailed vegetation mapping, and it is hoped that these will be available to local area managers for more fine scale fire management planning. A new on-line database, NatureMap, is being developed by

DEC which shows the known distribution of individual flora and fauna species, and the flora and fauna species recorded for any particular area (Paul Gioia, *pers. comm.*). This database is due to be released this spring, and will be an invaluable information resource for fire managers.

An ArcGIS9 project has been developed using data layers provided by the Department of Environment and Conservation (DEC) from their corporate data set. It is expected that this GIS project will be available for all fire managers and practitioners to assess relevant information for the local area being managed. It is anticipated that additional information will be added to this project as it becomes available, and as fire records are updated. An additional layer has been developed for these Guidelines which show broad vegetation classes which are based on the Beard-Hopkins vegetation association mapping.

Data layers used for this project include:

- Avon Basin NRM administrative boundary – the NRM boundary was chosen for this project over the Avon catchment boundary to simplify administration issues
- IBRA sub-regional boundaries – four sub-regions – Avon Wheatbelt 1, Avon Wheatbelt 2, Mallee 2 and Coolgardie 2 – were clipped to the Avon NRM boundary
- Beard-Hopkins pre-European vegetation mapping at 1:250,000 – shows all vegetation associations and systems, main species, with National Vegetation Information System (NVIS) hierarchy levels 1-6.
- Remnant vegetation for all land tenures – Goldfields data not available
- Modus fire history remote sensing mapping from 1993 to 2005 – year and month of fire
- Mean annual rainfall isohyets
- Hydrography – major and minor drainage lines and water bodies
- Land tenure – type of reserve, purpose, area and perimeter
- Major towns and roads
- Declared Rare Flora – populations of all DRF and priority species
- Threatened fauna - records of sightings or trappings from Fauna File
- Threatened Ecological Communities
- Salinity risk

Other data layers that ideally should be added to the project include:

- Landsat fire history data from 1970s onwards
- NatureMap flora and fauna records
- Recovery catchments and targeted landscapes
- Important Wetlands – national and regional
- Research sites – DEC, WA Museum, CSIRO, Universities, Wandoo Watch sites, flora survey quadrats, other research projects
- Vegetation change
- Bird Australia Bird Atlas - data survey sites and species recorded
- Private reserves, covenanted areas, Land for Wildlife and bushland with other voluntary management agreements
- Flora and fauna recovery areas and translocation sites
- Western Shield baited areas
- *Phytophthora cinnamomi* infested areas

In addition, there needs to be a Master Burn Database developed for the Avon Basin which contains fields with data relevant to each reserve, group of small reserves, defined fire management units in continuous vegetation and significant remnant vegetation. This database contains information based on the GIS analyses as well as notes on previous fire events and significant conservation and management issues for each reserve. It is used in the prioritisation and ranking process to determine areas most in need of fire management. This database would be a major undertaking in the Avon Basin but would be invaluable in fire planning.

3. RESPONSES TO FIRE

3.1 Plants

Patterns of response to fire vary greatly across species and communities, and various classification systems have been designed to describe and predict these responses. The system commonly referred to for fire management purposes is the ‘vital’ or life attributes of plants which classifies species according to their modes of dispersal, recruitment and post-disturbance persistence (ie seedling, vegetative or both), patterns of establishment and length of critical life stages (Noble and Slatyer, 1981). However, Whelan *et al*., (2002) conclude that fire responses are site-specific and that it is impossible to predict the effect of a fire on any species. Plant responses may more simply be grouped into four types: geophytes, ephemerals, obligate seeders and resprouters. Species with similar life history characteristics are expected to have similar responses to individual fire events, assuming they have been exposed to 100% leaf scorch. It should be noted that particular life attributes may have evolved in response to other factors, such as drought, grazing or low soil nutrient status, which may coincidentally enable them to persist in a fire-prone environment.

Geophytes – this group of species avoids the main impact of fire in time and space by having bulbs, corms, tubers and rhizomes below ground level. Above ground parts die back in summer or in unfavourable conditions, and are therefore out of phase with the ‘normal’ summer-autumn fire season and escape most wildfires (Bell *et al*., 1982; Pate and Dixon, 1982). In many species, flowering is greatly stimulated by a summer fire (Pate and Dixon, 1982). Examples of geophytes include species in *Drosera*, *Caladenia, Pterostylis, Thelymitra, Thysanotus* and *Burchardia*. However, the response of these species to fire during their growing season is largely unknown. Many orchids are known to be killed by fires between late May to November, while the leaf is green and before the new tuber is fully formed (Andrew Brown, personal communication). Fire intensity during the dormant period of December to March does not affect orchid survival, and often stimulates flowering. Many geophyte species have transient seed banks and germination must occur in the first year after seed release or viability is lost (eg Orchidaceae) (Keith *et al*., 2002).

(a)

(b)

Figure 3.1 Examples of geophytes which are killed by fire during the growing season (a) *Caladenia flavens* and (b) *Caladenia longicauda* subsp. *eminens*

Ephemerals – This group of species also avoids most fire by completing its life cycle within one year and produces soil stored seed which germinates in response to heavy rainfall or disturbance. A sub-group, called fire ephemerals, germinate in large numbers following fire and usually complete their life cycle before the next fire event (Fig 3.2). They have fast growth rates, early reproductive maturity and short life spans (3 months – 4 years) and utilise nutrient-rich sites following fire. This group of species is unlikely to be at risk from frequent fire. Examples include *Gyrostemon*, *Goodenia, Alyogyne, Solanum, Hibiscus, Podotheca, Stipa* and *Pimelia*. This rapid response of fire ephemerals contributes to the typical temporary increase in species diversity observed following fires.

Figure 3.2 An example of a fire ephemeral, *Codonocarpus cotinifolius* which grew rapidly but is reaching the end of its natural life several years after a major wildfire.

Obligate seeders – Adult plants in this group are usually killed by fire intensity that causes 100% scorch, and germinate from seed either stored in the soil, or in woody capsules on the plant (termed bradysporous or serotinous species). Many will survive fire at lower intensities, and some have the capacity to resprout after mild fire. Most obligate seeders are long lived (>15 years) and produce some flowers within the first 4 years of germination and continue to flower profusely building up a sizeable seed bank within 10 years (Bell *et al*., 1984), although this may take considerably longer for eucalypt woodland species, and species in drier areas. Examples include many species in *Hakea, Petrophile, Beaufortia, Callitris, Xylomelum, Allocasuarina, Dryandra, Eucalyptus* and *Melaleuca* (Fig. 3.3). For one species studied in the Stirling Range, *Hakea pandanicarpa* subsp. *crassifolia* , only 40% of individual plants were flowering by 7 years post-fire and only 67% had fruited by age 10 years. Another species, *Hakea corymbosa,* required a fire-free period of 8 years to reach reproductive maturity (Barrett *et al.,* 2004).

The bradysporous fire response sub-group is the most likely to suffer population declines through mismanagement of fire. If a population of regenerating bradysporous plants is killed by a second fire before it reaches reproductive maturity, then it is likely to decline and may become locally extinct. Similarly, if a population is not burnt before it reaches senescence, then the stored seeds may not have an opportunity to be released and germinate.

(a) (b)

Figure 3.3 Examples of bradysporous species found in the Avon Basin.
(a) *Actinostrobus pyramidalis* and (b) *Allocasuarina campestris* which store their seed in woody capsules.

Many bradysporous seeders, including *Eucalyptus* species, release small numbers of seed without fire (called incipient bradyspory), although survival and growth of seedlings are usually greater following a fire (Burrows *et* al., 1990; Yates *et al*., 1994; Dixon and Barrett, 2003). Indeed, this incipient bradyspory appears to be more common than absolute bradyspory (Lamont, 1991). Managing the inter-fire periods to optimise survival of these incipient bradysporous seedlings may be as important as managing seedlings flowing fire (Dixon and Barrett, 2003).

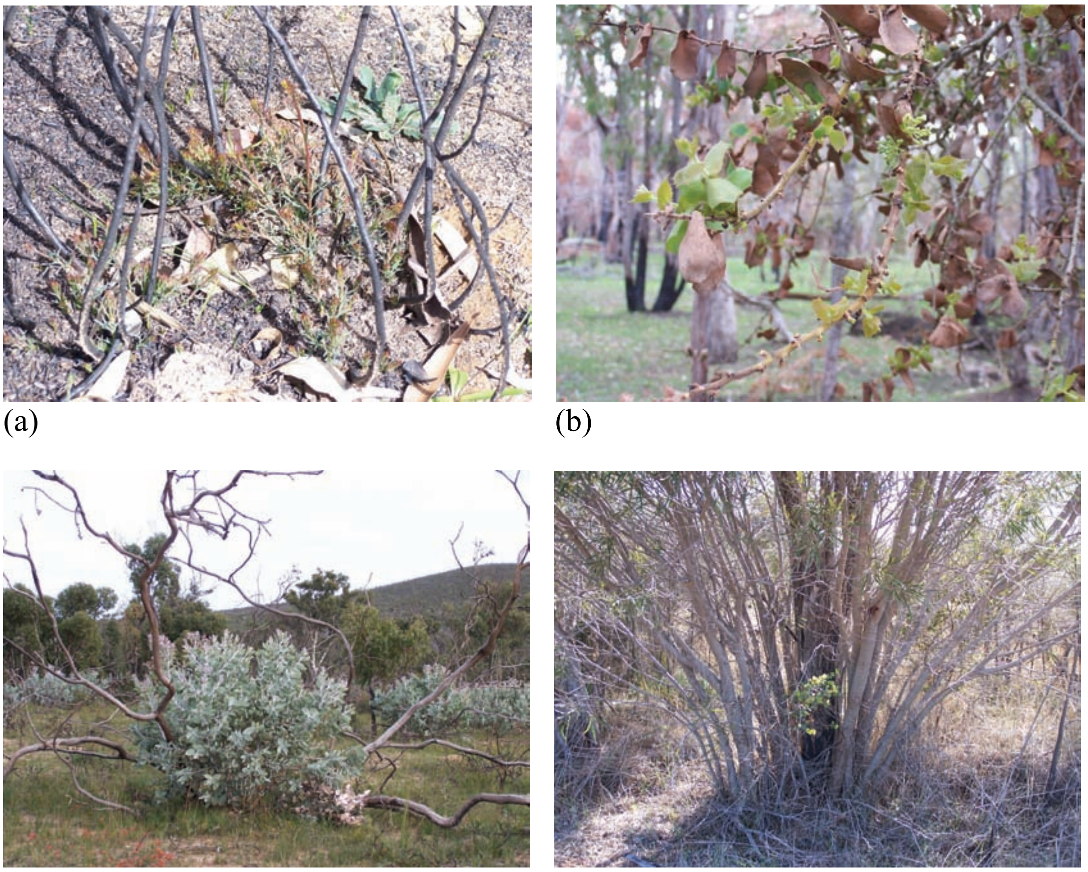
Seeds of bradysporous species usually only persist for the life of the plant. Most bradyspores have large seeds with adequate nutrients to seedling growth for up to 18 months (Milberg and Lamont, 1997), and do not rely on the release of soil nutrients following fire. Non-bradysporous obligate seeders usually produce long lived seed with hard seed coats (eg *Acacia*) and/or have seed dormancy (eg *Kunzea*) that may persist in the soil for 30-50 years. Germination of these more persistent seeds is dependent on heat shock or smoke cues (Dixon *et al*., 1995; Keith, 1996), and this varies with the intensity of the fire, soil moisture and the depth of seed burial (Fig 3.4). Seed stored in the soil over time is not readily depleted by repeated fire.

(a) (b)

Figure 3.4 Examples of prolific regeneration of seedlings from obligate seeder species following hot fires (a) *Santalum acuminatum* and (b) *Acacia pulchella*.

Resprouters – This group of species has the ability to resprout either from epicormic buds on the stem or base, or from rootstock (lignotuber) buds below the ground following fire, or other defoliating stresses (Fig. 3.5). Resprouters are generally long-lived plants and successfully compete for light, water and nutrients due to their shoot density, vigorous growth and ability to regenerate vegetatively from below ground storage organs (Bell *et al*., 1984). They have much reduced capacity to reproduce from seed, and seedling growth is generally slower than for obligate seeders, as more photosynthate is directed to building the starch reserves in fire resistant organs (Keith *et al*., 2002). All species will regenerate from seed under some circumstances, particularly after a major disturbance, such as severe fire (Tolhurst and Friend, 2003). Some woody resprouters may take up to 15 years to develop fire-resistant lignotubers (Keith *et al*., 2002). Bud and starch reserves may be depleted by high frequency fires, and some resprouters may be weakened or killed by high intensity fires where excessive heat penetrates the bark or topsoil (Burrows and Wardell-Johnson, 2003).

The species composition of the south-west flora, particularly heathland, is very complex and diverse and contains representatives from all fire response types. In addition, some species can vary in their response to fire depending on their stage of development, season and intensity of fire, and between varieties or subspecies (Burrows and Wardell-Johnson, 2003). However, the proportion of species in different fire response categories may reflect past patterns of fire in different communities and can be a guide to appropriate fire regimes. On drier upland sites with jarrah forest and wandoo woodland about 75% of all understorey species, including trees, woody shrubs and perennial herbs resprout after fire (Burrows and Wardell-Johnson, 2003). In less fire prone environments, such as riparian zones, swamps and granite outcrops where fuel if sparse or there is moisture or other natural barriers to fire, the proportion of obligate seeders is often 50 – 80%. These latter habitats often have greater numbers of threatened and fire sensitive flora.



(a) (b) (c) (d)

Figure 3.5 Examples of resprouting plants (a) *Hakea lissocarpha* resprouting from a basal lignotuber; (b) *Hakea prostrata* resprouting from epicormic buds underneath the bark; (c) *Eucalyptus pleurocarpa* readily resprouts to produce a many-stemmed mallee form, and (d) *Acacia acuminata* which can resprout from basal shoots or germinate from soil-stored seed after fire.

Single fire events rarely have an effect on species composition, and most species that occur on the site will regenerate, but using the same fire regime repeatedly can be detrimental. Frequent burning may cause the decline of absolute bradyspores if juvenile plants are burnt before their first flowering. Loss of shrubby species by frequent burning may then impact on fauna habitat (eg mallee fowl, splendid wren, woylie and tammar). Long unburnt areas may lead to the decline of species with short-lived seed or bradysporous species that only regenerate after fire. Low frequency fires may also lead to the decline of woody heathland understorey species due to competitive exclusion (Keith *et al*., 2002). Fire intensity can also alter species composition. Obligate seeders with hard coated seeds stored in the soil are sensitive to cool burns, as insufficient heat may be transferred through the soil to germinate the seed. Low intensity fires may also lead to low recruitment rates due to fewer seeds being released and greater competition from unburnt plants. Intense fires may damage epicormic buds and consume viable seeds, but are required to regenerate many woodland species (Burrows *et al*., 1990; Yates *et al*., 1994).

3.2 Animals

Many individual animals can die from the direct impact of fire, especially those that shelter in dense vegetation, grassy nests in trees or in tree hollows. Others can escape in patches of unburnt vegetation, in burrows, hollow logs, rock shelter and in moist gullies, or by fleeing the area. Following a fire, before vegetative cover and structure returns, predation and competition for scarce food and nesting resources become critical for population survival. The response of animals to fire is largely determined by the recovery patterns of plants and the associated food, shelter and nesting substrates they provide and by the time since fire or seral stage.

3.2.1 Mammals

Some small mammals increase in abundance during the early **seral stage** after fire that generally have non-specialised diets and a flexible breeding pattern (eg southern brown bandicoot (*Isoodon obesulus*), ash-grey mouse (*Pseudomys albocinereus*)). Mid seral stage species (eg dunnarts (*Sminthopis spp.*) generally require denser vegetation, shelter in more flammable refuges and have a less generalised diet. Late seral stage species (eg western mouse, western pygmy-possum (*Cercartetus concinnus*), honey possum (*Tarsipes rostratus*)) shelter in flammable vegetation, have specific diets and seasonal and synchronised breeding patterns (Friend, 1993). Honey possums are severely affected by intense large scale fires, but can survive low intensity patchy fires. Their abundance following fire is strongly correlated with the time for most plants to reach peak flowering, demonstrating their dependence on the availability of nectar and pollen. Successional changes after fire are more evident for small mammals than among medium and large animals in the south-west of WA (Friend and Wayne, 2003).

Numbats (*Myrmecobius fasciatus*) seem to prefer areas burnt relatively infrequently, but the availability of their termites, their main food source, are not directly affected by fire. However, the loss of habitat logs and *Gastrolobium* thickets may increase mortality through predation (Fig 3.6), and autumn fires every 20 – 30 years were recommended in the Dryandra Woodland to maintain a diversity of successional stages of vegetation (Burrows *et al*., 1987). A hot autumn fire destroyed about 50% of hollow logs suitable for numbats, but created almost as many, whereas a moderate spring burn destroyed about 25% of hollow logs and only replaced only about 3% (Friend and Wayne, 2003). Salmon gum and Wandoo woodlands produce the majority of habitat hollows in the wheatbelt but are being destroyed by all causes faster than they are being created. Numbats were previously found in York gum and mulga woodland where termites were plentiful but there were few hollow logs. This requirement for hollow logs was apparently not of prime importance before the arrival of the fox (Friend, 1995).

Tammar wallabies (*Macropus eugenii*) and woylies (*Bettongia penicillata*) are fire dependent animals that require intense fire under dry conditions every 20 – 30 years to regenerate thickets of *Gastrolobium, Bossiaea, Allocasuarina, Acacia* and *Melaleuca* (Fig. 3.6). The *Gastrolobium* thickets provide additional protection from introduced predators due to the natural 1080 poison in their foliage. Woylies reach a stable pre-fire population 4-5 years after fire, due to recolonisation from adjacent unburnt areas (Christensen, 1980). Frequent cool burns encourage prolific grass growth on which tammar graze, but fails to germinate these obligate seeder shrubs required for their protection, whereas longer fire intervals lead to senescence of obligate seeders. Both scenarios result in the loss of protective canopy cover and decline in population due to predation pressures (Christensen, 1980; Burrows *et al*., 1987). It is interesting to note that this apparent requirement for dense thickets is not evident on islands where tammar are able to exist in long unburnt vegetation in the absence of foxes and dingoes.

(a)

(b)

(c)

(d)

Figure 3.6 Fauna habitat in wandoo woodland (a) hollow habitat logs need to be maintained for numbats; (b) senescing stands of *Gastrolobium spinescens*. Dense regeneration of reseeder shrubs following hot fires provides good protection from native and feral predators in sheoak low woodland (c) *Allocasuarina huegeliana* and (d) *Gastrolobium* seedlings.

Red-tailed phascogale (*Phascogale calura*) occur in tall wandoo and rock sheoak woodland with *Gastrolobium* shrubs which confer some protection from introduced predators. It is an arboreal animal and uses tree hollows for refuge and nesting sites. It is most commonly found in long unburnt climax vegetation which provides both potential nesting hollows and sufficiently dense foliage for protection and foraging (Bradley, 1995). An experimental autumn burn at Tutanning killed some animals, but recolonisation was rapid from nearby unburnt patches. Following the fire which killed the rock sheoak nesting trees, phascogales started using hollows in wandoo trees. However, this change in nesting habit put them at greater risk of predation as they have to travel across the ground to reach foraging sites rather than leaping across branches in the dense sheoak canopy. Thus fire scale and intensity in relation to the mobility of the animal, and level of predation, will determine the response pattern to fire.

3.2.2 Reptiles and frogs

Reptile species which burrow and forage in open habitats tend to be early colonisers after fire, whereas species which shelter and forage under litter, or rely on prey that lives in litter, tend to be more common in late seral stages (Friend, 1993). Many reptiles escape fire by sheltering under rocks or logs, or in burrows (Fig. 3.7). A few reptile species in the wheatbelt of WA which seem to be vulnerable to single fire events include the arboreal gecko (*Diplodactylus spinigerus*), an agamid lizard (*Ctenophorus reticulatus*) and a skink (*Ctenotis pantherinus*). At least 4-5 years was required before mature adults of the lizard and skink recolonised burnt remnants where there was adjoining unburnt habitat (Lambeck, 1999). Most frog species in mallee heath are more abundant in older vegetation, breed in moist environments, and burrow which provides considerable resistance to fire.

Figure 3.7 The ornate crevice dragon (*Ctenophorus ornatus*) shelters in narrow rock crevices in rock outcrops and so can avoid most fire events.

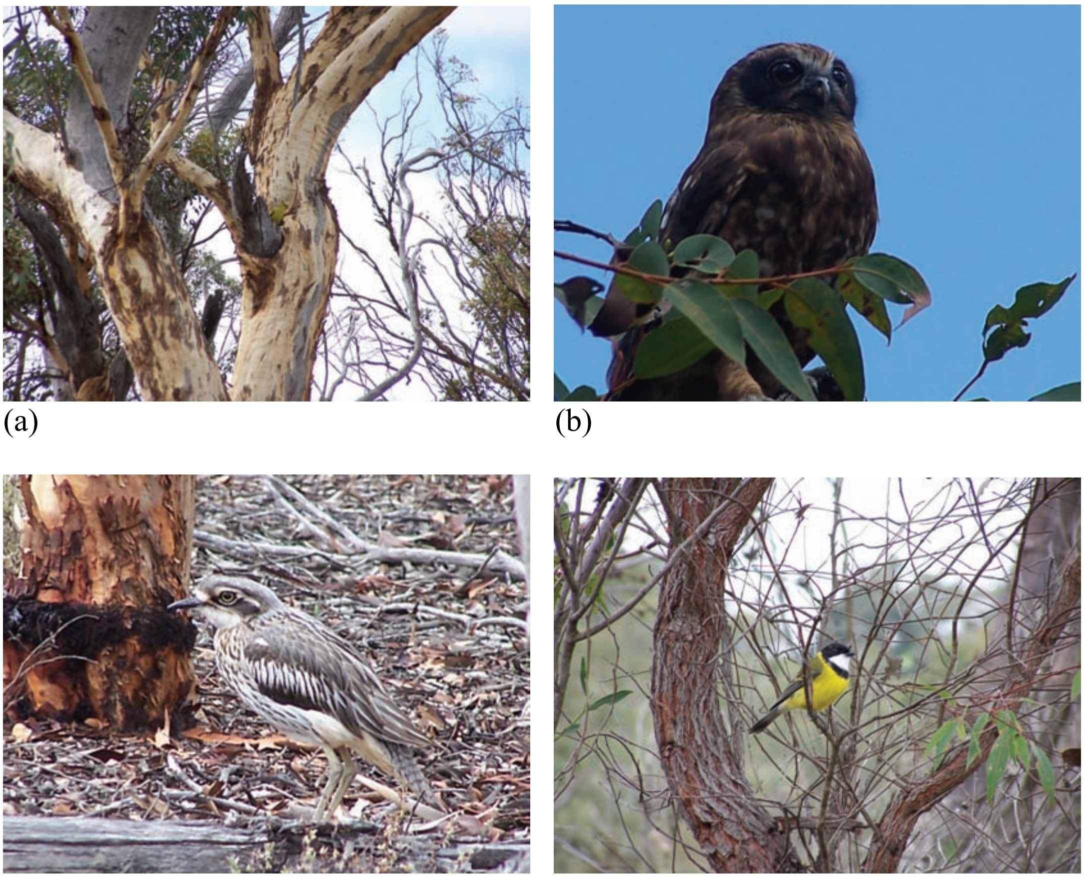
3.2.3 Birds

Wildfire can have direct and indirect effect on birds causing greater impact on species abundance than species diversity. Significant decreases in the abundance of thornbills, treecreepers, golden whistler (*Pachycephala pectoralis*) and grey fantail (*Rhipidura fuliginosa*) were found in regrowth eucalypt forests near Eden (Recher *et al*., 1985). However, other studies have shown that diversity and abundance of bird species may increase for 2-3 years following mild fires (Christensen *et al*., 1985). Depending on the intensity of the burn and scorch height, different feeding guilds of birds may be adversely affected. Some birds which prefer open ground or feed on seeds of annuals and grasses are favoured by frequent fire (eg black-faced cuckoo-shrike (*Coracina novaehollandiae*), pigeons, quail, parrots, white-winged triller (*Lalage tricolor (sueurii)*), dusky woodswallow (*Artamus cyanopterus*)). Scarlet robins (*Petroica multicolor*) often take advantage of easier access to find ground insects and numbers often increase immediately after a fire, but can suffer a long term decline (Brooker, 1998). *Dryandra sessilis*, an important food source for many birds, is known by apiarists to flower freely 3-4 years after fire, but does not set seed until 8 years or reach maximum nectar flow until 12 years after fire (Muir, 1985). Canopy gleaning and dwelling insectivores may be affected little by ground fires but may decline if the fire scorches the tree or shrub canopy. Following fire, epicormic shoots with high nutrient status attract psyllids and other foliage eating insects which favour pardalotes and some honeyeaters. Species that require dense shrubs for nesting may be affected even by mild fire (eg thornbills, scrubwrens, white-breasted robin (*Eopsaltria georgiana*), grey fantail, splendid fairy-wren (*Malurus splendens*)) (Recher *et* al., 1985; Ford, 1989).

Although birds may survive wildfire and mild prescribed burns, their subsequent reproductive success is often reduced. Brooker and Rowley (1991) found after a severe fire in heathland, breeding of splendid fairy-wrens in the year after fire was delayed by 3 – 5 weeks and the number of nests built by the second season nearly doubled due to a shortage of nesting material, difficulty in attaching nests to available substrates and subsequent nest failure. Fewer adult western thornbills (*Acanthiza inornata*) attempted nesting which was delayed by 5 weeks due to a shortage of food for egg production. Two other species (white-browed scrubwren (*Sericornis frontalis*) and white-cheeked honeyeater (*Phylidonyris nigra*)) did not nest for 2 years after the fire, and inland thornbill (*Acanthiza apicalis*) had not nested even 5 years later. There is a complex relationship between fire and bird populations which involves how a particular fire frequency, intensity and season affects their habitat ie shelter, nesting sites, foraging substrate and food abundance (Burbidge, 2003).

Many bird species depend on tree hollows in older woodlands for nesting and nocturnal and diurnal roosts. In the wheatbelt, most tree hollows used by cockatoos, corellas, parrots and owls are in salmon gum and wandoo, species which have been largely cleared for agriculture (Fig. 3.8). The threatened short-billed black cockatoo (*Calyptorhynchus latirostris*) is particularly vulnerable to loss of suitable hollows and competition for hollows by galahs and corellas. Distances between remaining nesting trees and feeding habitat in kwongan heathland have become so great that chicks starve to death (Saunders, 1977). Other birds use smaller tree hollows, such as striated pardalote (*Pardalotus striatus*), rufous treecreeper (*Climacteris rufa*) and tree martin (*Hirundo nigricans*). Hollows are continually being formed in trees by collapse of heartwood, branches breaking and excavation by termites and animals. High intensity fires can destroy existing hollows and fell older trees, but can also greatly increase the number of hollows available. The use of fire to regenerate these woodlands requires special care so that existing hollows are not lost at a faster rate than new hollows are formed.

Malleefowl (*Leipoa ocellata*) are another threatened species in the wheatbelt, with only scattered populations remaining in sub-optimal habitat. Pairs occupy permanent territory and require long unburnt tall mallee, low woodland or *Acacia* scrub on sandy soil with fairly complete canopy and abundant litter for nest mound formation. They feed on seeds and herbage in mulga (*Acacia aneura*) and other *Acacia* species. Fox predation is a major threat to malleefowl where there is insufficient vegetation cover and protection. This is exacerbated by inappropriate fire regimes, particularly large scale homogenous fires which can cause local extinctions. It may take 15 years before habitat is suitable to breed after extensive fires due to a shortage of litter material for nesting or greater exposure to predators, and at least 40 years before maximum populations are attained (Beshemesh, 2000). Malleefowl do feed in recently burnt areas but rely on unburnt refuges for shelter and nesting. A tight mosaic of burning may provide food plants and adequate cover (Beshemesh, 2000), but the appropriate scale of this mosaic patch burning is unknown.



(a) (b) (c) (d)

Figure 3.8 Two bird species which require suitable hollows in old trees for nesting are (a) the regent parrot (*Polytelis anthopeplus*) and (b) the southern boobook owl (*Ninox novaeseelandiaei*). The bush stone-curlew (*Burhinus grallarius*) (c) lives and nests on the ground in open woodland and is prone to wildfires, while the golden whistler (*Pachycephala pectoralis*) (d) nests in shrubs and uses light plant materials bound by cobwebs, which may be difficult to find for several years following fire.

In regions where native vegetation is highly fragmented, up to 20% of bird species may be at risk in situations where fire occurs with even moderate frequency because there are fewer opportunities for recolonisation (Burbidge, 2003). The southern scrub-robin (*Drymodes brunneopygia*) requires at least 20ha of suitable heath/shrub/mallee habitat to be within 2km (Brooker *et al*., 2001) to successfully recolonise an area if it is forced to leave its territory due to fire. Similarly, the red-capped robin (*Petroica goodenovii*) and the rufous treecreeper require 10ha of woodland within 5.3km and 8km respectively for recolonisation (Brooker *et al*., 2001). Lambeck (1999) considered 25ha was the minimum patch size for focal bird species in the Kellerberrin district with an inter-patch distance of 2km. These aspects of fragmentation have been modelled for birds in the Avon Basin and could provide a useful guide for fire management. For example, if there are no patches of similar vegetation of 25ha or greater within 2km of the area to be burnt, then a decision could be made either to not burn or to leave a minimum area of 25ha unburnt, at least until the burnt vegetation is once again suitable habitat. These decisions could be made after bird surveys to determine the presence of certain focal or indicator species.

A number of declining or specialist bird species can be used as key indicator species for fire management in AW1 (based on data from Brooker *et al.*, 2001). These include less mobile sedentary birds that feed mainly on the ground, small birds that need dense shrubby understorey for nesting and foraging, birds that require large areas of long unburnt vegetation and dense litter and birds that require maintenance of suitable nesting hollows (Table 3.1). Key indicator species have been chosen from those listed by Brooker *et al.* (2001) which are easily recognizable and more likely to be encountered in suitable habitat. Nocturnal birds, birds of prey and wetland birds and those confined mostly to salt lakes or succulent vegetation are not included.

Table 3.1 Potential key bird indicator species for fire management in AW1, showing their preferred habitat and conservation status. Declining species are those listed by Saunders and Ingram (1995). Endangered, vulnerable and near threatened species are those listed by Garnett and Crowley (2000).

Species name	Preferred habitat	Fire management issue	Conservation status
Malleefowl	Mallee shrubland	Large areas long unburnt shrubland, dense litter, low mobility	Vulnerable
Rufous treecreeper	Open woodland	Ground and bark feeder, hollow limbs	Declining
Southern scrub robin	Mallee shrubland	Ground feeder, dense litter, thickets, low mobility, needs large remnant	Declining
Bush stone-curlew	Open woodland/ sparse shrubs	Ground feeder, dense litter, low mobility, nests on ground	Near threatened
Painted button quail	Open woodland	Ground feeder, dense litter, low mobility	Declining
White browed babbler	Mallee shrubland	Ground feeder, structured habitat, low mobility, needs large remnant	Near threatened
Crested bellbird	Shrubland	Ground feeder, logs, low shrubs	Near threatened
Jacky winter	Open woodland	Ground feeder, needs large remnant	Declining
Red capped robin	Open woodland	Ground feeder, needs large remnant	Declining
Blue breasted fairy	Low shrubland/	Ground feeder, dense	Declining

wren	heathland	shrubs for nesting, needs large remnant	
Splendid fairy wren	Low shrubland/ heathland	Ground feeder, dense shrubs for nesting	Declining
Chestnut rumped thornbill	Woodland/mallee shrubland	Low shrubs, small tree hollows	Declining
White browed scrubwren	Woodland/shrubland	Ground feeder, dense shrubs for nesting	Declining
Western yellow robin	Woodland/shrubland	Dense thickets, needs large remnant	Declining
Rufous whistler	Woodland	Structured habitat, needs large remnant	Declining
White eared honeyeater	Woodland/shrubland	Dense low shrubs/grass for nesting, needs large remnant	Declining
Carnaby's cockatoo	Woodland/ heathland	Hollows for nesting, heath for feeding	Endangered
Western rosella (wheatbelt form)	Woodland	Nesting hollows	Near threatened
Mulga parrot	Woodland/mallee shrubland	Nesting hollows in branches near water	
Regent parrot	Open woodland	Nesting hollows in old wandoo or salmon gum	Declining

Many other declining bird species are only rarely encountered in the wheatbelt now and are therefore not very useful as key indicator species for fire management purposes. However, fire management should still consider the habitat requirements of these other declining species and not exacerbate the impacts of habitat fragmentation and predation. These species include Australian bustard, barking owl, shy heathwren, rufous fieldwren and crested shrike tit - all of which are listed as near threatened (Garnett and Crowley, 2000).

3.2.4 Invertebrates

In dry sclerophyll forest after fire, local species richness of the litter invertebrate community was reduced by one third. After 2-3 years post-fire, species richness increased due to young leaf-eating herbivores unique to the burnt areas. By late post-fire stage after 4 years, large decomposer species (cockroaches and earwigs) had higher richness than unburnt sites and pollinators (wasps and flies) had recovered to pre-fire richness (van Heurck and Abbott, 2003). In woodlands and shrublands near Kojonup and at Dryandra, Tutanning and Durakoppin, successional changes in invertebrate species following fires were related more to soil type, vegetation type and short term climatic conditions than time since fire. Most reviews conclude that invertebrates from mallee-heath shrubland are generally resilient to even large scale intense fires because they have evolved to survive the seasonal aridity and are pre-adapted to fire (van Heurck and Abbott, 2003). However, fire effects are more prolonged with increasing aridity due to slower plant growth and litter accumulation rates.

All scorpions found in the wheatbelt can survive a single fire. However, one scorpion (*Cerophonius michaelseni*) disappeared from its highly flammable habitat for at least 5 years following a hot autumn experimental fire (Smith, 1995). Given the extremely slow rate of dispersal by *C. michaelseni* back into burnt habitat, long inter-fire intervals are required for this species to recolonise the burnt areas (Lambeck, 1999). The trapdoor spider *Anidiops villosus* shows a similar response with no recruitment following a fire due to inadequate shade and litter, and possibly increased predation and reduced availability of prey (Main, 1995).

General recommendations for conserving invertebrate in the wheatbelt include using a mosaic of different vegetation age, incorporating fuel-reduced buffers that are burnt more frequently and long unburnt areas of greater than 20 years since fire, particularly in uncommon habitats such as granite outcrops and moist gullies. Small scale burn patches are recommended, particularly in small reserves and in habitats where fire sensitive relictual invertebrate species are known to occur. Fires during autumn are less likely to be detrimental as invertebrates are less active at this time. Plant growth and hence habitat resources are also more rapid in the winter and spring following an autumn burn, but variation in fire season would help to create a mosaic of fire age vegetation (van Heurck and Abbott, 2003).

3.3 Fungi

Fungi play an important role in biodiversity and ecosystem functioning, in terms of biomass decomposition, nutrient cycling, enhancing soil structure and nutrient uptake by plants and providing a food resource for some animals (Robinson and Bougher, 2003). Fire impacts significantly on the physical environment in which fungi persist, including the upper soil organic matter, leaf and twig litter and coarse woody debris. Fire can also destroy soil binding fungi leading to increased erosion of topsoil. In both burnt and unburnt forest the distribution and abundance of fruit bodies changed over time and species composition differed between regularly burnt and long-unburnt sites (Hilton *et al.,* 1989).

There is a recognisable succession of fungi following fire, and many of the fungi recorded in the first year following fire are unique to burnt sites. In eucalypt forests, saprophytic wood decay fungi colonize dead roots and logs and may fruit within several days. Alkaline conditions in ashbeds allow a specialist suite of fungi to fruit prolifically. Then as soil conditions revert and litter builds up, these fungi are replaced by species more commonly found in unburnt forests, including mycorrhizal fungi, which greatly increase nutrient uptake in eucalypts and many shrubs. In jarrah forest, mycorrhizal fungi that mainly inhabit the litter and organic soil layer are significantly reduced by fire, their recovery being strongly related to time since burning and litter build up (Malajczuk and Hingston, 1981). High intensity fires also cause scars on mature and young trees, that allow entry of wood decaying fungi, and weaken trees making them more susceptible to root and canker pathogens.

The occurrence of hypogeous (underground) fungi, on which woylies and other fauna depend as a food resource, is also related to litter depth, but the response to fire appears complex. Most hypogeous taxa have fleshy fruiting bodies close to the soil surface and are unlikely to be benefited by fire, whereas those that develop fruit bodies at depth are more likely to survive fire (Dell *et al.*, 1990). Following fire, species with fruiting bodies in the litter and organic layer are initially lost but then gradually increase in abundance between 4 and 10 years post-fire (Johnson, 1997). Within a week of a fire it is not uncommon to see fresh diggings by small animals for these fungi (Christensen, 1980; Robinson and Bougher, 2003). On the other hand, absence of fire may be related directly or indirectly to a decline in hypogeous fungi. Large areas of former woylie habitat at Dryandra Woodland were planted to brown mallet in the 1920's and protected from fire. These areas are now almost devoid of understorey with very little sign of diggings by woylies or southern brown bandicoot (Christensen, 1980).

The response of particular fungal species to fire age appears to be as variable as that of flowering plants. Fire can cause large reductions and changes in local species diversity but spatial and temporal separation of fires of different intensity can increase diversity across landscapes. Management of fire should aim for a mosaic of fire ages and intensities within communities and across landscapes to maximize and maintain fungal diversity (Robinson and Bougher, 2003).

4. ECOLOGICAL FIRE MANAGEMENT GUIDELINES

4.1 Determining appropriate fire interval ranges

The most challenging task for managers is to determine a fire regime that is appropriate for the likely array of fire sensitive species within a fire management unit. To determine appropriate fire interval ranges, vegetation type or fire management units need to be defined and mapped, and life attribute data is required for species within each unit. The fire interval range for each vegetation type is determined by the needs of flora and fauna most at risk of extinction from too frequent or too infrequent burning. In most cases, adequate life attribute data for all species is unavailable and so **Key Fire Response Species** can be used. These may include:

- Species that are common or structurally dominant
- Known bradysporous obligate seeder species
- Declared Rare Flora and priority flora
- Threatened fauna and their habitat species
- Fire sensitive fauna and their habitat species
- Fire sensitive invertebrate

These Guidelines have used the life attribute data approach for fire management which is a precautionary approach to avoid the possibility that populations of fire sensitive species may become locally extinct by frequent fires. A list of key fire response flora species and a list of known rare and priority flora species for each IBRA sub-region are provided in Appendix IV. It is accepted that cooler patchy burns that result in a fine grain mosaic do not kill all individual plants present in a population and this greatly reduces the risk of local extinction. Mosaic burning should be an accepted practice for all prescribed burns. However, until mosaic burning is adopted by all land managers, a precautionary approach should be taken.

The **minimum tolerable fire interval** is estimated as twice the longest juvenile period of species at risk of fire-related decline, ie species killed by fire but with seed that all germinates after one disturbance (Gill and Nicholls, 1989; Burrows and Wardell-Johnson, 2003). In fire sensitive habitats, this has been increased to 3-4 times the juvenile period for fire sensitive species (N. Burrows, personal communication).

The juvenile period is defined as the time taken for at least 50% of the population to reach flowering age. However, many species take many more years to reach peak flowering and maximum seed production. The primary juvenile period for any one species is likely to be longer in drier regions or in extended drought periods and cannot be assumed to be consistent across the Avon Basin. For some bird species (eg splendid wren), there is evidence that at least five reproductive seasons are needed following fire to ensure population viability (Keith *et al*., 2002).

The **maximum tolerable fire interval** for most species is not known, but is defined as the shortest time to senescence of fire response species that are short-lived and fire dependent, and which do not have a long lasting seed bank or means of vegetative reproduction. For fauna, this may be the time when critical habitat or foraging species begin to senesce or no longer provide adequate protection from predators. The senescence period may be defined for management purposes as the time when 50% of the population reaches senescence, or where 50% of the plant canopy has died, but there is little information about this in the literature.

The **appropriate fire interval range** for a particular vegetation type then is simply the time period between the minimum and maximum tolerable fire intervals. Given a fire interval range of say 15-50 years, fire managers then have flexibility to plan to burn anytime during this period with low risk of causing local extinctions of sensitive flora or fauna species. Some areas could be burnt every 15 years if they pose a high risk to human assets and other areas could be burnt every 50 years if they have a high weed potential, but it is the intention that each individual area experiences a variety of fire intervals within the tolerable range, fire intensity and season to maximise the biodiversity potential.

Many threatened and declining fauna species require longer fire intervals to allow regeneration of their habitat (ie protection from predators, shelter and replenishment of preferred food resources) and to accommodate their slow rates of dispersal into regenerating habitats. Therefore in many areas, the habitat requirements of threatened or fire sensitive fauna will need to be considered together with the requirements of threatened flora in determining appropriate fire intervals. The DEC draft fire management strategy for the Katanning district eastern reserves recommends a minimum fire interval of 30 years for all vegetation types, largely to provide suitable habitat for malleefowl (McCluskey *et al.* 2003).

In the Stirling Range National Park, 21 of the 27 threatened flora species are obligate seeders, five having canopy stored seed (all Proteaceae). One critically endangered species, *Dryandra montana*, does not flower until 9 years after fire and requires fire exclusion for at least 20 years to ensure survival of this species (Barrett *et al.*, 2004). A number of other species had primary juvenile periods of 6-7 years, suggesting a minimum fire interval of 12 years. The appropriate fire interval range, based on vital attributes for five eastern states bradysporous species, was somewhere between 13 and 30 years (Keith *et al*., 2002). Establishment of their seed bank took 6-13 years after fire, depending on site quality and rainfall, while senescence occurred between 30-50 years. These species represented only a small fraction of the total floristic diversity, but as larger shrubs, they contributed to structural diversity and their winter flowering provided nectar for honeyeaters and small mammals during a low resource time of year. Thus populations of these key fire response species, and many others, could be maintained by having fire frequency within the range of 13-30 years.

Woodland species are generally long lived, many living several hundred years before senescing. Brown mallet has a long juvenile period of 12 years and salmon gum woodland may take 300 years to begin senescence, so an appropriate fire interval for brown mallet-salmon gum woodland may be 24-300 years. Burrows *et al*., (1987) considered infrequent fires in the order of 20-60 years may be important for maintaining a diversity of successional stages in the Dryandra Woodland. The Fire Ecology Working Group (Tolhurst, 2002) estimated fire interval ranges of between 10 and 50 years for various Victorian woodland and forests, although their data was limited by the extent of fire history records. In south-east Queensland, a range of intervals between 7 and 25 years is recommended for shrubby eucalypt forests and woodlands (Watson, 2001). In general, Lambeck (1999) considered that fire should be used very infrequently as a management tool in this semi-arid environment and only where there is evidence that vegetation communities are senescing. In his opinion, a fire interval of no less than every 50 years is appropriate.

4.2 Ecological fire models

There are many **fire regime models** that predict fire interval distributions in ecological communities and which can be used to optimise diversity in age structure (Johnson and Van Wagner, 1984; McCarthy *et al*., 2001; Tolhurst and Friend, 2003). These mathematical models show that when an area is burnt randomly, the age class distribution (or time since last fire) of the vegetation in that area will approximate a negative exponential frequency distribution, ie there will be more patches of recently burnt young vegetation and few patches of long unburnt old vegetation (Fig 4.1).

Figure 4.1 Graph showing an ideal curve for fire frequency for acacia shrubland (VA1413) using a fire cycle of 30 years, a maximum tolerable fire interval of 60 years and a total area of 470,000ha.

The theoretical distribution for each vegetation type can be calculated from these models and then compared with the actual distribution (Fig. 4.2). This comparison shows where the actual distribution of age classes differs significantly from the ideal distribution for each vegetation unit, and areas in need of burning or protection are identified. The Fire Ecology Working Group (2004), in practice, attempts to maintain areas within ± 50% of the ideal distribution curve. These distributions are calculated using mathematical formulae and GIS technology which will not be considered in detail in these Guidelines. The Fire Ecology Working Group (2004) has produced an excellent manual to guide practitioners in these methods.

However, these frequency distribution models assume that all areas have been exposed to the same fire regimes, that areas are homogenous, that fire affects different patches randomly and that the fire regime has been constant. The shape of the distribution curve also depends on the flammability of the vegetation which can increase, reach a plateau or decrease over time. These assumptions cannot be met in real life situations, particularly in the highly fragmented areas of the Avon basin, but the theoretical distribution curves may provide useful broad guidelines for strategic fire management planning (Armstrong, 2004; Fire Ecology Working Group, 2004).

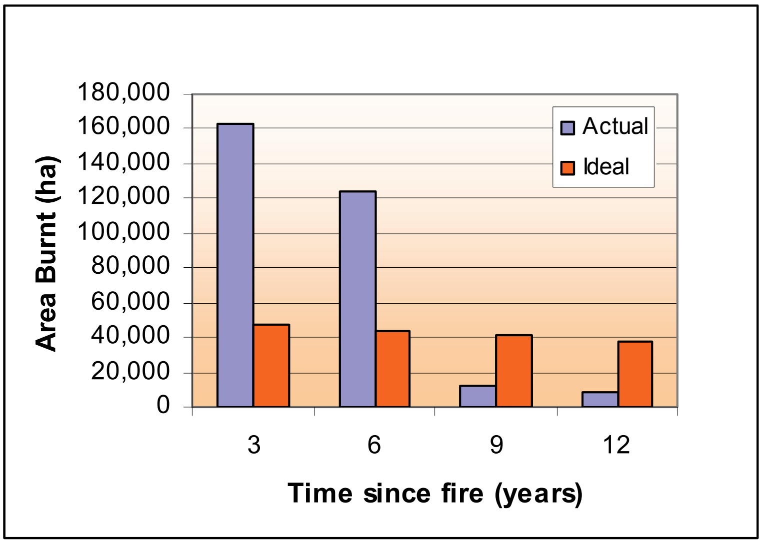


Figure 4.2 Graph showing ideal versus actual distribution of fire ages of acacia shrubland (VA1413) from fire scar data, and using a fire cycle of 30 years. Data taken from the actual area burnt each three years over the 11 year period (1994-2005).

Other simple models for setting desirable fire age distributions have been used, including a horizontal line distribution up to the optimal mean fire interval (eg Richardson *et al*., 1994). A hypothetical example of this may be seen in Fig. 4.3, where the minimum tolerable fire interval or juvenile period is 15 years and the maximum tolerable fire interval is 45 years. Areas above the horizontal line are over represented in three age classes and some areas in the 33yr and 48yr age classes may considered for burning. Over representation in the younger age class (12yr) is more difficult to manage as these areas should not be re-burnt until they reach the minimum tolerable fire interval and will remain over represented until that time. Figure 4.3 also show the negative exponential model for the same area as a comparison. This shows the decreasing age class distribution which decreases from 8% of total area in the first age class to 3% at 60 years. In the South African model, a constant proportion of total area of around 6.6% over 45 years is shown (or 5% over 60yr, or 10% over 30yr) and can be used as a desirable distribution over the expected mean fire interval. In this case, approximately 34% of the total area is in the juvenile stage, 25% in the mature stage and 40% in the senescent stage. Ideally, some of the older senescent vegetation should be burnt in around 3-4 years time.

The fire age distribution models depend on having life attribute data for key fire response species, vegetation and fire history mapping and flammability information. In most cases, this information is not available in the Avon Basin, but some estimation can be used to provide an initial approximation. Importantly, the models provide a framework on which to base ongoing data gathering, so that in future these models can be used to underpin strategic level fire planning processes. As more information becomes available, the models can be improved and adapted to local conditions.

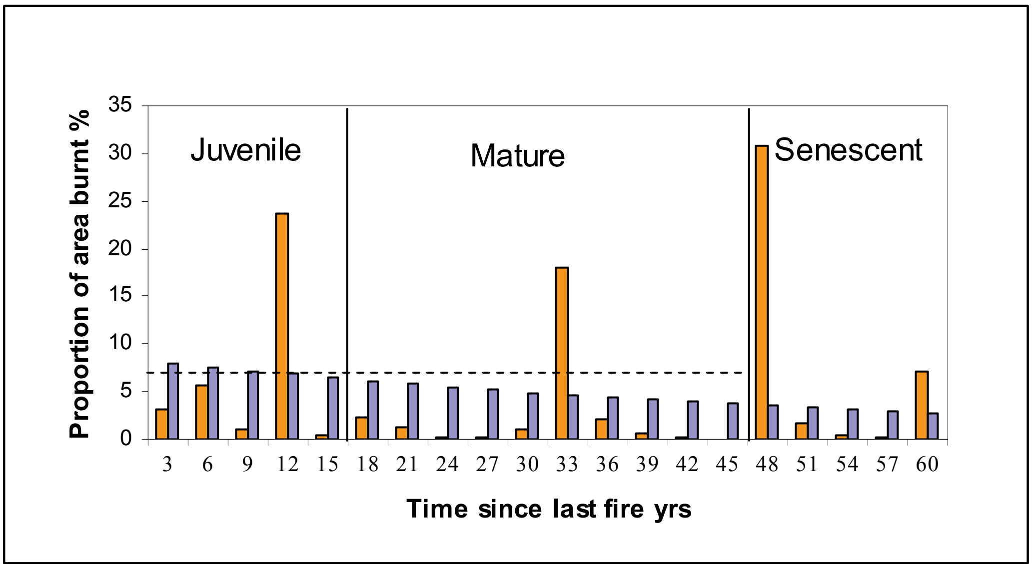


Figure 4.3 A hypothetical example (orange bars) showing distribution of fire age classes over 60 years (as proportion of total area) with a juvenile period of 15 years and senescence period after 45 years. The negative exponential model for that same area is shown (blue bars) while the South African model (dashed horizontal line) is shown as a constant value.

The **Fire Cycle** concept uses the minimum and maximum tolerable fire intervals of key fire response species and is a useful guide to work out what proportion of a vegetation type or fire management unit should be burnt each year (Tolhurst and Friend, 2003). The fire cycle for a vegetation type is the time period which is approximately the mean of the minimum and the maximum tolerable fire interval for that vegetation type. It is the period of time over which an area equivalent to the total area of the vegetation type is burnt (Ecological Working Group, 2004). This concept means that some patches within a vegetation type will remain unburnt while others may be burnt more than once within one fire cycle. For example, if there is 1,000ha in a vegetation type or fire management unit, then 1,000ha should be burnt within the fire cycle period. If the fire cycle is 30 years (ie mean of minimum fire interval of 10 years and maximum fire interval of 50 years – [(50+10)/2 = 30]), then the area to be burnt each year is the total area divided by the fire cycle, ie 1,000ha/30 years = 33.3ha per year, or about 100 ha over three years. Areas selected for burning in the 5 year Indicative Plan must have reached the minimum tolerable fire interval age.

When the fire cycle has been estimated, it can be used in the fire models to determine the area of each vegetation type that can be expected to be in each age class and so construct the theoretical age class distribution (Tolhurst and Friend, 2003). Age classes can be grouped into **early, intermediate and late seral stages** based on life attributes of key response species and observed successional changes within each vegetation type. For jarrah upland ecosystem, Burrows (in prep.) estimated that about 30% of the area should be in the early or juvenile stage (0-4 years after fire), 30% in the intermediate stage (5-12 years after fire) and about 40% in the late or senescent stage (13-34 years after fire). For the Katanning fire management strategy, it was suggested that at least 20% of each vegetation type should be in each of the early, intermediate and late seral stages (McCluskey *et al*., 2003).

Fire may be required in senescing fauna habitat earlier than that required simply to ensure the plant species remain in the community. For example, stands of *Gastrolobium* and *Acacia* may begin to senesce and thin out at about 25 years after fire but have hard coated seeds that could remain viable in the soil for another 30-50 years. However, these stands are most valuable during the first 20 years after fire when they are dense and protect threatened fauna (eg woylies, quokka, bandicoots, wallabies) from natural and introduced predators. Species that provide nectar and seed for fauna would also need to be regenerated before senescence to ensure the productive capacity of the plant community is maintained.

The post fire successional stages are not well defined in the literature, but can be recognised in the field by changes in plant species richness, understorey vegetation cover and height, total biomass and the proportion of dead vegetation over time. The transition from one seral stage to another is continuous but somewhat arbitrary, and the rate of change will depend on the severity of the fire and local edaphic and climatic conditions (Burrows, in prep.). Much work is needed to quantify these successional changes but simple observations and photographic references can be used as an initial approximation.

4.3 Obtaining fire regime variability

Given the range of fire responses found in plant, mammal, bird, invertebrate and fungal communities, the current thinking is that a diverse fire regime at the appropriate temporal and spatial scales is more likely to conserve biodiversity (Burrows and Abbott, 2003). Although the optimal range of patch size and fire age for fauna are largely unknown, smaller patches are considered to be better than larger patches, and a broad range of seral stages is better than a narrow range (Burrows and Abbott, 2003). Mosaic burning should be aimed for at the broad scale across the landscape and at the small scale within a vegetation community. Other fire management planning recognises the need to maintain mosaics at four spatial levels – regional, management zone, vegetation community and burn patch (Rose *et al*., 1999).

Fire frequency, intensity and season of burn are all inter-related and can be varied to produce the desired patchiness of fire age classes and structural diversity at a local or landscape scale. Areas of long unburnt vegetation with higher fuel loads tend to burn at greater intensity and leave fewer unburnt pockets than frequently burnt areas, while summer and early autumn burns are generally hotter and more complete than spring burns. Intense fires will often scorch or consume all shrub and tree canopies, whereas cooler fires may only scorch some of the understorey, providing vertical and structural diversity in fire regime. The scale of patchiness achieved will often be determined by the existing heterogeneity of the landscape, soil types and vegetation. Patchiness is also derived from changes in wind direction and weather conditions during the passage of a fire.

A fire mosaic should be aimed for both within a fire management unit, and between units of the same vegetation association or fauna habitat. To achieve a **spatial mosaic** for a vegetation unit, different regimes of frequency, intensity and season need to be applied within each unit. In addition, for any one area, different regimes should be applied over time to achieve a **temporal mosaic**. For invertebrates, the scale of patchiness may be at the micro-scale, measured in metres, while for macropods, patches of several hundred hectares may be adequate. The scale of patchiness desired in a fire management unit has to be decided at the local level, often with knowledge of the habitat requirements of specific fire sensitive species in that area (Williams *et al.,* 1994).

There are several ways to achieve a spatial mosaic of burnt and unburnt patches within a fire management unit:

- Use artificial fire breaks (disturbed earth, buffer burn, slashed)
- Use moisture differentials to protect sensitive vegetation in moist areas
- Use natural barriers (rock outcrops, creeklines, low flammability vegetation)
- Use more frequent burns to avoid high fuel build up and intense fires
- Use early spring or late autumn burns when fuels are not uniformly dry
- Use weather conditions on the day of burning to vary fire intensity and spread
- During mop-up operations leave areas unburnt that do not pose a threat to safety
- Vary the spacing and timing of ignition points

In **heterogeneous** vegetation landscapes, a mosaic is more readily achieved using natural barriers, differences in flammability among species and moisture differentials. Areas with more structural diversity and variation in canopy cover, floristics and litter cover will burn with variable intensity and can produce patches of burnt, partially burnt and unburnt vegetation over a few hundred meters (Atkins and Hobbs, 1995). Considerable patchiness was observed following two prescribed autumn burns in dense shrubland and low open heath (1988 and 1989) at Durokoppin Nature Reserve in the central wheatbelt (Atkins and Hobbs, 1995). Fire intensity varied with density of the vegetation and continuity of ground cover and with local variations in wind speed. A more complete burn resulted when wind speeds were more uniform and in the 10-20km/hr range, than when wind speed was variable and were 0-15km/hr, but even so some unburnt and partially burnt patches remained. This variability in fire intensity was related to post-fire regeneration of seedlings and was considered to be an important factor in the co-existence of shrub species in species-rich heath communities (Atkins and Hobbs, 1995).

In more **homogenous** vegetation landscapes, existing firebreaks can be used, or new firebreaks constructed, to initially break up the vegetation into appropriate patch sizes. Wind driven buffer burns have been trialled in more continuous vegetation areas and this method provides both spatial and temporal variability as the location of the burn strips can be altered over successive burns. These buffer burns are designed to prevent large areas being burnt in a wildfire by having strips of younger vegetation at right angles to the prevailing wind direction. However, they can also be designed to break up uniform vegetation to create smaller scale mosaics. Fire behaviour in successive burns will then be more variable and likely to perpetuate a fire mosaic.

At the landscape scale, patchiness should be obtained by varying the year in which nearby or adjoining management units or reserves are burnt. Within isolated reserves and remnants, it is recommended that no more than 30% of the area is burnt at any one time. This allows fauna to be protected in unburnt areas and later recolonise burnt areas. Larger areas of native vegetation should be broken up into manageable blocks of say 5,000 – 10,000 ha using firebreaks, scrub rolling or wind driven buffers, and adjoining blocks burnt using different fire regimes of fire season, intensity and frequency.

The effect of **fire season** for each successive fire needs to be considered in different vegetation types. In the northern sandplain shrublands, season of burn had a considerable impact on the re-establishment of obligate seeder species (Bell *et al*., 1987; Enright and Lamont, 1989). Seedling regeneration is most effective after autumn than spring burns, as seedlings can establish during the wetter winter months and have a deeper root systems going into their first summer drought. Following a spring burn, species which release seed after fire, but which don't germinate until opening rains, suffer greater levels of seed predation and lose viability during summer. Spring burning favours resprouter species as they already have an established root system. Autumn burning is less favourable for resprouter species as new growth is out of phase with their normal summer growth cycle. Spring burning is usually less complete and leaves more unburnt litter which can act as favourable microsites for seedling germination (Enright and Lamont, 1989).

The maintenance of biodiversity requires variability in fire regime, as discussed above. Variability in disturbance allows ecosystems to fluctuate between alternative states or seral stages, without reaching equilibrium where one group of species dominates over long periods of time and others decline to extinction (Keith *et al*., 2002). Attaining spatial variability requires planning to ensure that some long unburnt areas are maintained. This is particularly important for animal diversity, but the actual proportion of habitat required to be in the early, mid and late seral stages is unknown for many species. Simply creating a mosaic patchwork across a landscape without regard for the particular habitat requirements of threatened or declining species is ill-conceived.

4.4 Allowing for no planned burn areas

It is important to plan for areas that are to remain unburnt, or to attempt to exclude wildfire, in each vegetation unit as scientific reference areas. These should include representative areas of important vegetation types and are not just those which naturally have low flammability, or occur on rock outcrops, breakaways, salt flats and wetlands. Fire ecology studies have been strongly biased towards the early successional stage, and little is known about the structure, floristics and fire behaviour of vegetation that has developed in the absence of fire (Muir, 1985). Long unburnt communities provide opportunities to monitor the onset of senescence and patterns of recruitment before and after senescence. For example, a 63 year old stand of *Allocasuarina huegeliana* near Quairading was found to reproduce successfully without fire for 63 years and that, although the majority of plants died in their early years, a small number continued to be productive through to senescence (Muir, 1985). The age structure and openness of these old stands is characteristic of the late seral stage of which there are few examples.

Some individual plants can reach remarkable heights when left unburnt for long periods, compared with average heights in more frequently burnt areas. For example, in the WA wheatbelt, maximum height of *Banksia ashbyi* was 11m compared with and average height of 6m, *Melaleuca uncinata* was 12m compared with 2.8m, and *Eremaea pauciflora* was 4m compared with 1.5m (Muir, 1985). Muir (1985) suggests that these maximum heights give insight into the possible appearance of the wheatbelt bushland prior to clearing and burning by Europeans.

Similarly, Hopkins (1985) noted that the eucalypt mallee form in *Eucalyptus cylindrifolia, E. diptera* and *E. eremophila* was induced following a fire in 1938. In unburnt patches, these species formed woodland to 7m tall with single stemmed trees, whereas the 40 year old regrowth formed heathland with emergent multi-stemmed mallees up to 3m tall. He considered that it was possible that in the absence of further fire, these mallees could develop into the single stemmed tree habit. The mallee form may therefore be a mid seral stage for many species, with few examples left of the single stemmed late seral stage. Furthermore, the dense 40 year old regrowth mallee heath may be more prone to carry fire, causing the vegetation to remain in the mid-seral stage. Beard (1973) also made comments regarding *E. oleosa – E. floctoniae* woodland being mostly burnt and reduced to mallee. Fire in these communities did not change the floristics of the vegetation but lead to significant long term structural changes. These observations point to the need to preserve representative areas of long unburnt vegetation so that the complete life cycle of long lived species can be studied.

Many smaller reserves in the wheatbelt have been left unburnt as they are not considered a major fire danger to local landholders compared with larger reserves (Muir, 1985). Some have also been protected from fire due to the presence of threatened flora and to avoid the risk that burning may facilitate other threatening processes such as weed invasion. These isolated reserves are less likely to be ignited by lightning, being smaller targets and surrounded by well protected agricultural land (McCluskey *et al*., 2003), and may therefore be useful areas to leave as no planned burn areas. A further benefit of this practice is that it will reduce the likely impact of weed invasion. However, smaller reserves often burnt out quickly, when ignited by accidental fires from agricultural sources, as the fire passes through the reserve before fire trucks arrive. Long term protection of these smaller reserves is therefore not guaranteed, but this should not deter fire managers from accepting this level of risk and aiming to maintain a proportion of these small reserves as no planned burn areas.

No planned burn areas have been incorporated into several fire management plans, for example the Stirling Range (Barrett *et al*., 2004), the Perup Fauna Priority Area (Christensen and Maisey, 1987) and the Western Ground Parrot Interim Recovery Plan (Burbidge *et al*., 1997), to specifically protect threatened ecological communities or fauna habitat areas from fire. Aging vegetation does lose productivity which reduces food availability, causing some bird and animal species to move out of long unburnt vegetation to seek alternative foraging areas. However, the older vegetation may still be used for breeding, providing denser cover and protection from predators. For Western Ground parrots, no planned burn areas are recommended for all known populations (Burbidge *et al*., 1997). A number of other flora and fauna species are found only in long unburnt vegetation as discussed earlier, and although these areas may burn more intensely than more frequently burnt areas, the habitat advantages for these species in long unburnt vegetation may outweigh the disadvantages of occasional intense fire.

The actual area required to be protected as no planned burn areas will depend on the management objective, whether it be for protecting threatened flora or fauna habitat, or for research into late seral stage of various vegetation communities. The area required for the former purpose may be determined by population viability analysis. The area required as scientific reference areas could be quite small (25 – 500ha), and it would be a lower risk strategy to have a number of well spaced, smaller reference areas rather than few larger areas to avoid having the whole area burnt out in a single wildfire event. The main requirement is for each vegetation community to be represented, and where possible, located in different rainfall regions. There are likely to be many more patches of long unburnt vegetation left by chance in reserves and remnant vegetation and these areas need to be recognised for their scientific value and protected where appropriate.

4.5 Allowing for wildfire

Native vegetation in the wheatbelt is under much greater threat from wildfire burning into reserves and remnants from farmland and roadsides than vice versa (Department of CALM, 1995; McCluskey *et al*., 2003). Arson and accidental fires constitute a greater threat to native vegetation in wheatbelt reserves than lightning strike. Community concern about fires originating from native vegetation in the wheatbelt should therefore be considerably less than in forested regions of the south-west. In fact, there has been concern about the lack of fire in many reserves in the wheatbelt and the potential loss of fire dependent species due to ongoing senescence and lack of seed germination (Yates and Broadhurst, 2002).

However, in the eastern portion of the Avon basin, where there are large tracts of unallocated crown land and pastoral and mining leases with continuous native vegetation, wildfire in heath and shrubland can extend for days or weeks and cover over 100,000ha in a single event. Under equivalent burning conditions, fires in mallee-heath can spread at much faster rates (3-6 times) than in eucalypt forest (McCaw, 1998). There is little community concern about wildfire in these areas, fewer high value assets in terms of people, housing and infrastructure, few access tracks and very little access to heavy machinery and fire fighting resources (Daniel, 2003). Consequently, wildfires are often left to burn out unchecked, with considerable potential for ecological damage.

Fire managers have to consider unplanned as well as planned fires when managing the ecological needs of vegetation. For example, in Victoria it is known that on average 2% (115,500ha) of all public land is burnt by wildfire each year and a further 3% (200,000ha) is burnt by prescribed fire (Fire Ecology Working Group, 2003). Some wildfire will occur at less than the minimum tolerable fire interval for a vegetation community and management has to focus on minimising the impact of these events. Wildfire in other vegetation types may be too infrequent and occur outside the maximum tolerable fire interval, so managers need to plan to introduce fire. The total area burnt in the Avon Basin since 1994 is shown in Fig 4.3 and includes all wildfire and areas of prescribed burning that were captured using Landsat imagery. This demonstrates the variability in wildfire extent from about 4,000ha to nearly 320,000 per year, most of the large areas burnt being in single wildfires in the Coolgardie bioregion. The average area burnt over this 11 year period was 81,679ha which is 2.8% of the total area of remnant vegetation in the Avon Basin.

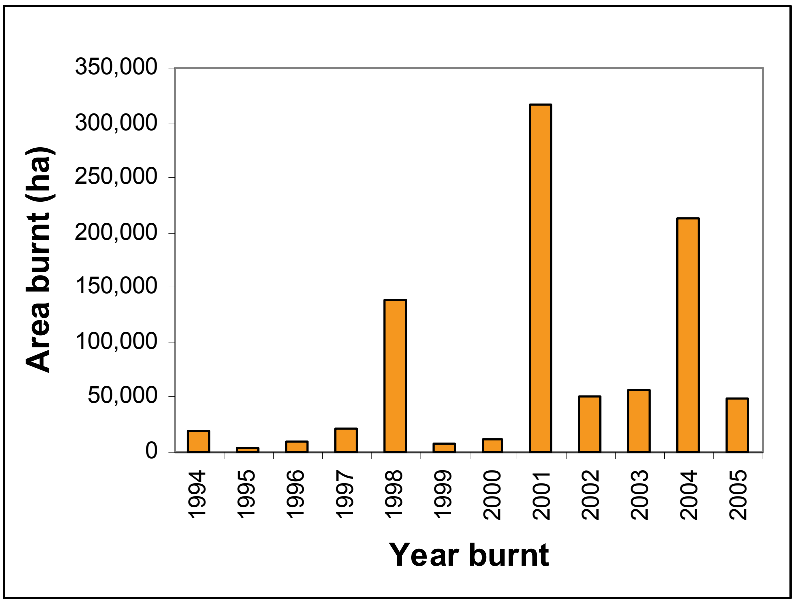


Figure 4.3 Total area of all remnant vegetation burnt by wildfire and prescribed burns in the Avon Basin using data from Landsat imagery (Fire Scar database) over the 11 year period 1994 - 2005. Some areas may have been burnt more than once.

Fire history records, where available, can be used to assess the past frequency and ignition source of wildfires. Although wildfires cannot be predicted, it is important to include an estimate of future wildfire occurrence in different vegetation types or fire management units. The Wildfire Threat Analysis will help in this regard and will show areas prone to more frequent fires (Mueller, 2007 in preparation). Annual or three yearly averages of wildfire events, in terms of area burnt and percent of total area of vegetation type, can be calculated including years with no wildfire events (Fire Ecology Working Group, 2003). This estimate is then deducted from the area planned to be burnt in the Indicative 5 year Plan for that vegetation type. This may mean that some areas nominated for prescribed burning may be removed from the 5 year plan if there is found to be an adequate proportion of affected vegetation types within each seral stage following a wildfire.

An example of this calculation is if there is 10,000ha of mallee heath with a fire cycle of 40 years, then the planned burn area is 250ha per year, or 1,250ha every 5 years. If it is known that wildfire affects about 100ha per year on average, then only 750ha may be nominated to be burnt in a 5 year planning period. However, this calculation should be regarded as a guide only as the actual area burnt each year can vary enormously (Fig. 4.3).

In the Avon Basin there are small and large areas of eucalypt woodland with little understorey and it may be prudent to leave some of these areas unburnt, other than when they experience wildfire. The appropriate fire interval for these woodlands can be up to 200 years and during this time it is likely that most areas will be subject to a regenerating disturbance, whether it be fire, storm, flood or drought. Often these areas will not carry fire other than after a period of unusually high rainfall or flooding which stimulates growth of understorey grasses, herbs and shrubs.

Allowance should be made in predicting the risk of wildfire occurrence due to changes in vegetation cover, farming practices and demographics in the last 30 years. With dramatic decreases in the amount of native vegetation cover in the wheatbelt in the last 60-80 years, there is a much lower likelihood that a patch of bushland will be affected by lightning induced wildfire due to the lack of continuity of canopy cover. From 1945 to1970, there was extensive clearing of vegetation for agriculture and associated burning which resulted in many fires escaping into nearby reserves and remnants. Considerably fewer wildfires have been recorded since the 1970s following a reduction in land clearing (Department of CALM, 1995). During that time, farming practices in many areas have also changed from stubble burning to minimum tillage farming systems, and combined with a lower risk of ignition from modern farm machinery compared with 50 years ago, there should be a much reduced incidence of fire escape from farmland. Furthermore, advanced on-farm, local government and agency fire fighting capacity is able to rapidly respond to minimise the spread of wildfire. On the other hand, there are more recreational visitors to the region and a greater incidence of arson generally, which has increased the likelihood of wildfire in some areas. Long term averages of wildfire events and areas affected could therefore be quite misleading, and it is recommended that only the last 30 years be used for this calculation.

4.6 Allowing for drought

Prescribed burns should not be implemented in low rainfall regions if the area is suffering from prolonged drought as this will seriously diminish the likelihood of successful regeneration. Given the unpredictability of both unseasonal drought and high summer rainfall in this environment, a great deal of caution is required when planning fire regimes. One management strategy to minimise this risk is to burn a number of smaller areas over several years rather than one large burn in one year to provide more opportunities for recruitment during the productive life of mature plants. This would avoid the risk of widespread seedling failure should there be a serious drought following a prescribed burn. Another strategy would be to only burn areas if the previous year's rainfall for the area was at least 80% of annual average rainfall to ensure that the soil profile was not unduly dry. Longer term weather forecasting, particularly of El Ninõ events, should be consulted during the fire planning stage.

Seedling mortality is characteristically high in many species during the first summer drought and is typically density dependent (Enright and Lamont, 1989; Burrows *et al*., 1990; Yates *et al*., 1994; Yates and Broadhurst, 2002). Yates and Broadhurst

(2002) found seedling mortality of *Acacia* seedlings during the first summer following an autumn burn was nearly 70%. Enright and Lamont (1989) found seedling mortality rates in *Banksia* seedlings over the first summer to be considerably greater in sites burnt in the previous autumn (76%) than in spring (28%), due to the higher initial density of seedlings from the autumn burnt sites. Competition by seedlings for soil moisture and nutrients in the first summer when root systems are being established can lead to self thinning and significant mortality. This will be accentuated if germination is followed by a period of extended drought. Prescribed burning may result in considerable adult mortality without successful seedling recruitment which could lead to local extinctions (Yates *et al*., 1994).

In semi-arid environments, water and nutrient resources are likely to limit growth of seedlings to a greater extent than competition for light. Seedling recruitment under *Eucalyptus salmonophloia* and *E. wandoo* is rare unless the adult tree canopy is removed, either by fire, storm damage, drought or logging (Burrows *et al*., 1990; Yates *et al*., 1994). *E. wandoo* germinated after experimental fire almost exclusively in ashbeds which provided a number of advantages for seedling growth (Burrows *et al*., 1990). In valleys, where clay loam soils become hard and crusted in summer and waterlogged in winter, the heat treatment caused the surface soil to become friable and probably improved soil drainage. Seedlings growing in valley ashbeds were taller with greater biomass than seedlings growing in mid-slope ashbeds. On the drier mid slopes, seedling mortality was high and growth was poor due to poor root development and drought death.

In the goldfields, Yates *et al*. (1994) found that 52% of *Eucalyptus salmonophloia* seedlings from a burnt site and 99% from a storm-affected site died during their first summer. The lower seedling mortality at the burnt site was related to summer rainfall (Dec-Feb) which was three times the long term average (178mm), whereas summer rainfall at the storm affected site was considerably less and below average (51mm). This suggested that seedling recruitment following disturbance may not always occur and may depend on a succession of above average rainfall years following the disturbance. Seedlings also established well in a flooded site where adult trees did not suffer canopy removal. Post-fire seedling recruitment of mallee *Eucalyptus* spp. has been shown to be successful in only 10% of fires over a 15 year period (Wellington and Noble, 1985). Enright and Lamont (1992) considered that the coincidence of fire and above annual average rainfall years suitable for establishment of some populations of *Banksia attenuata* may only occur 3 times in the 300 year lifetime of an individual.

Flowering and seed production is also affected by drought and so there will be fewer seeds for recruitment if fire is planned following a period of drought. Yates and Broadhurst (2002) found that flowering, fruiting and seed production in two rare *Acacia* spp. were significantly less in 2000 when rainfall was well below average (255mm) compared with the previous year when rainfall was well above average (595mm).

Drought may also stimulate regeneration and there are many similarities in the response of heath vegetation to both fire and drought (Hnatiuk and Hopkins, 1980). However, in the goldfields woodland study, there were fewer seedlings and trees in the 0-10cm size class of *E. salmonophloia* in drought and severe storm affected woodlands than in fire and flood affected woodlands (Yates *et al*., 1994).

4.7 Community protection burns

Pre-suppression activities in native vegetation are undertaken to minimise the severity of wildfire and are part of community and asset protection plans. Activities include liaison with other agencies and landholders, construction and maintenance of firebreaks, access tracks and water points, and some prescribed burning and buffer burns to reduce fuel loads in high risk areas, such as near townsites or as strategic breaks in large areas of vegetation. Prescribed burning on government land in the wheatbelt region has been minimal in the past, mainly due to a lack of resources and low incidence of wildfire, but also to lack of knowledge about the possible negative impact that fire may have on the many fragmented populations of threatened flora and fauna in the region.

Most of the wheatbelt region has little in common in this regard with the south west regions dominated by jarrah forests where regular prescribed burning has been practised since the 1960s, principally for community protection but also to protect the high value timber resource from the devastating impacts of wildfire as experienced earlier in the century. Fire frequency was based on attempting to maintain forest litter fuel levels below 8 - 10tn/ha, which in turn should reduce the intensity of wildfire to less than 1,000kW/m under most weather conditions and make wildfire suppression safer and more effective. Fire intensity greater than 3,000kW/m leads to complete crown scorch in most forests, including low open woodland, and direct attack to suppress the fire is not possible (Mueller, 2001). However, the rate of litter accumulation in the lower rainfall wheatbelt vegetation is considerably slower and often plateaus at fuel loads well below those in jarrah forest (Burrows, 1985; Bradstock and Cohn, 2002). Burrows (1985) estimated that wandoo woodland with a basal area of 10m^2/ha only reached a fuel litter load of 8tn/ha after 30 years. On the other hand, wheatbelt soils are drier for a longer period than south-west forest soils and this has a large bearing on the flammability of the vegetation and its ability to carry fire (McCaw, 1998; Bradstock and Cohn, 2002).

Any prescribed burning for community or asset protection in the Avon Basin should be considered within the context of ecological burning and conform to the same approval processes and requirements for appropriate fire regimes. This may mean that areas close to townsites and high value assets are burnt at the lower end of the appropriate fire interval range. These areas are included in the assessment of spatial variation in fire frequency for those vegetation types. For example, if a vegetation type close to a town has an appropriate fire interval range of 10-35 years then it could be burnt every 10 years to minimise wildfire risk and intensity. However, this practice does not conform to the requirement for temporal variation in fire frequency for a particular area (see section 4.4). To counter this, the season and intensity of these community protection burns should be varied and a patchy mosaic maintained within the vegetation unit.

5. PLANNING FRAMEWORK, PRIORITIES AND PROCESSES

5.1 Planning guidelines

It is assumed for these Guidelines that there is community acceptance of the 12 principles above, and that all wildfires and prescribed burns have an ecological outcome. It follows that all wildfire suppression activities and prescribed burns should adequately consider the ecological values of an area, including burns prescribed for community protection.

A workable planning framework should contain the following elements (Keith *et al*., 2002):

A. A means of reducing complexity of biodiversity conservation to simple and measurable goals;
B. Flexibility to deal with stochastic environments and uncertain knowledge;
C. A means of resolving conflicts to meet multiple management goals;
D. A means of assessing performance, obtaining new knowledge and incorporating this new knowledge into management practice.

A. These Guidelines provide a framework for managing fire, based on the life attributes of key species in each broad vegetation classes. The Beard-Hopkins pre-European vegetation mapping provides a basis for grouping broad vegetation types according to their likely fire regime requirements so that areas can be prioritised for fire management. These regimes are based on the tolerable fire intervals of structural, threatened and fire sensitive species within each vegetation class. The strategic goal of maintaining approximately 30% of each broad vegetation type in the early, mid and late seral stage is measurable and simplifies the complexity of trying to manage every species or community according to their individual requirements. These goals can be measured using remote sensing techniques once these systems are in place.

B. Allowance for unplanned wildfire events must be built into the strategic planning, with predictions of annual wildfire based on the Wildfire Threat analysis. Area, month and intensity of all wildfire events must be included in the fire history database, and fire frequency distributions of each affected vegetation association updated. Severe weather conditions can be unpredictable and there should be flexibility in the scheduling in the 5 year plan. Unpredictable weather conditions can also affect post-fire recovery. At all levels, there is uncertainty in our understanding of flora and fauna fire responses and fire behaviour that needs to be openly acknowledged. Prescribing fire regimes purely on the basis of known fire responses for one or two species is likely to be detrimental to other species. However, we cannot wait until we have perfect knowledge and must therefore proceed with caution and using an adaptive management approach.

C. Often there are competing conservation goals within a reserve as well as conflicting goals between conservation, production and community protection. Compounding these difficulties is the general lack of understanding about the ecological role of fire regimes in the community and the fear of change. The Guidelines are intended to address some of this lack of understanding and provide a structure for assessing the conservation values for each area so that priorities can be determined. Stating clear ecological objectives for any proposed burn and communicating these objectives to the wider community will also significantly reduce any misunderstandings. A management structure involving people responsible for managing fire across all tenures in the Avon Basin should help to diffuse conflict and the fear of change. An annual forum to review fire management and share new knowledge is strongly recommended. It is also intended that a community education program will follow these Guidelines which should improve the understanding of ecological fire regimes.

D. Each prescribed burn and wildfire should be viewed as an opportunity to obtain knowledge about fire responses of species and communities and about fire behaviour. It is incumbent upon fire managers that clear objectives are set for each prescribed burn and that these objectives are measured in a way that can improve our knowledge and practices. Monitoring and recording fire responses and behaviour can and should be implemented at the species, community and landscape level. A vast array of monitoring techniques is available, from simply counting recruitment seedlings in a plot, to interpreting Landsat imagery to map fire intensity across vegetation types and topography. Greater emphasis needs to be placed on ensuring that those people researching the effects of fire are suitably skilled or trained in experimental design and data analysis to increase the usefulness of the data collected. Monitoring always requires additional resources, but unless the outcomes are assessed, we may cause further species decline through ignorance and neglect.

5.2 Levels, objectives and responsibilities of fire management planning

It is recommended that three levels of fire management planning are developed so that areas are nominated or selected for prescribed burning according to a predetermined set of priorities, rather than on an ad hoc basis.

- Strategic biodiversity conservation planning
- District indicative 5 year planning
- Individual burn prescription planning

This will require new management structures being developed with designated responsibilities and communication processes across all land management agencies, including DEC, FESA, local government, and volunteer bushfire brigades. It is strongly recommended that other community representatives be involved at the strategic level of planning, and that strategic and 5 year plans be presented to community forums before finalisation.

[A] Strategic fire planning

This level of planning operates at the regional and bioregional scale. The aim is to set overall ecological fire management strategies, priorities and objectives for the whole of the Avon Basin. Information in the corporate databases is used at this stage. It is envisaged that a strategic fire planning team will need to be assembled and comprise regional fire control officers in DEC, FESA and local government, a dedicated GIS fire planner and community representatives eg Avon Catchment Council, Land for Wildlife landowners and environment groups.

Strategic objectives:

1. Ensure all vegetation prescribed to be burnt is within the appropriate tolerable fire interval ranges for vegetation associations present within those areas.

2. Ensure that prescribed burns incorporate variability in season, intensity and scale of patchiness of burn to maintain all biological components in the environment.

3. A strategic objective of maintaining at least 30% of each broad vegetation class and vegetation association across the Avon Basin in the early, intermediate and late seral stage is desirable to achieve an appropriate distribution of fire ages.

4. Ensure that no more than 30% of the total area of any known threatened flora population, threatened fauna habitat, or threatened ecological community is planned to be burnt in a single prescribed burn operation.

These objectives can be measured from fire history data using GIS technology and are surrogates for the ultimate goal of retaining species diversity (Possingham, 2001). Fire history needs to be compiled for each vegetation class and vegetation association, all threatened flora populations, threatened fauna habitats and threatened ecological communities in the Avon Basin and compared with an agreed optimal spread of fire age distribution for these communities. However, there is currently little information available to define the endpoints for each seral stage, based on life attributes for key fire response species and there is no agreed optimal distribution of fire ages for different communities in the Avon Basin (see Section 4.2). In addition, as prescribed burning becomes more complex, and mosaic burning is achieved, then some measure of fire patchiness will need to be captured in the fire history statistics.

It is envisaged that strategic planning will be responsible for:
1. collating and updating life history attribute data from monitoring and scientific data;
2. collating and updating fire history data, including fire season and a measure of fire intensity and patchiness;
3. designating fire management units or polygons in the GIS that can be managed by district fire crews in a reasonable time period;
4. developing and maintaining a Master Burn Database that has fire management attributes (fire history and regime) for all fire management units (see Section 5.5);
5. maintaining the GIS databases and project layers to be available for all fire managers in the region;
6. ensuring that an appropriate range of fire age classes, fire seasons and fire intensities are maintained for each broad vegetation class and vegetation association;
7. ensuring that an appropriate range of fire age classes and seasons is maintained for threatened populations and communities;
8. nominating areas to be left unburnt for reference areas and ensuring these areas are not included in the burn plans;
9. using the prioritisation process to select areas with high priority for burning and ensuring there are sufficient resources to carry out these burns;
10. allocating burns selected from the prioritisation process to district fire control officers in DEC, FESA and local government;
11. developing a strategic monitoring program to ensure fire behaviour and fire responses in all vegetation associations and in threatened flora and fauna populations are measured;
12. ensuring that burn objectives and ecological objectives are being monitored and that results are collated, analysed and audited;

13. inviting interested stakeholders to be involved in the strategic planning process; and
14. ensuring the fire management program is reviewed annually and results of monitoring and trials are shared with all stakeholders.

Other considerations:
An important strategic consideration for fire managers in the Avon Basin has been how to manage fire in different sized reserves. McCluskey *et al.,* (2003) recommended different levels of prescribed burning, wildfire suppression and use of firebreaks according to the size of the reserve in the Katanning district eastern reserves. However, it is this author's view that prescribed burning should be based primarily on the biological needs of the flora and fauna present, rather than the size of the reserve, and appropriate means to effectively carry out prescribed burning need to be developed for different situations.

This being said, areas of continuous vegetation in the Coolgardie sub-region, will need to be broken up into fire management units of about 5,000 – 10,000ha for prescribed burning operations using vegetation association boundaries wherever possible. This will allow a regional level mosaic to be developed in this area over time where adjoining units are burnt at different times. Wind driven buffers, scrub rolling and graded tracks may be used to break up this vegetation into these management units, and to reduce the impact of wildfire events (see Section 8). The advantage of wind driven buffers is that they can be applied in different places over time and increase the amount of patchiness for the area. Other areas with isolated reserves and remnant vegetation in the wheatbelt may be clustered together into landscape fire management units around more significant Nature Reserves.

The ArcGIS database compiled for this project contains data from the DEC corporate datasets clipped to the Avon Basin and IBRA sub-regions for strategic planning. Vegetation mapping is at a broad scale (1:250,000) and boundaries are likely to be inaccurate on the ground, but this data provides useful strategic planning units. Basic fire history data is available from 1993 to 2005, and earlier for particular reserves. However, detailed fire history data for the whole of the Avon Basin is urgently needed to determine the spread or frequency of fire age classes for any vegetation unit or reserve. When this data becomes available, it will underpin this important analysis stage which sets the framework for the ecological fire management strategy.

The frequency distribution graph of fire ages will show whether there is an appropriate balance of areas in each vegetation type in the early, intermediate and late seral stages. If there is a disproportionate area in the intermediate or late seral stages then some of this area can be selected and nominated downwards for the 5 year indicative planning stage, according to the prioritisation process. Representative areas to be left long unburnt and fire sensitive communities needing special protection from fire are identified at this regional level.

Prioritization of burns is required at this stage and this process is explained in Section 5.3. The primary prioritization is based on the frequency distribution of fire ages (or time since fire) of each vegetation class and vegetation association across all land tenures contained within the Avon Basin. Other levels of prioritization (eg conservation values, viability) are then imposed to select a subset of areas in greatest need of regeneration by fire. At this stage there is no corporate or regional database which can be used to assess viability of reserves, other than salinity risk. This assessment will need to be done at the district level and then fed back up to the strategic planning level. It is hoped that all reserves can be assessed for viability over time to reduce the time delay in the prioritisation process.

Nature Conservation officers, flora officers and other land managers may also nominate specific areas upwards to be considered at the strategic planning level that have been identified as needing regeneration, such as threatened fauna habitat, DRF and priority flora populations or threatened or special value ecological communities. These burns may be part of a species or threatened ecological community recovery plan, an area management plan where available (eg Katanning eastern reserves), or a research trial with specific ecological burn objectives. A list of all known DRF and priority flora in each of the IBRA sub-regions is provided in Appendix V along with known or inferred fire responses to assist with planning.

Other burns nominated by land managers, including local government and private landowners, may be directed to local DEC staff to assess the conservation value of the area concerned, and whether the proposed burn fits into the frequency distribution of fuel ages for that vegetation type. All burns could be captured into the fire history database, but there is no obligation of private landowners to comply with the strategic management planning processes.

Fire management includes wildfire suppression as well as prescribed burning for community protection and ecological outcomes. Expanding the level of fire management in the Avon Basin will require additional DEC resources for fire planning and operations. Currently, resources from FESA and the local government voluntary bushfire brigades are called on to suppress wildfires and conduct some community protection burns, and it is hoped that these resources may also be available for some ecological fire management activities. Some training, education and capacity building within communities will be required to meet the need for ecological fire management. Assessing the requirement, availability and capacity of these additional resources will be critical at this strategic planning stage to determine the total area and type of burns that can be realistically managed in the given time frame.

[B] District indicative 5 year plan

This level of planning is at the district and local government level, where burn areas nominated, prioritized and approved from the strategic planning stage are allocated to the respective DEC and FESA district fire managers for further assessment. Areas assessed as having low conservation value or low priority at the strategic or district planning levels may be allocated to the local government Chief Fire Control officer with general guidelines as to the preferred fire regime, and managed by the Voluntary Bushfire Brigades.

District Objective:

1. Ensure that no more than 30% of an individual isolated reserve or fire management unit is burnt in a single prescribed burn operation.

2. Ensure that all burns are managed to obtain an agreed level of patchiness within the fire management unit or reserve.

3. Ensure that all burns adequately address the ecological needs of the biota present during the burn prescription planning and operational stages

These objectives seek to ensure there are sufficient fauna refuge areas and sources of seed bearing plants to prevent local extinction of species. This is more critical in isolated remnants of vegetation where the opportunity for recolonisation is limited. Where there is suitable unburnt vegetation in adjoining remnants to allow recolonisation, then the first objective is less critical. The area left unburnt can be either a discrete area, eg one side of a reserve, or as mosaic patches spread across the whole reserve, but not as one vegetation type within a reserve eg creekline vegetation within a reserve dominated by woodland. The creekline and the woodland both need to have some areas left unburnt. Mosaic burning within fire management units in the continuous vegetation in the Coolgardie sub-region will also be needed to achieve the finer scale patchiness desired to conserve biodiversity.

It is envisaged that district planning will be responsible for:
1. ensuring that all burns allocated to the district are assessed on the ground and managed according to accepted standards;
2. assessing any conservation risks and issues posed by the burn and resolving any conflicts;
3. setting ecological burn objectives and ensuring they are monitored appropriately;
4. ensuring that proposed burn areas are searched adequately for new threatened flora and fauna populations and that the health and location of known populations is updated in the corporate databases;
5. assessing viability of each nominated reserve and burn area before prioritisation at the strategic level of planning;
6. maintaining a reserves database to collect relevant fire management and biodiversity conservation information not available or pertinent to the corporate databases;
7. allocating individual burn prescription plans to appropriate officers to complete and maintain a register of district burn prescriptions;
8. ensuring all statutory planning requirements are met, including requirements for taking declared rare or priority flora;
9. ensuring all pre-burn management practices such as weed control, firebreak construction and maintenance and fox baiting are conducted in a timely manner and according to best management practice;
10. ensuring that any high priority burn areas deferred due to conservation issues or the need to control weeds, predators or grazing are brought to the attention of appropriate DEC managers for further action;
11. ensuring all fire management requirements contained in area management plans or recovery plans are incorporated into the strategic fire planning process;
12. implementing the strategic monitoring program to capture fire behaviour and fire response data;
13. ensuring that all neighbouring landholders are kept informed of the planned burns and invited to assist with complimentary management on their properties, such as weed control and fox baiting.

An indicative rolling 5 year plan for each DEC district in the Avon is proposed to allow time to prepare burn prescriptions and allocate resources, as well as meet statutory requirements such as applications to “Take” Declared Rare Flora during a burn operation. Areas with known or likely populations of DRF need to be adequately searched for new populations, and other operations, such as preparing firebreaks and pre-burn weed control, require timely management actions. Designing and implementing a monitoring program to measure the outcomes of the burn, such as regeneration of seedlings, can also require a long lead time.

It is expected that some nominated burns will not proceed in any one year due to unfavourable weather conditions or resource limitations. These areas should remain on the 5 year rolling burn plan until conditions are more favourable. Areas assessed at this stage as low priority according to the prioritisation process may be directed to the Volunteer Bush Fire Brigades with general guidelines for fire management.

DEC district nature conservation coordinators are best placed to assess nominated burn areas on DEC managed land as to their likely biodiversity conservation values, identify conservation risks and issues, set and resolve conflicting ecological burn objectives, and decide on the most appropriate burn regime to meet those objectives. Consultation with the district and regional fire coordinators will assist this process. Nature conservation and flora and fauna officers should be responsible for maintaining the Reserves database which should contain detailed local information such as weed species and areas infested, dieback areas, fire sensitive areas (eg small wetlands, cockatoo hollows), tracks and firebreaks, gravel pits and other details that are not available in the corporate databases. The Reserves database is currently used to capture reserve inspection data but has the capacity to collate useful fire management information and could be upgraded for this purpose. Consultation with the regional ecologist may be required to identify populations of flora and fauna or vegetation communities requiring special management during a burn operation.

Many populations of declared rare flora are located on private property and it is the role of the DEC flora officer to monitor the health of these populations and discuss with landowners which populations may require fire regeneration (Appendix V). Depending on the complexity, these burns may be managed jointly by landowners and the local bushfire brigade, under direction from the DEC. DEC district nature conservation coordinators may also assess other remnant vegetation and reserves of medium and high conservation value, such as covenanted bush areas, Land for Wildlife and some shire and roadside reserves, to highlight special conservation issues that require management action before, during or after a prescribed burn. These burns provide opportunities for private landowners and members of the Volunteer Bush Fire Brigades to learn about burning remnant vegetation for conservation and also the need to manage DRF populations appropriately.

There are a number of local area management plans which include fire management guidelines (eg Dryandra Woodland management plan, Department of CALM, 1995), and there are also fire management plans for specific reserves (eg Katanning eastern reserves fire management strategy, McCluskey *et al*., 2003; Wildfire Threat Analysis for crown lands between Coolgardie and Southern Cross, Daniel, 2003). DEC district staff need to ensure any burn requirements contained in these plans are incorporated into the strategic planning process and fire regimes implemented according to the those plans.

[C] Individual burn prescription planning
Individual burn prescriptions are completed for each reserve or fire management unit, however that may be defined. All of the conservation issues and management actions (eg weed control kangaroo control) identified at the 5 year planning stage need to be addressed and pre-burn management actions completed. Clear ecological management objectives and prescribed burn objectives must be defined and the outcome of the burn monitored (see Section 10.1). It is vitally important that good information is gained from all burns in the Avon Basin to build up the scarce knowledge base, particularly fire responses of individual species and the behaviour of fire in different vegetation communities.

The processes involved in preparing individual burn prescriptions within DEC are well developed in the forested south-west regions and can be readily adapted to the wheatbelt region. However, the major difference in fire management between these DEC regions is the smaller size of reserves and the greater degree of habitat fragmentation in the Avon Basin, and these differences need to be taken into consideration at all levels of planning. The DEC burn prescription process and checklist has recently been amended and is very comprehensive. Additional ecological checklists may be required to manage fire in isolated remnants and reserves such as assessing the presence of suitable adjoining remnant vegetation for fauna refuge areas, fox baiting in adjoining private property, kangaroo control and burning only a proportion of an isolated reserve at any one time. It is important that all burns aim for a mosaic pattern of burnt, partially burnt and unburnt vegetation and that the specific requirements of fire sensitive flora and fauna are considered a priority when preparing the burn prescription.

It is not intended to discuss this level of fire planning in any detail as it is beyond the scope of this project.

5.3 Priorities for fire management

With approximately 4.6 million ha of native vegetation in the Avon Basin, and assuming a hypothetical average fire cycle of 30 years (ie fire interval range of 60 years), then about 150,000ha would need to be burnt each year to maintain an acceptable fire age distribution. Some of this area will be burnt by wildfire each year, but it still leaves a very large area to be managed with minimal resources. Therefore a process for prioritising at the strategic planning level is essential to ensure those vegetation types, reserves and areas of high conservation value in greatest need of regeneration are given highest priority. This hierarchical process is sometimes followed intuitively by experienced land managers, but it should be developed into a clear, consistent and accountable process to meet ecological objectives and make most efficient use of available resources.

Assessment of reserve ‘condition’ and ‘conservation value’ for prioritization using multi-criteria analysis can be problematic and requires scores to be attributed to each criteria, often with little scientific basis (Safstrom, 1995). Simple criteria have been used in developing these initial priority matrices for fire management, which may be adapted as more information becomes available. They are necessarily subjective and should be regarded as a starting point for discussion, development of values and identification of knowledge gaps. The prioritization process can be applied at the strategic level and also at the local area or district fire planning level.

A separate prioritisation process for community protection during wildfire events, based on the Wildfire Threat Analysis, is being developed concurrently which includes assigning values to ecological assets (eg rare and priority flora populations, threatened ecological communities) and an assessment of fire behaviour in each of the broad vegetation classes.

The **first stage** of prioritization for ecological fire management is based on the **fire responses** of each broad vegetation class and/or vegetation association. As further knowledge becomes available, individual vegetation associations may be assigned different priorities, for example wandoo woodland may have a higher priority for fire management than salmon gum woodland and an expanded matrix may be required. The criteria for high, medium and low priority for fire response are assigned for the ten broad vegetation classes as follows:

High - heath, mallee shrubland, shrubland
Medium – woodland, low woodland, mosaic, hummock grassland
Low – succulent steppe, rock outcrops and wetlands

Salt lakes, claypans and adjoining areas of saltbush and samphire have no requirement for fire and are not considered past this point.

The **second stage** of prioritisation is related to the area, distribution and **fire history** of broad vegetation classes and vegetation associations remaining in conservation reserves, other reserves, unallocated crown land and on private property. The frequency distribution of fire age classes will show which vegetation types have an unacceptable proportion of area in the late senescent stage and are in greatest need of regeneration. For example, mallee shrubland may have only 10% of its total area in the juvenile stage compared with 30% for heath, so areas of older mallee shrubland will have a higher priority for burning. The criteria assigned for these fire age distributions are as follows:

High – vegetation associations with <10% total area in the juvenile stage
Medium – vegetation associations with 10 – 25% total area in the juvenile stage
Low – vegetation associations with >25% total area in the juvenile stage

These two factors, ie the response to fire and fire age distribution, are considered together in a matrix to determine the **'need to burn'** (Fig. 5.1).

When a subset of vegetation associations with high priority for fire management have been selected, individual areas can be identified in the ArcGIS database that are known to be long unburnt. For example, mallee shrubland may be found to have <10% of its total remaining area in the juvenile stage (<15 years since last burn), and would have a high priority for fire management. In this case, areas of mallee shrubland that are approaching senescence or older should be preferentially, but not exclusively selected for regeneration burning. Unfortunately, fire history data will at best provide information going back 30 years, well before the senescence age of many structural species, and so reasonable estimates of fire age are needed. A standard system for visually estimating time since last fire or growth stage would be extremely useful where no fire history data exists. Anecdotal evidence of previous fires from adjoining landholders should be sought and recorded wherever possible.

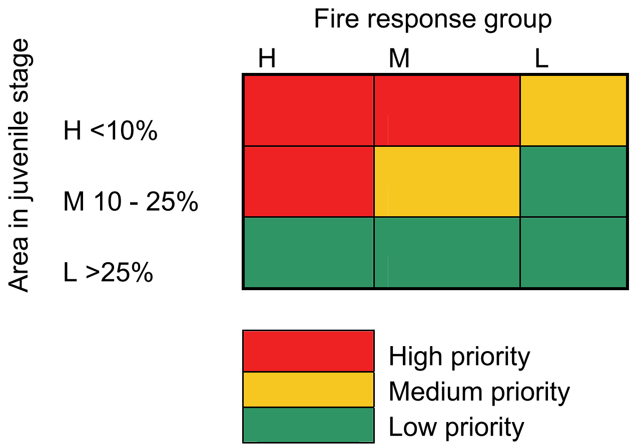


Figure 5.1 Matrix showing the overall priority for fire management of each vegetation class or vegetation association based on the criteria of fire responsiveness and proportion of total area in the juvenile stage. See text for criteria.

The **third stage** of prioritisation is based on known **special conservation values** for a reserve or area of remnant vegetation. Areas with declared rare flora and priority species, threatened ecological communities, poorly represented communities and threatened fauna habitat known to be in need of regeneration should have higher priority for burning than areas with no special conservation values. Location of rare and priority flora, threatened ecological communities and threatened fauna sightings have been included in the ArcGIS project with these guidelines. This is a risk minimisation approach to avoid the loss of threatened species. The criteria assigned for conservation values are as follows:

High – contains declared rare or priority flora populations
- contains known threatened fauna habitat
- contains threatened ecological community
- contains poorly represented vegetation associations (<1,000ha of pre-European extent)

Medium – contains none of the above conservation values
- contains geographically restricted flora
- contains likely fauna habitat for threatened or locally extinct taxa
- contains known habitat of declining species
- contains moderately represented vegetation associations (1,000 – 5,000ha of pre-European extent; or 5,000 - 10,000ha and <15% of pre-European extent).

Low – contains none of the above values
- contains well represented vegetation associations (>5,000ha and >15% of pre-European extent).

In terms of representativeness, there are 56 vegetation associations in the high conservation value category (including 9 with no area remaining and 11 with <20ha remaining), 25 in the medium category and 41 in the low category (including 11 with <15% of pre-European extent but with > 10,000ha). These figures only refer to areas within the Avon Basin boundary and additional areas of each vegetation association may be found in adjoining areas. These categories need to be attributed in the ArcGIS project.

Many poorly represented vegetation associations occur as fragmented isolated patches in reserves and remnants and cannot be managed as discrete units on the ground. For this reason, vegetation associations with total areas less than 1,000ha have been amalgamated into other vegetation associations to provide more sensible fire management units. However, vegetation associations in this category that contain fire sensitive species need to be identified during the strategic planning stage and managed accordingly, or nominated as threatened ecological communities.

Likely habitat for threatened or locally extinct fauna (Fauna Habitat Zones) needs to be identified and mapped from local knowledge of prior occupancy, sightings and museum records, as has been done for the southern forests in the Fauna Distribution Information System (Christensen, Liddelow and Hearn, 2005). Burn prescriptions can then be prepared with due regard to predicted occurrence and fire sensitivity of threatened and priority fauna. A comprehensive list of declining species and their preferred habitat is also required for the Avon Basin.

The **fourth stage** of prioritisation is based on the **viability** of the reserve or area to be burnt. This reflects the condition and degree of disturbance and fragmentation of the area and its ability to maintain ecological values and processes. Safstrom (1995) developed a framework for evaluating the conservation values of small reserves in the central wheatbelt and used simple criteria (including visual assessment) to assess viability (shape and size of remnant, position in the landscape, adjacent land uses and occurrence of weeds). He considered that small reserves with significant areas less than 100m wide, were low in the landscape and affected by rising water tables, were affected by drainage of saline water from adjacent land or with few areas of intact native vegetation have low viability.

Ideally, these viability attributes should be included in the Arc-GIS regional database so that they are considered at the strategic planning stage. However this level of knowledge is currently only accessible at the local level, and often not recorded in a consistent format or not recorded at all. In many cases, the condition of a reserve will need to be assessed following its nomination on the Indicative 5 year plan, and a decision to proceed with the burn made at this stage. The criteria used to assess viability of reserves include:

High - >500ha in total area, and
- <30% weed cover, and
- <20% affected by soil disturbance, and
- high in the landscape and unaffected by rising saline groundwater

Medium – 30 to 500ha in total area and >100m wide, or
- 30 to 60% weed cover, or
- 20 to 50% affected by soil disturbance, or

- mildly affected by rising groundwater or adjoining land uses ('areas at risk of becoming saline')

Low - <30ha in total area or significant areas <100m wide, or
- >60% weed cover, or
- >50% soil disturbance, or
- low in the landscape and significantly affected by rising groundwater or adjoining land uses ('recent saline areas and older saline areas')

Areas affected by rising groundwater and secondary salinisation are unlikely to respond to burning, but this supposition still needs to be tested (see Section 6.4). **Salinity risk mapping** from Land Monitor is a useful layer in ArcGIS that can be superimposed on the vegetation maps to assess, at the strategic level, which remnants should be given a lower priority for viability. This salinity risk mapping has four categories, areas at risk of becoming saline, recent saline areas (1995-1998), older saline areas (1989-1991) and unaffected areas. These can readily be adopted as one component of the viability assessment (see above). Further assessment of salinity risk mapping will be required at the local level to determine the validity of the satellite based information. More recent data capture is also needed to review the changes in areas affected by salinity since 1998.

The third and fourth stages of prioritisation are considered together in a matrix to assess **'value for money'** for each proposed burn (Fig. 5.2 a-c). It provides further discrimination of areas selected from the first matrix (Fig. 5.1) to determine which areas of each vegetation association should be nominated for the Indicative 5 year plan. In the above example, mallee shrubland was found to have a high priority for fire management. The database is queried to find areas of mallee shrubland that meet the criteria for high or medium conservation value and that also meet the criteria for high or medium viability. These selected areas then have the highest priority for prescribed burning and are nominated to the Indicative 5 year Plan.

A **fifth stage** of prioritisation involves some consideration of the **urgency** of the management required. Populations or habitats that are declining most rapidly will require the most urgent attention and be given higher priority than those that are declining slowly. At present, there are few situations in which the rate of decline is being monitored. There may also be some areas that urgently need burning for community protection reasons and this should increase the ranking of those areas in the strategic planning process.

This decision process could be used to create a standard query based front end for the ArcGIS project developed for these Fire and Biodiversity guidelines. It involves sequentially querying the datasets to find subsets that meet all the criteria for high priority for fire management. Further queries are then conducted to find subsets for medium and lower priorities until the total area that can be managed for prescribed burning in a particular year is met. The areas selected from this prioritisation process can be sequentially ranked or weighted according to an agreed process. Resources may dictate that only those areas in the high and medium (red and amber cells in Fig. 5.2) priority category are considered for prescribed burning.

Alternatively, a Master Burn database could be developed for the Avon Basin that contains all fire management information on a fire management unit basis ie individual reserves, groups of small reserves or remnants, or blocks defined by firebreaks or tracks in the areas of continuous vegetation or larger reserves (>5,000ha). Each fire management unit in the database can be attributed with fields for tenure, the areas of each vegetation association, fire history, conservation values and viability or management issues. The whole prioritisation process (stages 1-4) can be analysed from this data and a ranking assigned to each fire management unit, which is updated every three years eg as time since fire increases or as an area is burnt. The database could then be queried to show all fire management units with a ranking greater than (say) 5 or queried iteratively until the area available for burning matched the available resources. This would be a more efficient system than performing prioritisation queries using an ArcGIS project to determine areas available for burning on an annual basis. However, there is currently no agreed system for determining fire management units in the DEC wheatbelt region which is a basic requirement for developing a Master Burn database.

A similar approach was used for fire management in the Cape Province in South Africa, based on simple rule-based models (Richardson *et al.*, 1994; van Wilgen *et al.,* 1994). Their model also included a fire hazard rating for the area which had the potential to increase the ranking or priority of the burn, for example if neighbouring properties were in danger. An important aspect of these models was the computer generated summary tables and graphs such as current age distribution of vegetation, and percentage of area burnt in different months in the last year. These graphs are used to show whether burning programs for different areas are achieving the desired ecological patterns and are an effective means of monitoring fire management.

This prioritisation process may seem, and often is, a relatively straight forward process but there are many instances where it is not so clear cut. For example, many critically endangered flora species in the wheatbelt are found only in weed infested and narrow roadsides. Regeneration of the few remaining adult plants by fire in these conditions may be vitally important for the species survival. In addition, prescribed burning of these plants requires a long process of approval and a disproportionately high level of resources. Balancing conservation value with condition of reserve or remnant is important and areas of low viability should not be disregarded before they have been adequately assessed for conservation value.

All remnants in the wheatbelt have some conservation value, even grazed or weed infested woodlands and roadside remnants provide important habitat for many bird species, provide stepping stones to other larger remnants and may contain populations of plants that are geographical variants. It is important to recognise that these areas may still benefit from burning and indeed appropriate fire management can be used to improve their conservation value.

However, many small areas of remnant vegetation do have low conservation value and low viability such as narrow degraded roadsides, railway reserves, road realignments and heavily grazed vegetation on private property. After an initial ground inspection these areas may be ranked as low or very low priority for fire management, and may never appear on the burn program. It would be sensible to omit these areas from the Master Burn database, but keep them as a colour coded layer in a GIS database for future reference. Managers of these areas of remnant vegetation could be provided with information about using fire to regenerate the remaining species, along with advice for controlling weeds and managing grazing pressure.

Often several management actions are required (eg control of weeds, control of rabbits or kangaroos) before plants can be successfully regenerated by fire, even in larger reserves. There is no sense in prescribing fire to regenerate jam and sheoak woodland if the kangaroo population is so high that all seedlings will be rapidly removed by grazing. These management issues can only be determined at the local level from on-ground inspection and experience. It requires a **'limiting factors'** or **'factor resolution'** approach, to identify what factors are actually limiting the regeneration of the population, habitat or community (Yates and Broadhurst, 2002). Only then can the priority for burning be determined. Prescribed burning may have to be delayed for 5 -10 years until the kangaroo population has been controlled or the weed infestation reduced, after which it can be re-nominated to the 5 year planning process.

In reality, most reserves will have some areas that have high conservation value (eg a population of declared rare flora) and some areas that have medium (eg likely threatened fauna habitat) or low (eg old gravel pit) value. Each individual reserve or remnant could contain several different vegetation associations that may need to be managed differently. The mix of vegetation types, fire history and relative areas of high, medium and low conservation value will determine the decisions:

a) whether it needs to be burnt,
b) its overall conservation value, and
c) the most appropriate fire regime to conserve biodiversity.

Other methods for assessing conservation value have been developed which could be useful for further development of this prioritization process for fire management. A key to assess the ecological significance of on-farm bush remnants in the wheatbelt was developed under the "Save the Bush" program and includes an assessment of the complexity of the vegetation, presence of understorey and gazetted rare flora, level of disturbance and impact of grazing domestic animals (Mollemans, 1993). Habitat complexity scores based on the area or structural diversity of a patch can also be used as surrogates or indicators of biodiversity or conservation value by predicting suitability of reserves or remnants for fauna (eg Lambeck, 1997; Newby, 1999; Freudenberger, 2001; Watson *et al.,* 2003).

5.4 Fire planning model

Considering all of the above there needs to be a clear process to ensure that strategic objectives are being met and that the planning decisions and priorities are accountable to the general public.

There are two main ways that areas can be nominated for prescribed burning:
1. Areas available to be burnt should routinely be identified through the strategic GIS analysis or Master Burn database and prioritisation process described above.
2. DEC district offices, local government, landholders, bush fire brigades or research staff may also identify areas that require burning to protect specific community assets, regenerate particular areas of high conservation value or to conduct fire response or fire behaviour research.

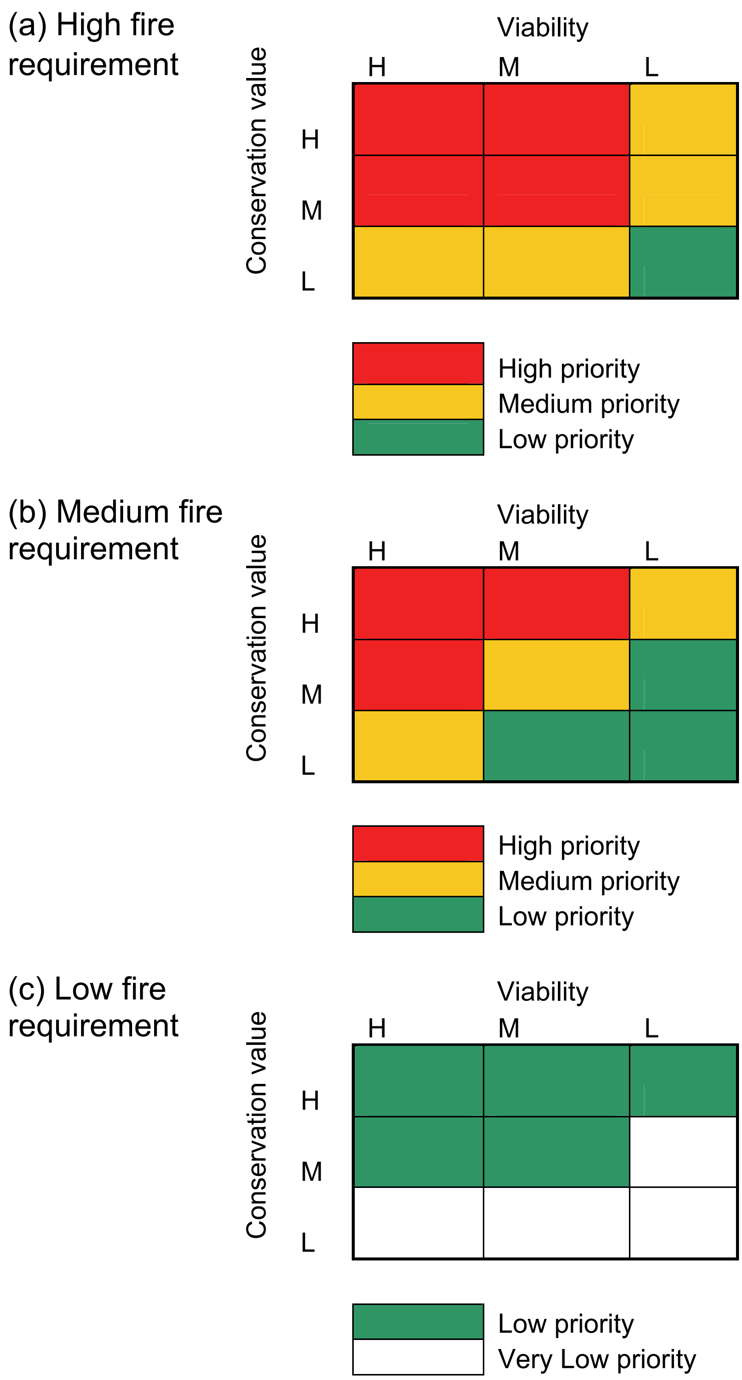
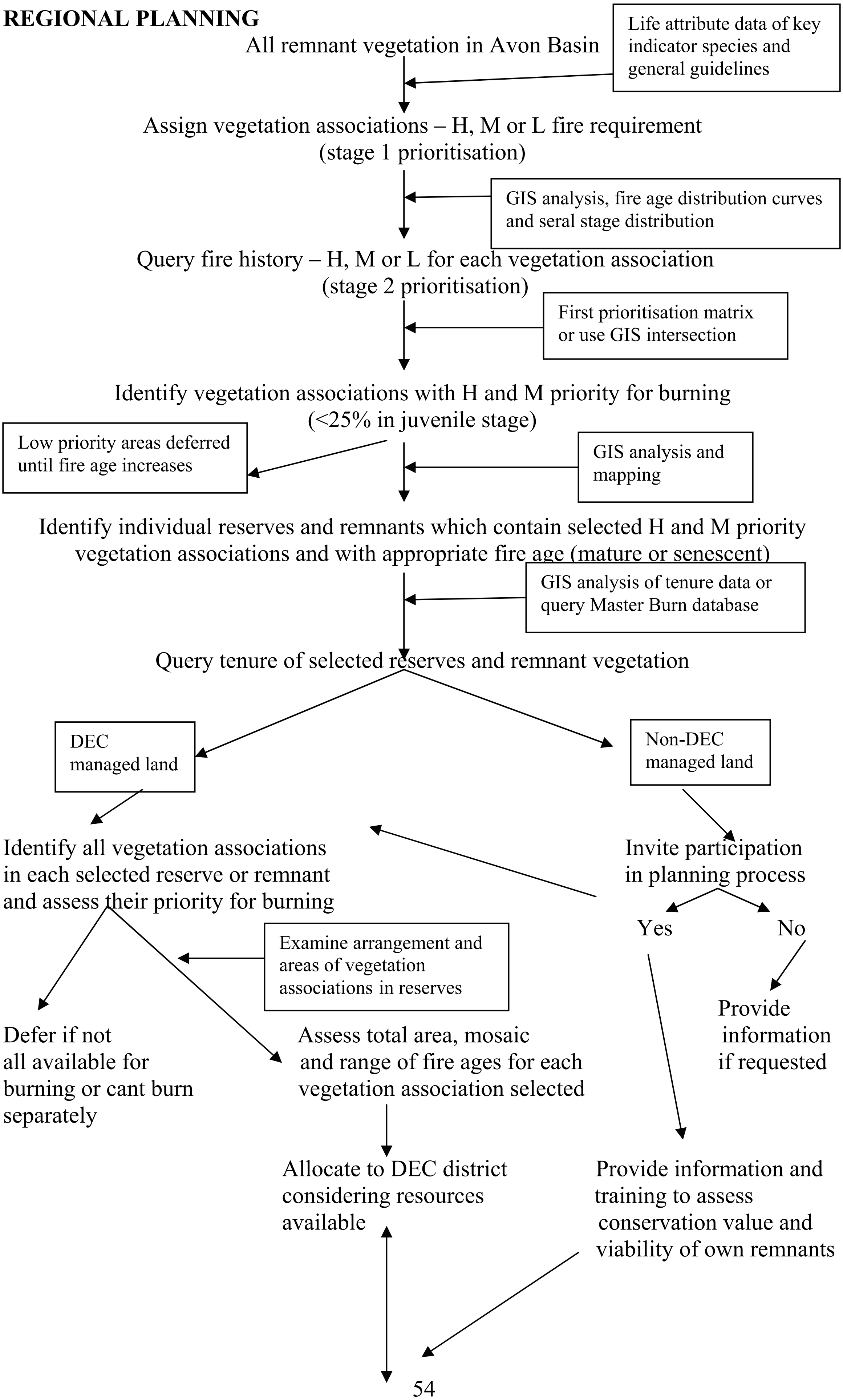


Figure 5.2 Priority matrices for (a) high, (b) medium and (c) low fire management priorities (see Fig.5.1), based on criteria for conservation value and viability.

The following diagram (Fig. 5.3) provides a basic decision model and indicates some work flows at the regional and district levels of planning. There are many more steps and pathways than can be shown in this way including the iterative processes involved between districts and regional staff and the Fire Management Services within DEC before burns are nominated to the 5 year Indicative Plan and before final Burn Prescriptions are approved.

REGIONAL PLANNING



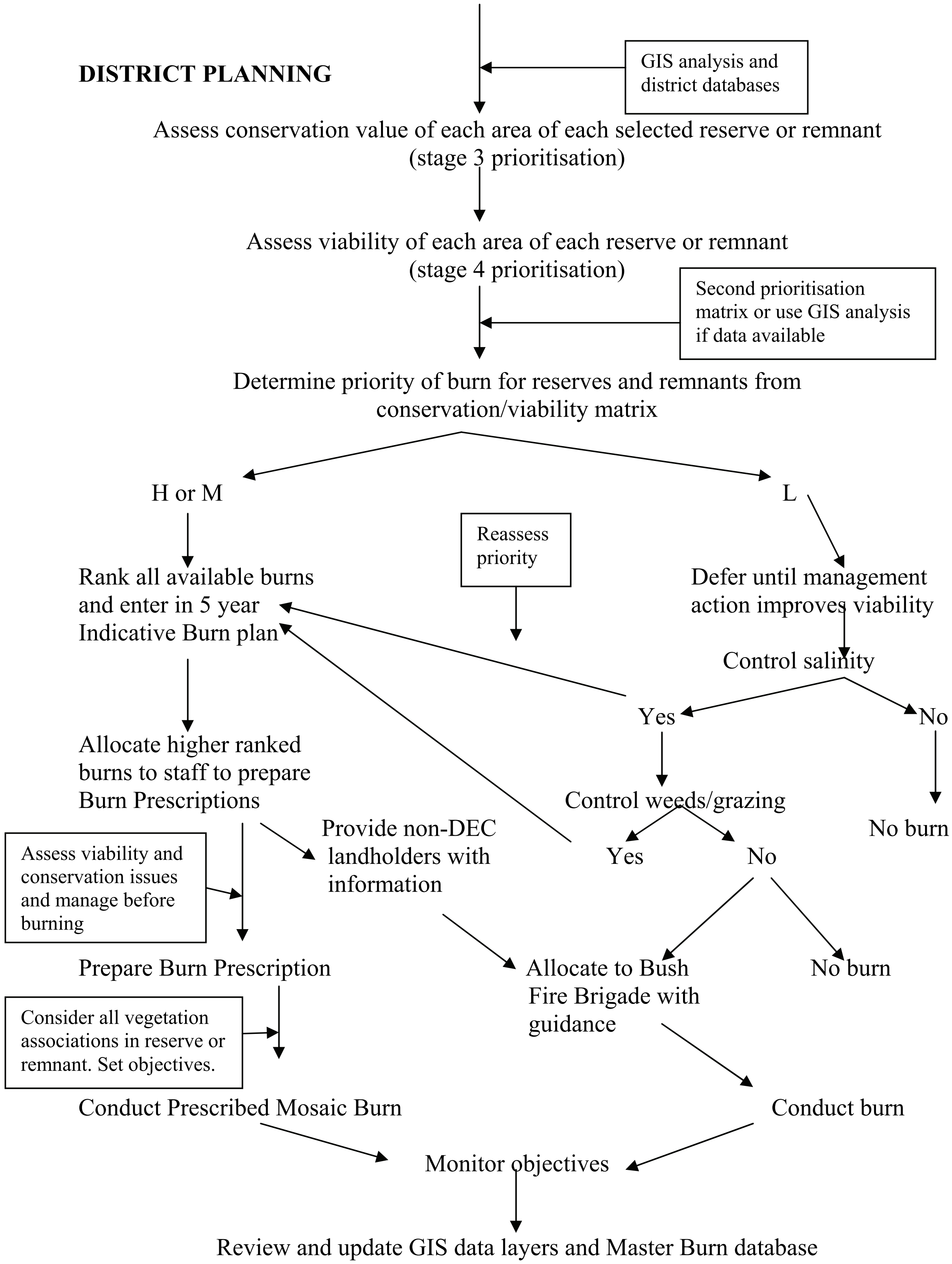


Figure 5.3 Diagram showing decision model for fire management planning in Avon.

5.5 Fire planning for private remnant vegetation

These guidelines cover native vegetation across all land tenures in the Avon Basin, including privately owned remnant vegetation. The main reason for seeking to involve private landowners in the process is that much of the remaining native vegetation in the Avon Basin is privately owned. Approximately half of know populations of declared rare flora in the wheatbelt occur on private property. Another important reason for coordinating fire management with private landowners is to improve the likelihood that threatened fauna may safely disperse to, and recolonise from, adjoining areas of vegetation during and after burn operations. Coordination of fox baiting and weed control on adjoining properties prior to burning greatly increases the effectiveness of these control methods. Private landowners involved in this overall planning process can improve their understanding of appropriate fire regimes on their own properties and help to conserve biodiversity across the Avon Basin.

There have also been requests to DEC, Land for Wildlife and local governments for assistance with fire management in private remnant vegetation. In a major survey of 143 farmers in the Pingelly, Tammin, Kellerberrin, Lake Grace and Dumbleyung shires in 1996, 29% indicated they would be willing to burn their bush (Jenkins, 1996). Jenkins (1996) stated that "Many people who felt that their bush needed to be burnt to encourage regeneration had not done so because they recognised that they had insufficient knowledge of the effects (of fire) on the vegetation and wildlife. When people stated that they would be prepared to burn their bush, they qualified this by saying that this would be only if they had input from experts." Nearly half of the farmers (46%) considered that the bush would regenerate itself if fenced and left alone. Clearly there is interest by landholders in regenerating their bushland but they are apprehensive about using fire and acknowledge their lack of knowledge in this area. Given the appropriate advice, knowledge and skills needed many could be encouraged to engage in ecological burning practices.

An excellent publication is available for managing private remnant vegetation ('Managing your Bushland', Hussey and Wallace, 2003) which contains a chapter on ecological fire management. A 'Land for Wildlife' newsletter also provides advice on burning small remnants (Hussey and Baxter, 2006). Both these publications contain useful information and checklists to help guide the decisions of landowners as to why, when and how to burn their remnant vegetation. Further information, field days and training on fire management in native vegetation should be provided to private landholders on request. Volunteer bush fire brigade members should be provided with training on fire management for biodiversity conservation so they have a greater understanding of the different fire responses of flora and fauna in the Avon Basin, and the need for adequate planning and reporting processes.

6. MANAGING FIRE IN A FRAGMENTED LANDSCAPE

6.1 Fragment size, shape and connectivity

Large areas of the original vegetation in the wheatbelt have been cleared for agriculture or grazed over a long period with a significant loss of habitat and devastating impact on wildlife. Woodlands in particular have been extensively cleared with as little as 3% of some woodland types remaining (Yates and Hobbs, 1997). Woodlands and other vegetation associations now occur as remnants of varying size, quality and isolation. Many remnants are under threat from further clearing and firewood gathering, rising saline water tables and increased waterlogging, grazing by livestock and rabbits, nutrient enrichment, soil compaction and structural decline, increased exposure to storms and wind damage and invasion of exotic weeds. Many of these disturbances are present within remnants and often act synergistically. In addition fire regimes have altered over time and in some cases remnants are subjected to frequent intense fires or remain unburnt for long periods driving changes in vegetation and may have exacerbated the impacts of other threats. Fire management planning in fragmented areas therefore must consider all these threats and determine if the planned introduction of fire may benefit the situation or cause further deterioration.

Size of remnant in itself should not be seen as an over-riding criterion for prioritizing areas for prescribed burning. In one study in the northern wheatbelt, a 15ha grazed remnant of salmon gum and York gum used by six species of cockatoo for nesting has shown considerable decline in condition. There has been no evidence of regeneration of woodland species since 1929 when sheep grazing was first introduced to the area (Saunders *et al.,* 2002). This tree decline was exacerbated during a period of lower than average rainfall (1978 – 1981) and has impacted significantly on the local availability of nesting hollows for cockatoos. Regeneration of this small woodland remnant will require appropriate prescribed burning together with removal of livestock to maximise the chances of seedling survival.

Habitat fragmentation reduces total population size and creates small genetically and geographically isolated populations which may result in loss of genetic diversity and population viability. Depending on their breeding systems, increased self pollination leading to inbreeding, lack of pollinators or reduced pollination and seed set and can lead to reduced reproductive success, but this is not always the case (Yates and Ladd, 2004). For example, small populations of the rare *Verticordia fimbrilepis* subsp. *fimbrilepis* in isolated remnants (eg on roadsides) had greater diversity of insect pollinators and similar seed set to larger populations in large undisturbed conservation reserves (Yates and Ladd, 2004). Fire killed the adult plants but resulted in mass recruitment of seedlings compared with sporadic seedling emergence in the absence of fire. Yates and Ladd (2004) concluded that suppression of fire in these smaller remnants may have an adverse effect on the survival of some rare species.

The spread of fire across fragmented landscapes differs from that in continuous vegetation landscapes (Yates and Hobbs, 1997). Fire regimes have altered since European settlement and may have become less frequent in some communities. This could be due to active suppression of wildfires, cessation of aboriginal burning practices, and reduced fuel loads and spread of fire during summer in the agricultural cereal cropping and pasture lands surrounding the remnants. In other communities, fragmentation may have been associated with increased fire frequency, such as fires that escaped from clearing burns in the 1940's-1970's, and from stubble burning or accidental fires from harvesting equipment.

One of the main fire management objectives should be to avoid having any isolated reserve or remnant (large or small) being burnt out completely by a single fire

(Hopkins, 1985). This is particularly relevant for the Avon Basin where there are hundreds of small reserves in a matrix of cropping and pasture land with few viable connecting corridors (eg road reserves, fenceline vegetation, shelter belts). Connectivity to other suitable habitat is critically important for flora and fauna species with poor dispersal abilities. Complete burning of isolated remnants can lead to local extinctions of species that have poor survival and dispersal mechanisms. In contrast, burning one part of a remnant may alter the habitat quality and resource availability for a time but suitable habitat may be available elsewhere within that remnant until the burnt area recovered (Hobbs, 2002).

These aspects of habitat suitability (size and connectedness) have been modelled for a number of declining bird species in the wheatbelt (Brooker *et al.,* 2001; Brooker and Lefroy, 2004), and provide a basis for determining optimal areas for revegetation. They could also be used to determine minimum areas to be left unburnt within remnants or within a landscape of connected remnants, to increase the likelihood of recolonisation after a fire. For example, the Golden Whistler and Jacky Winter require more than 50ha of woodland in a remnant to have >60% probability of occupancy (Lambeck, 1998). Retention of 50ha of unburnt habitat in a landscape may increase the chances of continued occupation by these species.

Edge to area ratio of reserves can be a useful criterion for assessing viability and likelihood of invasive species. Narrow isolated reserves less than 100m wide are considered to have low viability as they constitute mainly edge habitat which is likely to be degraded (Safstrom, 1995). These reserves are also more exposed to being completely burnt out by fire from adjoining properties and would have a lower priority for prescribed burning.

Management of fire for biodiversity conservation in a fragmented landscape depends on understanding ecological processes across the landscape, and requires some sympathetic management of adjoining remnants across tenure boundaries on private property. Each fragment has to be considered a part of the whole functioning landscape and containing a portion of the whole population. These aspects require effective communication and coordination with all landholders to avoid local species extinctions.

6.2 Fire and weed interaction

Weed invasion is one of the greatest challenges in conservation management particularly in small isolated reserves in the wheatbelt. Long narrow remnants are more prone to weed invasion than those with a low perimeter to area ratio, due to the greater interface with agricultural land and dispersal of weed seeds across this boundary. Remnants adjoining long settled agricultural land and those which have been subjected to past soil disturbances (eg gravel pits, picnic sites, rubbish dumps) can have very high levels of weed infestation. These historical differences need to be taken into account when assessing reserve condition and considering options for restoration.

Fire can and often does exacerbate this problem by stimulating weed species to germinate and provide the additional flush of nutrients, water and light needed by these species to flourish. A high proportion of weeds in wheatbelt reserves are annuals (Fig 6.1) which rapidly respond to fire and have the capacity to complete several life cycles post-fire before being out competed by slower growing native shrubs, and so density and extent of weed populations can increase markedly following fire. Where a dense cover of weeds has established, it may be very difficult for native species to regenerate. On the other hand, healthy native vegetation with dense canopy cover, including shrubs and ground cover species, can prevent weed establishment in many cases, and this balance can be manipulated by the careful use of fire.

Figure 6.1 Typical jam-sheoak low woodland in a wheatbelt reserve with understorey almost completely replaced by annual exotic weeds, and in an early senescence stage.

Weeds that are fast growing and widely dispersed have had adverse effects on the recruitment of many native species including some rare species (Yates and Broadhurst, 2002). Annual weeds had a major inhibitory influence on seedling recruitment of two critically endangered *Acacia* species in the northern wheatbelt (Yates and Broadhurst, 2002). *Acacia aprica* seedling survival and subsequent seedling growth were significantly greater in weeded than unweeded plots following experimental fires.

Many wheatbelt reserves and roadsides are already severely infested with weeds and decisions have to be made as to whether there is any benefit in burning them. This will depend on the conservation value of the reserve in terms of presence of threatened flora and fauna populations, threatened ecological communities or poorly represented vegetation associations as well as its functional role in the landscape (see Section 5.3). There are various methods for assessing these values (eg Mollemans, 1993; Hussey, 1999; Brown and Brooks, 2002; Moore and Wheeler, 2002), which require on-ground survey and evaluation of both the conservation value and the level of weed infestation. If it is considered that the reserve has medium to high conservation value, or that a regeneration burn is required, then methods of weed control must be investigated before fire is introduced and before any firebreaks are maintained or constructed to prevent exacerbating the weed problem. In some cases, such as salmon gum and York gum woodlands, the trees may be in urgent need of regeneration to avoid massive structural decline of the vegetation and a combination of burning and chemical weed control may be the only viable option.

Other considerations that need to be taken into account in determining if it is beneficial to burn a weedy reserve include:

- the direct impact of physical or chemical weed control on native species
- the weed species present and their impacts on native vegetation
- the extent and spatial pattern of weed invasion
- response of the weed species to fire
- the likelihood of re-invasion from outside the burn area
- accessibility for controlling the weeds
- whether physical and/or chemical control methods are required
- whether resources are available for post-burn weed control

Fire can sometimes be used to aid in the control of weeds (McMahon *et* al., 1994; Brown and Brooks, 2002). Mass germination of some weeds following fire can provide the opportunity for effective one-off control and reduce the need for staggered follow-up spray programs. In some situations a second fire soon after the initial fire can destroy weed seedlings before they produce more seed, but great care is required to avoid the loss of beneficial native species with long juvenile periods. Intense wildfire may also destroy the seed bank of some weed species, especially if the seed is lying on the soil surface among litter rather than buried. Fire in some seasons may be more effective in controlling weeds (eg Victorian tea-tree) than others (Molnar *et al*., 1989), but we have little information on this possible method of control. Clearly, much research and monitoring is needed to investigate how the balance between weed and native species regeneration can be manipulated using fire, herbicides and alternative forms of soil disturbance.

It should be noted that some vegetation types appear not to be invaded by weeds to any great extent and this is currently being examined in eastern wheatbelt (Colin Yates, personal communication). Where shrubland and shrub heath vegetation are dense and respond rapidly following fire there is little opportunity for weeds to become established. Sandplain soils are naturally very low in nutrients and this combined with the low rainfall in eastern regions does not favour weed seedling survival even in close proximity to agricultural paddocks. The role of dense leaf and twig litter as a mulch which may prevent weed establishment also warrants further investigation.

6.3 Fire and grazing pressure

Native vegetation is regularly grazed by kangaroos, euros, wallabies and other smaller fauna, and by grasshoppers, caterpillars, psyllids and other invertebrates. Grazing directly reduces canopy cover, vegetation biomass and reproductive capacity of adult plants, alters vegetation structure and can cause significant seedling mortality. Grazers are often selective and can alter species composition. This loss of canopy cover and altered structure may decrease the habitat suitability for other dependent fauna.

Grazing pressure varies with season and vegetation composition, but can increase significantly after a fire when seedlings and new shoots are more palatable. The greatest impact is on germinating grasses and herbaceous species, but in drier conditions, regrowth from rootstocks and shoots of shrub and tree species are affected. In small reserves and remnants, or in smaller patchy burns, this interaction can greatly alter the vegetation structure and species composition, which can subsequently affect the amount and distribution of available fuel and behaviour of future fire events. This impact may be less in larger non-patchy burns, but will still depend on the total grazing pressure.

Seedlings are most susceptible to damage and mortality loss by grazing after fire, with more palatable species, including leguminous species, being more affected. Exclusion from grazing after fire resulted in greater seedling emergence and increased growth during the first year in two critically endangered *Acacia* taxa (Yates and Broadhurst, 2002), although this effect tended to be site specific. Regeneration of woodland tree species (*Callitris preissii* and *Melaleuca lanceolata*) and *Acacia rostellifera* scrub on Rottnest Island was largely prevented by overgrazing by quokka during the 1930's to 1950's when they were protected, resulting in a marked reduction in the area of woodland on the island (Rippey and Hobbs, 2003). A major wildfire in 1955 burnt two-thirds of Rottnest Island and the area covered by woodland was further reduced as surviving quokka soon ate out most of the regenerating *Acacia* scrub. After a smaller fire in 1997, *Acacia rostellifera* thickets dominated within grazing enclosures, where it had not grown for over 40 years since after the fire in 1955, and reached 3-4m tall. Some *Melaleuca lanceolata* seedlings also germinated and were expected to emerge when the shorter-lived *A. rostelliffera* declined. Outside the enclosure, few seedlings of *Acacia, Melaleuca* or *Callitris* survived in the presence of the large quokka population (Rippey and Hobbs, 2003). Other grazing exclosure experiments have shown that nearly 3-4 times more seedlings established on ungrazed plots relative to grazed plots after fire (Leigh and Holgate, 1979). Grazing by native animals after fire may have greater impact on vegetation than the fire itself.

Livestock grazing of reserves and remnants in the wheatbelt has had a significant detrimental impact on native vegetation structure and composition. Large areas of native vegetation were grazed by sheep since early settlement and rabbits have caused further loss of vegetation. The impacts of livestock grazing was studied in salmon gum woodlands in the central wheatbelt (Yates *et al.*, 2000). They found that heavy grazing of remnant vegetation on farm by sheep caused a decline in native perennial cover, an increase in exotic annual cover, reduced litter cover, increased soil erosion, decreased soil water infiltration and soil structure decline. They also measured significantly greater daytime soil temperatures in grazed plots (36.1°C cf 25.7°C in rarely grazed plots) and greater daily fluctuations in soil temperatures. These changes may seriously affect the ability of seeds to germinate and fine surface roots to take up nutrients.

Grazing and trampling by hard hoofed animals leads to much plant and soil degradation and loss of ecosystem function. In some remnants in the wheatbelt, there is little evidence of eucalypt trees regenerating since sheep were first introduced in the 1920's (Saunders *et al.,* 2002). Simply removing livestock grazing may not restore plant species diversity and community structure (Yates *et al.*, 2000). Woody debris and litter need to be retained and soil roughness promoted to capture and retain soil water and nutrients. In addition, low shrub canopies improve soil and surface microclimates and provide more favourable habitat for beneficial soil fauna. These aspects need to be considered when prescribing fire in heavily grazed remnants, as fire may exacerbate the loss of canopy cover, litter, woody debris and soil organic matter needed to restore soil conditions for regeneration of vegetation.

In some cases, the impacts of fire and grazing may be considered to be substitutable (Hobbs, 2002). Grazing reduces the available fuel load and hence the fire hazard, and in grassy woodlands in eastern Australia, light grazing pressure with appropriate resting phases has lead to increased species diversity. Grazing opens up the grassy canopy and allows tree, shrub and herb seedlings to germinate (Leonard and Kirkpatrick, 2004). However, heavy grazing removes considerable fuel to the extent that fire may not carry and regeneration of vegetation is prevented.

In fragmented reserves and remnants of the wheatbelt, there is increasing anecdotal evidence that the total grazing pressure has increased, most likely from increased population size of kangaroos. Many landholders have reported increased kangaroo numbers on their properties and increased damage to crops and pastures. Several enclosures have demonstrated the huge impact of grazing on vegetation cover and composition (Fig. 6.2), yet we have little real data on changes in kangaroo populations or on their impact on native vegetation with or without fire. Several species were only found within the enclosure in Fig. 6.2(a) and there was a complete lack of regeneration of *Acacia acuminata* outside the enclosure due to grazing. Preferential grazing by kangaroos of the more palatable *Allocasuarina huegeliana* seedlings in Fig. 6.2(b) is likely to have a major impact on the future structure of this vegetation. It is likely that management of kangaroo populations will be needed before fire is prescribed in small and isolated fragments of remnant vegetation to minimise the impacts of their grazing on germinating seedlings.

(a) (b)

Figure 6.2 Examples of the impacts of kangaroo grazing on vegetation. (a) An unburnt area enclosed by a 2m high fence about 6 years ago to protect a critically endangered orchid, showing the lack of regeneration outside the fence caused by excessive grazing by kangaroos. (b) An area burnt by a hot fire showing the dense regeneration of *Allocasuarina huegeliana* seedlings inside the enclosure, while very few seedlings of this species were found outside the fence.

6.4 Fire and salinity

The impact of fire in areas affected by salinity has not generally been investigated. Much of the native vegetation around lakes, wetlands, broad valley systems and creeklines has deteriorated due to rising saline groundwater and in many cases the original vegetation has died and is being replaced by more salt tolerant shrubs or annual exotic grasses (Fig. 6.3). In a major regional biodiversity survey of the wheatbelt, it was found that fifteen plant assemblages containing 472 native taxa are at particular risk from secondary salinisation, including taxa from both non-saline and naturally saline wetlands (Lyons *et al.*, 2004).

(a) (b)

Figure 6.3 Land affected by rising saline groundwater is a common sight in the wheatbelt and affects many nature reserves that are low in the landscape. (a) This shows that the additive effects of salinity and grazing together can cause severe degradation. (b) Rising saline groundwater is encroaching into this area of wandoo woodland. Fire management is unlikely to be of any benefit in these salt affected areas.

Fire is not required to regenerate succulent shrubs and saltbush, but may stimulate a mass release of seed from *Melaleuca* and *Allocasuarina* species and other taxa that may persist in mildly salt affected land. Remaining vegetation is generally at greater risk of degradation from increasing salinity levels and waterlogging than from fire, and indeed, fire may accelerate the rate of decline. A sudden loss of canopy from burning may trigger a more rapid rise in the groundwater level and further impede the regeneration of seedlings and coppice growth. It is likely that seedlings will be less tolerant of saline and waterlogged soil conditions than adult plants and the risk of high seedling mortality following a prescribed burn should be given serious consideration.

There may be some situations, however, when fire may regenerate a stand of *Melaleuca* or *Allocasuarina* shrubs or *Eucalyptus* trees (eg *E. loxophleba, E. kondininensis, E. sargentii, E. spathulata*) surrounding salt affected land which could help to lower the saline water table and improve conditions for other less salt tolerant species. Without some fire or other disturbance, these species could die out simply from lack of such a regeneration opportunity. Trials need to be conducted to determine which situations (vegetation type, extent of degradation, senescence stage) might benefit from some introduction of fire. It is recommended that only a small section of fringing vegetation be treated with fire and the response monitored carefully. It would be useful to combine this with monitoring the groundwater response.

Fire may also be used in some low lying areas to regenerate fauna habitat. In the Perup forest area, fire is used to regenerate dense stands of *Melaleuca* along creeklines which provide useful protection for tammar wallabies from predation. Sections of creekline are sequentially scrub rolled and burnt on a 15 year rotation to ensure that the habitat does not completely senesce and is not all at risk from being burnt out in a wildfire event.

Areas affected by salinity have been mapped using Landsat data and Landmonitor digital elevation models combined with air photo interpretation and field observations and is included in the DEC corporate GIS datasets (Fig. 6.4). The salinity risk layer is part of the ArcGIS project developed for these Guidelines and provides a useful tool to assess areas already affected and likely to be affected by salinity. The Beard-Hopkins pre-European vegetation association mapping can also be used to identify succulent steppe vegetation types which occupy low lying broad valleys, often with sparse or open woodland (see Fig. 7.2). These two layers combined can help to determine which vegetation types are most at risk and which conservation areas have a significant proportion affected by increasing salinity and will help with priority setting for prescribed burn areas. They will generally have a low priority for burning due to their low viability status but should not be neglected completely, as judicious fire management could benefit some of these areas.

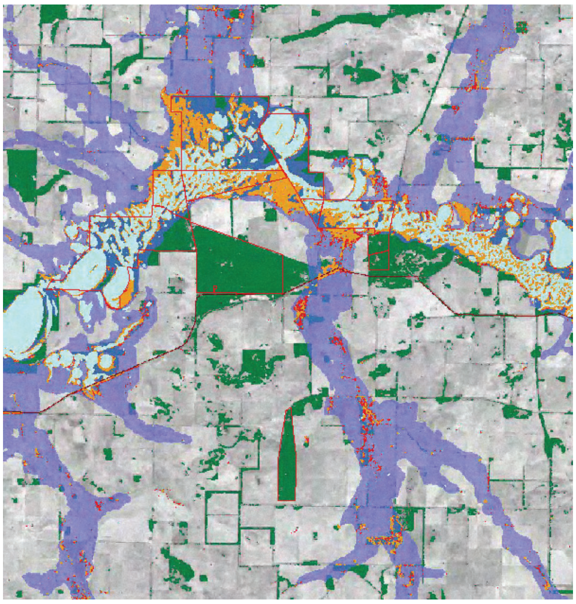


Figure 6.4 An example of salinity risk mapping for Lake Gounter Nature Reserve (red boundary lines), including a major salt lake chain (pale blue). Areas coloured in red are affected by primary salinity, those in orange are affected by secondary salinisation, areas in mid blue are at risk of secondary salinisation and areas in green are unaffected remnant vegetation.

7. VEGETATION MAPPING

7.1 Vegetation mapping in the Avon Basin

To use the ecological fire model and calculate appropriate fire regimes, it is necessary to decide on a vegetation mapping system for the Avon basin. This can best be done by grouping vegetation with similar fire responses and life attributes into fire planning units. The difficulty is in choosing the appropriate scale of vegetation mapping that is manageable at the fire management level and also meaningful in terms of biodiversity conservation.

In the south-west regions, 315 Vegetation Complexes have been mapped at 1:250,000 corresponding to all possible combinations of mapping overlays of geomorphology, landforms, soils and climate (Mattiske and Havel, 1998; Mattiske and Havel, 2004). Landform mapping was used as a basis for map units because the pattern of vegetation strongly reflects the underlying landform and soil patterns. These vegetation complexes were then agglomerated into 119 Ecological Vegetation Systems at 1:500,000 (Havel and Mattiske, 1998) primarily to reduce the number of mapping units and to allow for variations in scale of underlying landform and soil mapping units. A further simplification of mapping was done to form 26 Landscape Conservation Units (LCU) as transparent overlays to the above mapping systems (ie 1:250,000 and 1:500,000) (Mattiske and Havel, 2004). These units agglomerate various Vegetation Complexes and often comprise valleys and plateau and other sub-units which are ecologically heterogenous, but reflect recurrent landscape patterns. These large Landscape Conservation Units now form the basis for fire management planning in the forested regions (Armstrong, 2004).

Soil landform mapping has been completed for some of the Avon Basin but does not have sufficient coverage to develop vegetation complex mapping as for the south-west forests. Instead, vegetation mapping for this area has been based on the Beard pre-European mapping (1:250,000) completed in the 1970s and 1980s using aerial photography and field survey techniques, with reference to early soil surveys. The mapping linework was later refined and digitised and the data attributed into a GIS layer in the DEC corporate dataset and referred to as the Beard-Hopkins pre-European vegetation mapping. Floristic and structural details of these mapping units have been entered into the National Vegetation Information System (NVIS) developed to underpin the National Land and Water Resources Audit (NLWRA, 2001). The NVIS database allows data retrieval at six different hierarchical levels, depending on level of detail of floristic and vegetation structure required (DEH, 2003).

The different NVIS hierarchical levels of the Beard-Hopkins vegetation mapping provide a useful tool for strategic fire management, but have limitations at the finer scale of operation. Up to five species in each vegetations stratum have been entered into the database for each vegetation sub-association (level 6), which comprise the dominant structural species, and the common or characteristic flora species in the understorey. However, it does not necessarily include all key fire response species or threatened species.

The Beard-Hopkins data is the only comprehensive vegetation mapping for the Avon Basin and although it is at a large scale, will have to suffice for fire management planning for the foreseeable future. Other detailed vegetation mapping and floristic surveys have been completed at a finer scale for individual reserves in the Avon Basin but these have not been compiled into a useable format. Where this information is available, it should be used in any determination of fire cycles at the 5 year and burn prescription planning stages, but it is beyond the scope of these Guidelines to include all known species.

7.2 Ecological fire planning units

The definition of a useful ecological fire management unit in the Avon Basin is somewhat arbitrary and is determined by the scale of management planning. Setting boundaries at any scale will cause problems with managing natural populations – there will always be a proportion of a population outside the boundary that needs to be considered in overall management. This is particularly so for threatened species, where all populations of a species need to be managed as a whole to avoid having all populations burnt under the same, possibly deleterious conditions. For strategic planning purposes, the Avon Basin NRM boundary has been chosen to comply with other natural resource management and planning activities. Within the Avon Basin NRM boundary, only those areas that are within the four main IBRA sub-regions (Avon Wheatbelt 1, Avon Wheatbelt 2, Mallee 2 and Coolgardie 2) are covered in these Guidelines. All data developed for the ArcGIS project and in the various tables and Appendices in these Guidelines relate to these boundaries.

Within the Avon Basin, areas of similar vegetation, soils and landforms have been mapped into vegetation associations and vegetation systems (Appendix I and III). A vegetation system consists of a series of vegetation associations recurring in a catenary sequence or mosaic pattern and is linked to topographic, soil or geological features (Beard, 1969). Some of these vegetation systems and vegetation associations overlap into adjoining NRM regions and IBRA sub-regions which are not covered by these Guidelines.

The Avon Fire and Biodiversity Guidelines are based on using Beard-Hopkins pre-European vegetation associations as the basic planning units as they are most likely to contain vegetation and habitat that has similar fire responses and should therefore be managed in a similar way. For these Guidelines, the vegetation associations in the Avon Basin have been grouped into 9 major broad vegetation classes (eg woodland, shrubland, mallee shrubland) based on life form, structure and dominant plant height, according to Beard (Figs 7.1 and 7.2) (see also Appendix II). Wetlands are recognised as a separate class but there is no broad scale mapping of wetland vegetation. The broad vegetation classes are described in Part Two of these Guidelines with some general guidelines for fire management based on a review of relevant literature and life history attributes. A list of key species found in the vegetation associations in each sub-region and their fire responses are given in Appendix IV.

GIS analysis shows that there were about 122 Beard-Hopkins pre-European vegetation associations contained wholly or partially within the four major IBRA sub-regions within the Avon Basin NRM boundary (Appendix I). Nine of these have no area remaining uncleared while 11 have less than 20ha remaining, although there may be additional areas outside the Avon Basin boundary. These, together with 36 vegetation associations with 20 – 1,000ha remaining in the Avon Basin, have been amalgamated with larger vegetation associations that have the same dominant species and similar vegetation structure (Appendix II). Reference to the NVIS level 6 descriptions, which includes up to five species per stratum, as well as height and cover classes for the dominant species, allowed closer matching of units.

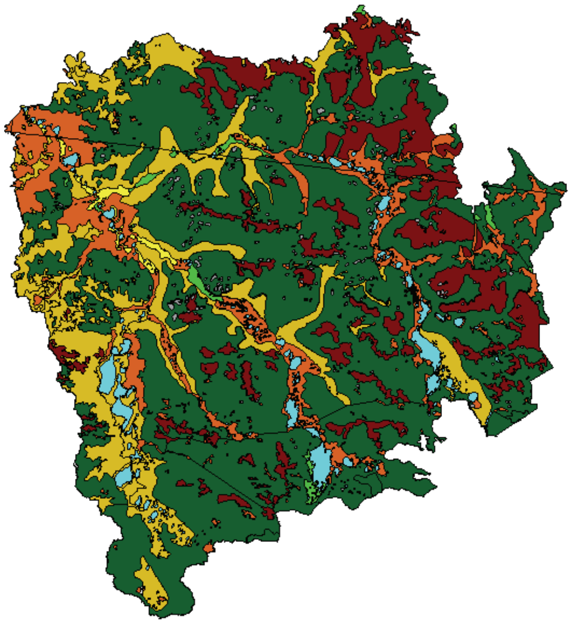


Figure 7.1 Vegetation classes for the Mallee2 IBRA sub-region showing pre-European extent of mallee shrublands (dark green), mosaics (mustard yellow), woodlands (orange), shrub heath (brown), shrubland (light green), succulent steppe (light yellow), low woodland (dark brown), salt lakes (light blue) and rock outcrops (grey).

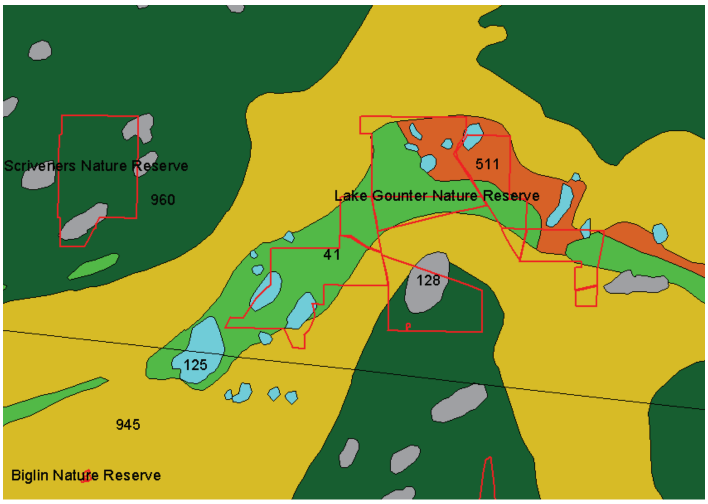


Figure 7.2 An example showing pre-European vegetation classes at the reserve level for Lake Gounter and Scriveners Nature Reserves (red boundary line). Map units are as in Fig. 7.1. Numbers are the vegetation associations (Appendix I).

Areas less than 1,000ha total remaining are considered to be too small to manage separately in a strategic fire planning process, particularly as they may occur as a multitude of small remnants or areas within remnants on private property or in reserves. Most of these minor vegetation associations within the Avon Basin are well represented outside this boundary (Table 7.1) but nine are poorly represented in WA (<15% remaining) and have more than 70% of the remaining ara located within the

Avon Basin. These nine vegetation associations may be considered as threatened ecological communities in the future and if so, then they should be managed separately rather than being amalgamated for fire management. They are currently being considered under other Avon Catchment Council projects (ND002 – Threatened Ecological Communities, or ND003 – Healthy Ecosystems).

These amalgamations resulted in 66 vegetation associations, or fire planning units, for which appropriate fire regime interval ranges can be determined for the whole of the Avon Basin (Appendix II). Further amalgamations of smaller units or separation of larger units may be necessary in the future to provide a more consistent approach to managing the vegetation. Smaller units are more likely to contain species with similar fire responses than larger units, and are preferred, but may make development of burn prescriptions unwieldy. It may be useful in the future to re-interpret the Beard-Hopkins vegetation association mapping to develop more relevant fire management units based on known fire responses of flora and fauna, and fire behaviour in different vegetation communities.

Table 7.1 Nine minor vegetation associations with <1,000ha in the Avon Basin that are poorly represented in WA (<15% remaining), and have >70% of the remaining area contained within the Avon Basin. (* This error may be due to some recent changes in GIS linework).

Vegetation Association	Pre-European area	Area remaining in WA	Area remaining in WA	Area in Avon	Area Avon/area WA
	ha	ha	%	ha	%
25	10,747	1,287	12.0	953	74.1
145	9,482	402	4.2	322	80.1
948	1,707	139	8.1	115	82.7
966	3,844	157	4.1	379	241.6*
1025	2,261	42	1.9	32	76.7
1058	11,040	296	2.7	244	82.5
1059	2,650	16	0.6	13	82.5
1065	8,878	641	7.2	448	69.8
1080	4,562	92	2.0	81	87.8

GIS technology is available now, and will be greatly improved in the near future, to allow fire managers to access and analyse relevant biodiversity data at a range of planning scales. DEC is developing a GIS interface with the WA Herbarium (NatureMap) so that any area can be queried to identify and compile a list of species, or to query the distribution of all known populations of a species from the WA Herbarium specimen data. Obviously this will not show the complete distribution of species, but it will enable correlations to be made between mapped vegetation associations and species occurrence. This GIS technology will be extremely useful for fire managers and will mean that any fire management area can be defined and queried for key fire response species. A simple correlation database already exists in some DEC regions called EndFlor which associates known threatened flora populations with various landforms to improve predictions for locating new populations (I. Wilson, personal communication).

A Fauna Distribution Information System (FDIS) is a decision support tool developed for south-west forests to predict the presence or absence of threatened and priority vertebrate fauna (Christensen, Liddelow and Hearn, 2005). It is based on an expert interpretation of known WA Museum and local records, along with the fauna habitat value of vegetation complexes, to form about 50 Fauna Habitat Types. This system allows fire managers to query an area of interest to determine the likelihood of threatened and fire sensitive fauna, and provides information on fire management needs for these species and habitats. This predictive tool is extremely useful for fire management planning but is not currently available for the Avon Basin.

7.3 Managing complex vegetation landscapes

It is evident that within some reserves or fire management units, there may be several different vegetation associations which may require different fire regimes. Burn prescriptions are prepared for logical fire management units, ie areas bounded by natural or artificial boundaries from which fire fighters can safely work, such as firebreaks, roads, major creeklines, rivers or lakes, rock outcrops or from burn buffers or recently burnt vegetation. The fire management units may vary in size from a 20ha shire reserve to a 20,000ha area of unallocated crown land east of Southern Cross. Several smaller reserves or remnants may be combined to form a larger fire management unit. Availability of resources to manage the burn and its complexity are determining factors in deciding the size of fire management units.

Where there are several vegetation associations within a reserve or fire management unit, then a decision is needed as to whether these can be managed together for fire. This will depend on what are the other vegetation associations (eg creekline, rock outcrops, heath patches), their fire history, conservation value and viability assessment. If some areas with more fire sensitive vegetation have not reached the minimum tolerable fire interval, then burning of the whole area should be deferred. If the more sensitive areas can be excluded and protected from fire, then the fire management planning may proceed past this stage. More complex burns will require greater skill and experience in planning and knowledge of fire behaviour.

In some complex vegetation communities, several burns may be required in different seasons to make best use of moisture differentials and to avoid loss of species that are sensitive to fire during winter or spring, such as orchids, nesting birds and obligate seeder species. In most cases, some areas will need to be left unburnt within the fire management unit to ensure refuge is available for fauna and species have an ability to recolonise the burnt area after fire. There may also be a requirement to provide different regimes to different components within the same vegetation association eg woodlands with heath or shrub understorey. This apparent conflict can be resolved if the woodland is burnt with low to medium fire intensity to maintain the heath without scorching or damaging the tree crowns, and then very occasionally plan for a more intense fire that will generate tree seedling growth.

Complex burn prescriptions have been prepared for areas in the south-west forests which incorporate fire regimes for a number of fire sensitive flora and fauna populations in the same burn. Objectives for protection (fire exclusion) and regeneration of threatened flora populations and fauna habitat, protection of recreational infrastructure, reduction of fuel loads and protection of karri regrowth may be included in the one burn prescription. These prescriptions may take several years to plan, and requires additional training of operations crews, and understanding and cooperation from the public.

It will be some time before appropriate fire regimes are determined for each vegetation association as much of the information required is not available, but ideally this will be provided to practitioners for preparing burn prescriptions in the future. These general Guidelines provide an initial planning framework on which to make broad strategic planning decisions but do not set out appropriate fire regimes suitable for use at the burn prescription level. In the meantime, an adaptive management approach is needed by all on-ground staff to implement a range of techniques to create variety in fire regimes, and to take advantage of all burn and wildfire events to gain experience and data on fire response and behaviour patterns in different vegetation communities.

8. ECOLOGICAL IMPACTS OF PRE-BURN PREPARATION

8.1 Firebreak construction

Firebreaks in native vegetation are constructed for several reasons:

- to prevent wildfires from entering or exiting from reserves and areas of remnant vegetation,
- to break up larger areas of vegetation to help with the suppression of larger wildfires and
- to allow prescribed burning to operate at a pre-determined patch scale across the landscape.
- to protect sensitive areas from burning too frequently
- to protect reference areas indefinitely as no planned burn areas

Fire management should always aim for a patchy mosaic at both the landscape and local scale, but the optimum scale of these patches is yet to be determined in most cases. Well placed firebreaks can be instrumental in achieving this at the landscape scale, while fire frequency and fire season, vegetation type, landscape discontinuities and weather conditions can be used to achieve a mosaic at the local scale. In general, isolated reserves should never be burnt completely in one fire, unless it is a very patchy burn that leaves many unburnt patches for fauna refuges and foraging habitat. Location of lower flammability areas such as rock outcrops, breakaways, drainage lines and wetlands may be useful to burn away from to create burn buffers but these have high sensitivity to fire and there is a risk of unplanned fire penetrating into these communities. Similarly, differential fire behaviour in various vegetation communities (eg woodland vs shrub heath) and moisture differentials in various landscape positions (eg ridge vs lower valley) can be used to break up larger fires and in some cases obviate the need for constructed firebreaks.

In many cases, wheatbelt reserves are connected with remnant vegetation on private property and a system of firebreaks at the landscape scale including private property is required as well as considering the need for internal firebreaks. These strategic landscape scale firebreak systems should to be designed in conjunction with local fire brigade members and property owners with knowledge of local conditions.

To reduce the impacts on biodiversity, particularly in smaller reserves, it would be preferable to maintain vehicle access tracks around the outside of the reserve or remnant boundary, ie in private property if this can be negotiated. In many cases, pasture and crops do not grow well for several meters outside the boundary fence due to root competition from native vegetation and this land is relatively unproductive.

Many smaller reserves in the wheatbelt region already have legal (and illegal) internal tracks that can be used to break up the area for prescribed burning, although they may not be in the most appropriate location with respect to the different vegetation types present. Numerous wheatbelt reserves do not currently have perimeter firebreaks and it may be sufficient to use existing internal tracks rather than constructing new boundary firebreaks. Where the different vegetation types can be separated using moisture differentials or other methods, and if patchy mosaic burning can be achieved, then firebreaks may not be required at all in small reserves. Each reserve needs to be assessed individually to see which tracks are useful for managing fire and whether there is a need to construct additional internal breaks to protect sensitive flora populations or fauna habitat.

In large reserves and areas of continuous native vegetation in the Coolgardie sub-IBRA region, it is preferable to break up the area into manageable 5,000 – 10,000 ha blocks to prevent very large scale wildfires, and to provide an opportunity to develop a reasonable landscape scale patch mosaic. Wind driven buffers and mosaic burning can then break up the vegetation within these blocks to provide variation in spatial fire intensity at the landscape and block scale. Smaller block sizes may be required for management of malleefowl habitat and other special requirements, but these can be superimposed on the larger block scale.

Construction of new mineral earth firebreaks has the potential to cause much ecological damage to native vegetation and must be carefully planned. Ecological impacts can include:

- loss of vegetation and fauna habitat
- loss or disturbance to threatened flora populations (Application to Take Rare Flora must be approved by the Minister)
- introduction and spread of weed species
- introduction and spread of *Phytophthora cinnamomi* dieback disease
- soil disturbance leading to soil erosion
- soil disturbance leading to nutrient release and greater weed growth
- greater access for feral predators
- greater access for the public and increased risk of accidental fire, dumping of rubbish, removal of firewood and disturbance to native fauna populations

However, there are significant risks in **not** planning and constructing internal firebreaks in suitable positions, such as having the whole reserve burnt out in a single fire, or having a bulldozer construct a temporary fireline during a wildfire without proper environmental assessment. This latter scenario is one that should be avoided at all costs, and particularly in small reserves, as it takes many years for the vegetation to recover, if at all, when the topsoil has been removed (Hopper, 2000). There is usually insufficient time during a wildfire event to check for the location of threatened flora populations or adhere to dieback prevention protocols or avoid spreading weed seeds throughout the reserve or avoid sensitive habitats before attempting to halt a fire with a bulldozer. It is strongly recommended that firebreaks are designed, constructed and maintained in advance with all environmental protection processes in place and located where they will provide the optimal protection and minimal disturbance to sensitive areas. Fires should then be allowed to burn out to these firebreaks and back burning used where necessary to contain more intense fires. Control lines can readily be constructed in adjacent paddocks with minimal effort and little disturbance to the soil using agricultural machinery.

Various types of firebreak can be implemented that have significantly less impact on the environment than a bulldozer or grader track. Some of these have been trialled on a small and large scale, including a raked firebreak, scrub rolling, slashed vegetation strips and buffer burning. The advantage of a raked firebreak is that it provides a mineral soil break and removes much of the combustible litter without massive disturbance to the topsoil. In woodlands, where fire does not readily carry, this may be all that is required to prevent the spread of wildfire. A raked firebreak will not encourage greater public access but will stimulate the germination of existing weeds, which will need to be managed. Raking may also stimulate the germination of some native species in need of regeneration, simulating native fauna digging and soil turnover disturbance. Further experimentation is needed to find the most efficient machinery for this task and the most suitable width for different vegetation types. These alternative firebreak types need evaluating both for fire management effectiveness and ecological consequences.

Modified vegetation strips have been trialled around Lake Magenta and Dunn Rock Nature Reserves and Unallocated Crown Land (Woodman Environmental Consulting, 2002; Mitchell Davies, personal communication). A 40m wide fuel reduction strip was constructed covering about 70km around sections of each reserve boundary, in various vegetation types other than woodland. This involved the chaining and subsequent burning of these strips to protect the reserves and surrounding private property and helps contain the spread of wildfire. Burning of the vegetation needs to be within a few weeks of chaining, before seed is released from woody capsules (McCaw and Smith, 1992). Some species release seed in response to dessication soon after chaining or scrub rolling and these may be incinerated in the subsequent fire. Chaining allows more complete burning of the vegetation in the buffer strip and reduces flame height so the fire can be more readily controlled. This buffer strip has not been fully tested by wildfire although there were some minor escapes during the prescribed burn operation into adjacent mallee and heath (M. Davies, personal communication). Slashed buffer strips have been utilised in many other places and have proved to be effective in reducing the spread of wildfire (R. Armstrong, personal communication). One disadvantage of this method is that large areas of vegetation are disturbed and these need to be surveyed for threatened flora populations which can be expensive. The vegetation does regrow and it may be 20-30 years before the vegetation can be chained again without risking loss of species. There is also the potential for spreading dieback disease and weeds around the perimeter of reserves by the heavy machinery. However, it may be necessary to accept some loss of vegetation or species in the slashed buffer as a ‘sacrificial lamb’ to protect the remaining vegetation.

At Dryandra Woodland, prescribed burn buffer strips have been used since 1985 to provide protection of remaining vegetation from wildfire (Department of CALM, 1995). These strips are 50-100m wide along major roads and tracks and use wind driven burns planned to burn a short distance from the road before self extinguishing. The buffers only are burnt in spring or autumn when fuel loads reach 7-8tn/ha, but there is greater risk of larger areas being burnt from the autumn burns. Alternative sides of the road are burnt each time to reduce the frequency of fire on sensitive flora. Perimeter burns are not practiced due to the risk of weed invasion from adjoining properties. This technique is strongly recommended for larger reserves in the Avon Basin, especially in the Coolgardie sub-region along roads and constructed firebreaks to widen the fuel reduced buffer area, while leaving the remaining vegetation unburnt for significantly longer periods.

Another method used where there are extensive areas of largely inaccessible native vegetation is wind driven buffers or strips burning using aircraft. Fire is ignited at a point or line and allowed to run with the wind to a secure edge, for example vegetation that has been burnt in the last 4-5 years. These strip burns are best started in the early evening and in spring or late autumn after rain so there is greater chance of the fire going out overnight. However, there is a significant risk that even a slight change in wind speed or direction can change this benign strip into a large head fire and burn out large areas (Barrett *et al.,* 2004). This method has many ecological benefits in that it does not cause any soil disturbance or encourage the invasion of weed species or dieback disease, it does not require expensive firebreak construction and the strip can be located in different areas over the years allowing flexibility to create mosaics of burnt vegetation. It can be applied by fixed wing aircraft or helicopter and the fire intensity and direction can be fine tuned on the day according to prevailing weather conditions. Obviously ground crews are required to monitor the progress of the fire and control any unwanted outbreaks, but with experience, this method has the potential to have the least ecological impact and the greatest benefit in terms of creating landscape scale burn mosaics.

Where weeds are prevalent around the perimeter of smaller reserves, or parts of reserve boundaries, it may be appropriate to simply spray a firebreak with herbicide. This may need to be repeated on an annual basis in Sep-Oct, before most annual grasses set seed. This method will have the added benefit of reducing the spread of weeds and minimising soil disturbance and is regularly used in the plantation industry. Care is needed to avoid spray drift into the reserve or pollution of waterways and to select chemicals that have minimal impact on nearby native vegetation. This herbicide technique should only be used in highly disturbed vegetation and after on-ground assessment of any conservation values at risk.

Following on from the discussion above is the consideration that if areas of native vegetation are burnt in a patch mosaic, then there is less need for regular firebreaks as there is less potential for large uncontrolled wildfires to develop. This should apply in small and large reserves and it is recommended that some experimental burns be conducted to test this hypothesis.

8.2 Weed control

Weed seeds are dispersed by wind, water, birds and fauna, including sheep and rabbits and different distribution patterns may be observed. Areas that have been grazed by sheep will have pasture weeds (eg annual grasses, capeweed, radish) dispersed throughout while rabbits will spread weeds mainly around the edges and in sandy areas where they burrow. Birds can carry seeds long distances and cause sporadic outbreaks eg bridal creeper, non-native *Acacias* and asparagus fern. Weeds in reserves are often found growing along tracks and drainage lines where the soil has been disturbed and where moisture is more available, and where garden refuse has been dumped. Reserves and remnants with a high perimeter to area ratio that are surrounded by farmland are most prone to weed invasion.

The best defence against weeds is to prevent their establishment, so management must aim towards controlling grazing by sheep, goats and rabbits, limiting access to vehicles, limiting soil disturbance and unnecessary recreation operations and preventing illegal dumping of domestic and agricultural rubbish, particularly garden refuse. Fencing, gates, bollards and signs may be required to alert landholders in the neighbourhood and the local community to help prevent weed invasion.

Monitoring of weed populations before and after prescribed burning can provide much needed information about the response of many weed species to soil disturbance and to fire in different vegetation communities, and should be an integral part of all fire management operations. The distribution, cover and species of weeds need to be recorded and preferably mapped using aerial photography or GIS mapping tools (Brown and Brooks, 2002). Some species of weeds are highly invasive (eg English broom, tagasaste, bridal creeper) while others are more benign and take decades to establish more than a few plants (eg some eastern states *Acacias*). By knowing these characteristics of invasiveness for different species in different areas, a system of prioritising weed management can be developed. Similarly, knowledge of the fire response and life attributes of weed species (ie seeder / resprouter, juvenile period, soil or canopy stored seed, seed bank viability, dispersal method) can be used to find a weakness in their life cycle when fire of another form of weed control will be most effective in reducing population size.

A common means of weed dispersal is by soil movement during firebreak preparation, either by grading firebreaks and drains or pushing dangerous trees 20m into the reserve before or after a burn. All weed propagules, such as seeds, corms, tubers and rhizomes can be spread by soil movement, while the soil disturbance itself stimulates nutrient release from the soil and germination. Prevention of weed spread requires a number of management actions, similar to the prevention of jarrah dieback disease:

- mapping the location and cover of weed populations along existing tracks and firebreaks, and any proposed firebreaks;
- controlling adult weeds prior to seed set in the year before firebreak maintenance is proposed at the most effective time for the chosen method of control;
- ensure graders follow soil hygiene precautions as if they were entering or leaving a *Phytophthora cinnamomi* area, to ensure they are not spreading weed propagules beyond the existing populations. Methods of cleaning machines need to be developed for the wheatbelt as has been done for *P. cinnamomi* control in the south-west and south-coast regions. Dry soil conditions are preferred;
- further weed control measures are usually required following track maintenance as soil stored seeds will be stimulated to germinate or resprout.

Depending on the timing of track maintenance with respect to the burn operation, this flush of new growth of weeds could be eliminated by the burn itself;

- follow up monitoring and weed control are required in infested areas for several years post-fire to prevent further spread of weeds and to encourage the native plants to regenerate and fully occupy the space;
- weed control is also important following any soil disturbance associated with wildfire suppression. The use of emergency tracking in conservation reserves is not recommended and fire should be allowed to burn out to an established track or firebreak.

Perimeter firebreaks are particularly prone to annual weeds, being adjacent to agricultural paddocks. Weed seed is dispersed into reserves by wind or water movement or by rabbits in their dung for 10-20m into the native vegetation. Firebreak maintenance then spreads these seeds further around the edge of the reserve on machinery. Where there are existing internal firebreaks or tracks, these may be preferred from a weed management perspective, unless regular weed control along perimeter firebreaks can be implemented. However, internal firebreaks have the potential to bring weeds into the centre of the reserve which is undesirable, but with good hygiene, this means of dispersal can be minimised. Internal firebreaks also encourage greater recreational use by the public. Where weeds are prevalent throughout the reserve, then it is preferable to maintain any existing perimeter firebreaks than construct new internal ones, or use other fuel reduction techniques such as fuel reduction buffer strips (see above). At Dryandra Woodland, buffer strips are only used along internal tracks and roads because of the high risk of weed invasion along boundary fences.

Where weeds are invading from an adjoining property along tracks, firebreaks or drainages lines, then direct control of weeds using herbicides is recommended. Hobbs *et al.,* (1992) observed in York gum woodlands, that both exotic and native species varied in their responses to herbicides depending on the type, concentration and timing of application. Weed abundances returned to pre-treatment levels in the year following application of the residual herbicide Simazine®, with no increase in native species richness or cover. Chemicals may also be applied by stem injection, basal bark spray or painting on cut stems to reduce the impact on native plants. Use of herbicide is only one method of control, and there are many effective means of physical control including hand pulling, smothering and felling of adult plants. In some cases, weed species can be confused with local native species and great care needs to be taken to correctly identify the species being controlled. A number of herbicides have been registered for use in native bushland and there are several good references on methods of weed control in bushland (Hussey and Wallace, 2003; Brown and Brooks, 2002). However, most herbicides have been manufactured for agricultural situations rather than use in native vegetation, and the risk of damage to non-target native species can be high.

8.3 Soil erosion control

There are many instances of soil erosion occurring as a direct result of firebreak construction around the perimeter of a reserve, with little consideration to the slope or soil type. Within a few years of firebreak construction the erosion gullies have deepened to an extent that the firebreak is no longer trafficable. In drier areas wind erosion is also a concern where light sandy soils are cleared of all native vegetation and the loose sand causes sandblasting to adjacent vegetation. Soil erosion on fire breaks and tracks can lead to siltation of waterways and damplands which can impact on stream flow and sensitive wetland flora and fauna populations including frogs and invertebrates.

Water erosion can occur on all soil types where run-off occurs and soil is detached from unstable aggregates. Medium textured clays (10-35% clay), loose surfaces and unstable aggregates are highly susceptible to water erosion. Run-off can occur when the slope angle, length and surface condition are sufficient to encourage surface water to move, and when rainfall intensity exceeds the infiltration rate (Grealish and Wagnon, 1995). Hard setting surface soils and rocky outcrops are more prone to run-off than well structured soils. Slopes of 3-8% are likely to encourage run-off and should be avoided where possible for firebreak construction. In addition, safe disposal of water run-off must be carefully considered, as it will carry soil particles, weed seeds and possibly dieback propagules and may pollute the discharge area. The construction of cut-off drains to reduce the flow of surface water involves more soil disturbance and drains may be taken up to 15m into the native vegetation. Firebreaks, tracks and drains all need maintenance from time to time and each run will cause additional soil disturbance and opportunity for weed and disease infestation.

To avoid soil erosion it is recommended that land managers:

- only construct firebreaks where the risk of soil erosion is low (ie slopes < 3%, away from rock outcrops, soil type is neither light sand, medium clay or hard setting surface);
- relocate the firebreak so it avoids going over soils and slopes prone to erosion (ie use internal firebreaks where necessary);
- prevent siltation by leaving tracks ungraded for 50m either side of creeks and wetlands;
- use alternative methods of fuel reduction (eg slashing, raking, buffer burn strips) in areas prone to soil erosion.

Some shires are moving away from compulsory firebreaks, other than around buildings, because of the high risk of soil erosion and the environmental damage that firebreak construction has caused. Less damaging means of controlling fire spread should be sought wherever possible.

8.4 Disease control

Plant diseases caused by fungal pathogens (*Phytophthora* spp., *Armillaria* and stem canker) are common in the Avon Basin, particularly when the vegetation is under stress. These diseases have the capacity to cause loss of plant species diversity and abundance, changes in floristic structure as well as loss of food resources and habitat for fauna (Wills, 1993; Department of CALM, 1995). Approximately 40% of the flora in the South West Botanical Province is susceptible to *Phytophthora cinnamomi* and 14% (800 species) are highly susceptible (Shearer *et al.,* 2004). Soil borne fungal pathogens, *Phytophthora* spp. and *Armillaria luteobubalina* (Honey Fungus) may be spread during soil disturbance activities, particularly track and firebreak maintenance. Tracks and drains direct surface water flow into reserves which may carry the fungal propagules. Drains constructed with tracks and firebreaks can also change the soil moisture conditions downslope and provide more favourable conditions for sporulation.

P. cinnamomi is most prevalent and damaging in the south west of WA where annual rainfall exceeds 800mm, and is widespread though not extensive in areas receiving 600-800mm. In the Avon Basin, where annual rainfall is mostly less than 600mm, climatic conditions are generally not conducive to the spread of *P. cinnamomi.* However, the disease may survive under certain conditions, eg after heavy summer rainfall or around granite outcrops in seepage or runoff areas, and localised patches of vegetation can suffer severe damage with intervals of recovery between drier periods. *Phytophthora cinnamomi*, *P. citricola* and *Armillaria luteobubalina* infections have been recorded in or near to Dryandra Woodlands with mean annual rainfall of about 500mm (Department of CALM, 1995; Sage *et al*., 2004). Sage *et al.* (2004) found eleven species affected by *P. cinnamomi* in four infestation sites in the Narrogin district. All infestations were located on water gaining sites, (ie along a water course, a drain or near a dam) or where there had been high disturbance in areas that were also low in the landscape. There is no record of *P. cinnamomi* establishing in natural ecosystems receiving less than 400mm annual rainfall (Podger, 1996).

Other exotic *Phytophthora* species, including *P. citricola* have been recorded in the Avon Basin and have the potential to cause sporadic outbreaks of disease. The extent of these other diseases and the conditions for their survival and spread in the Avon Basin are not well known. *A. luteobubalina* is a naturally occurring fungus with a large host range that spreads by root to root contact in undisturbed environments. Although the areas of infection are generally smaller than for *P. cinnamomi,* the environmental impact can be very high. Differences in susceptibility to fungal diseases are evident both between and within species. For example, *Eucalyptus wandoo* is not susceptible to *P. cinnamomi,* but is highly susceptible to *A. luteobubalina*. Marri (*Corymbia calophylla*) is resistant to *P. cinnamomi* but is susceptible to *P. citricola* which can cause extensive lesions and persist under dry soil conditions (Shearer and Tippett, 1989).

To manage fungal diseases during pre-burn preparations, strict disease hygiene practices need to be adhered to for any work requiring soil movement. Spores survive well in moist soil and any movement of soil can spread these fungal diseases. Surface water movement can also spread the zoospores and construction of road drains can exacerbate this means of dispersal. Risk assessment procedures need to be followed before any soil movement or drainage works are conducted, including firebreak maintenance and construction of cut-off drains or culverts that redirect water and create more favourable soil moisture regimes for survival of *P. cinnamomi*.

Departmental guidelines, protocols and training are already available for this and will not be duplicated in these Guidelines. However, a few comments can be made in relation to fire management:

- Prepare a DEC Dieback Management Plan prior to any works involving soil movement and drainage of water;
- Assess risk of disease spread and impact to susceptible vegetation and fauna habitat;
- Assess risk of disease to susceptible threatened flora populations;

- Demarcate infested areas and control access to these areas;
- Ensure all machinery is clean on entry to all work sites, reserves and state forests to remove soil and root material (this applies to weed seeds as well as to fungal disease propagules);
- Develop an efficient means of cleaning machinery in isolated areas;
- Treat water points used in fire suppression and track maintenance with fungistats;
- Work under dry soil conditions only;
- Avoid directing drainage water into susceptible vegetation;
- Use only uninfested and certified raw materials for earthworks, and not taken from gullies;
- Permanently close illegal and unwanted tracks and control public access into susceptible areas;
- Close any tracks that pass through inundated, infected or high risk areas, particularly when the risk of disease spread is high (eg after summer rainfall);
- Vehicle access tracks across boggy creeklines should be avoided or built to provide a hard all weather surface and allow natural drainage;
- Granite outcrops, seepage areas, wetland and riparian vegetation are particularly at risk of infection and disease and require extra care and protection;
- Monitor disease spread in infested areas and monitor uninfested areas of susceptible vegetation following earthworks;
- Report and map any new disease outbreaks and diagnose the cause;
- Develop a strategic disease management plan for the Avon Basin.

These guidelines for preventing disease (and weed) spread should be followed in the Avon Basin, even though most areas are considered to be at low risk from *P. cinnamomi*. Other fungal diseases may be more tolerant of drier conditions and a precautionary approach is required. Greater awareness of the risks and causes of disease and weed spread, and training of road maintenance operators and supervisors, should minimise the potential for serious environmental damage.

8.5 Feral animal control

Pre-burn preparations should always include an assessment of the need for feral animal control in areas to be burnt and in adjoining buffer areas. Following a prescribed burn or wildfire the risk of predation is increased due to greater access for predators and reduced protection of native fauna. Many native fauna may also be weakened following a fire due to lack of food or greater distances travelled to obtain food resources, as well as increased exposure to the weather, and this will leave them more prone to predation. Control of feral animals is a critical component of fire management in the Avon Basin and can significantly improve the chances of survival and dispersal of native fauna following fire events.

Eleven mammals in the critical weight range have become locally extinct in large areas of the Avon sub-regions (Beecham, 2003; Beecham and Danks, 2003) and the management of re-introductions will depend to a great extent on providing suitable habitat free from fox and cat predation. There are another twelve mammal species that are listed as threatened or conservation dependent. Management of these species will require a program of fox and cat control combined with judicious habitat regeneration of vegetation to ensure a range of fire ages, with patches of vegetation suitable for foraging, nesting and shelter within safe distances.

Feral fox and cat numbers need to be controlled in areas known to have threatened fauna populations up to one year before a prescribed burn, and at least two years following a burn or wildfire event. In areas that are already baited, consideration should be given to increasing the frequency of baiting from once a year to four times a year to ensure adequate protection. Hand baiting will be required in small reserves while aerial baiting can be applied in larger reserves. Community baiting programs and fox shoots need to be coordinated with prescribed burning wherever possible to maximise effectiveness.

Rabbits need to be controlled before and after prescribed burning, especially in smaller reserves, as they can cause considerable damage to regenerating shoots and seedlings of native vegetation. Control requires location of rabbit populations and burrows and is most efficiently carried out with the cooperation of adjoining landholders.

Some control of kangaroos and wallabies may be required prior to a prescribed burn, especially in smaller reserves. Timing of the burn may help to alleviate this impact, where an autumn burn will stimulate new growth in winter and spring when there is an abundance of other fresh green vegetation, whereas a spring burn will stimulate new growth in summer and autumn when there are few alternatives and be more attractive to grazers.

Monitoring of feral animal populations is required to assess the likely impact of feral animals on fauna and flora populations as the risk may be minimal in some areas. Monitoring of threatened species is also required to assess the effectiveness of any feral animal control program implemented. Guidelines for methods and protocols for feral animal control and monitoring are available from the Department of Environment and Conservation and need not be repeated in these current Guidelines.

9. MANAGEMENT AND RESEARCH GAPS

Many research, information and management gaps have been identified in these Guidelines that need to be addressed before there can be efficient and effective ecological fire management in the Avon Basin. The gaps have been grouped into three levels (High, Medium and Low) according to the perceived need by planners, researchers and practitioners during general discussions and public and project team meetings. They have been simply listed as action items in these Guidelines, and need further discussion and development to be taken to the next stage. Monitoring gaps are discussed in the following section. There are also gaps in training, education and community engagement which are expected to be addressed in the follow up to this project.

9.1 Management gaps

HIGH

- Interpret Landsat imagery for last 30 years to construct fire history spatial database (season and intensity)

- Access to NatureMap in GIS project for all WA Herbarium specimens
- Define fire management units for whole area including individual reserves, groups of smaller reserves and remnants, and larger blocks in continuous vegetation
- Encourage wider debate within DEC and the community as to what extent fire management planning should cater to the requirements of individual populations or species
- Develop Master Burn database with attributes for each fire management unit for tenure, area, fire history (frequency, season and intensity), vegetation associations, threatened flora and fauna habitat, TECs, condition (weeds, salinity, grazing and disturbances)
- Provide the ArcGIS project developed for these Guidelines to all DEC and FESA districts and fire control officers with training
- Develop and implement the prioritisation system to determine areas in greatest need for prescribed burning
- Develop a standardised system for monitoring ecological fire management objectives
- Develop closer links with DEC Science staff to design monitoring programs and develop systems to collate, analyse and interpret monitoring data
- Conduct annual regional and district reviews of fire management to present monitoring and research data, review planning processes and share fire management experience. Discuss how new information can be used to improve existing procedures
- Conduct trials using perimeter buffer burns and wind driven buffer strips in a wider range of vegetation types and monitor results
- Develop techniques for creating different scale fire mosaics in a range of vegetation types and ages
- Develop a strategic planning structure or team including DEC, FESA, ACC, community and local government and bush fire brigades
- Employ a dedicated GIS fire planner and database manager for the region
- Develop management processes by which these Ecological Fire Management Guidelines are integrated with the Wildfire Threat Analysis
- Develop a system to capture field verification and changes to vegetation association types and boundaries
- Develop a system to capture spatial information on weed infested areas, including weed species, declared or environmental weed species, total area affected, invasiveness and response to fire
- Conduct trials to control kangaroo grazing in isolated reserves and measure the regeneration response with and without fire

MEDIUM

- Develop maps for each shire showing strategic fire breaks and water points for all fire management units in DEC managed land
- Conduct prescribed burns in some appropriate nature reserves near major towns to demonstrate willingness of DEC to tackle the problem
- Conduct field days and seminars for bush fire brigades and the community to raise the awareness of the need for ecological burning, and the planning, objectives and techniques used to achieve ecological outcomes
- Determine final boundaries for regional fire management planning and resolve difficulties of managing Avon Basin across various DEC boundaries

- Develop a simple system to rate DRF in terms of fire sensitivity (colour coded) based on known or likely fire responses, and include in wildfire threat analysis to assist fire fighters during wildfires
- Develop robust approvals processes for taking some populations of DRF during burn operations with clear monitoring objectives
- Continue to improve threatened flora database to include fields for fire responses of species and population monitoring information so that all populations of one species can be queried at one time to assist fire planning
- Conduct trials using harrows or similar to make temporary firebreaks through smaller reserves to separate fuel types or leave some areas unburnt
- Collate all detailed flora survey information and enter into a GIS layer that can be accessed for fire management
- Collate all research sites, quadrats, survey sites and permanent monitoring sites into a spatial GIS layer for use in fire planning
- Select areas to be left as unburnt reference sites in each vegetation association and across the rainfall gradients and enter into GIS and Master Burn databases
- Incorporate Woodland Watch sites and survey data into the ArcGIS project for fire management
- Incorporate all Birds Australia survey data (species lists for sites) into an ArcGIS layer for monitoring responses to fire
- Incorporate all Land for Wildlife and covenanted areas of private bushland into an ArcGIS layer for fire management
- Develop a GIS layer showing areas baited under the Western Shield program and areas baited by private landholders
- Develop an ArcGIS layer showing fresh and brackish wetlands and riverine systems with viable fringing vegetation or useful waterbird habitat that needs to be protected from fire or managed to regenerate vegetation
- Develop a database (eg Endflor) that has DRF populations and vegetation associations that can be queried to predict which vegetation associations are likely to have particular DRF species to improve efficiency of searching for new populations prior to burning
- Expand the DEC district Reserves database to include assessment of viability and conservation value for fire management which can be rolled up to the corporate database. Requires spatial weed mapping, assessment of salinity and soil disturbance, grazing pressure as well as confirmation of DRF populations and assessment of known and suitable threatened fauna habitat
- Assess whether vegetation associations with small areas remaining in the Avon Basin (eg<1,000ha) constitute threatened ecological communities and whether they have particular requirements for fire management

LOW

- Develop a self assessment brochure for private landholders to assess the condition or viability and conservation value of their own remnant vegetation
- Database areas of privately owned remnant vegetation with medium to high viability where consent is given by owners
- Develop fire management guidelines and brochures specifically for private landholders and bush fire brigade members who want to burn and monitor ecological outcomes

- Consider using a zoning of fire management units with areas closer to community assets being burnt at the lower end of the tolerable fire interval range
- Train staff who prepare burn prescriptions how to recognise different vegetation associations in the field and to verify and map boundaries between them to improve fire management and update database records
- Document and demonstrate differences in floristics within each Beard-Hopkins vegetation association (eg salmon gum woodlands) so that with experience, areas of each vegetation association can be managed according to the species present rather than by a set prescription.

9.2 Research gaps

HIGH

- Fire responses of all declared rare and priority flora. Provide estimate if unknown and qualify entry in database
- Conduct targeted surveys of bradysporous species and key fire response species after fires and prescribed burns to update fire response database
- Research fire responses of threatened and declining fauna species, time to successful breeding, rates of recolonisation
- Conduct surveys of older vegetation of known fire age to assess senescence and natural recruitment of key indicator species and collate in life attribute database
- Develop techniques to recognise and define the start of senescence in different communities (. Visually assess areas
- Develop fauna habitat mapping of known and predicted suitable habitat for threatened and declining species using known sightings and museum records
- Understanding optimal habitat patch size of burnt and unburnt areas for threatened, declining and poorly mobile fauna
- Develop a system and statistic to measure patchiness of fires
- Determine whether patchiness of a burn is retained in subsequent burns and if so under what conditions
- Conduct trials using aerial ignition in cooler weather to develop techniques for creating patchy burns in older age vegetation and a range of patch sizes
- Assess different responses to fire (time to first flowering and seed set) of vegetation communities in the medium and low rainfall regions
- Conduct trials to determine if fire can be used to control the spread of some weed species and develop a fire response and life attribute database for weed species
- Conduct trials and surveys of reserves to assess impact of native fauna grazing pressure on regeneration of flora species. Develop a system to estimate grazing pressure (eg kangaroo counts, defoliation, dung pellets/m^2) that can be used by district staff to determine if populations of native fauna should be culled to allow regeneration of vegetation

MEDIUM

- Develop a fauna habitat description system for mapping and database so that suitable habitat for threatened fauna can be protected and restored
- Greater understanding of natural fire frequency in granite outcrop communities and *Acacia/Allocasuarina* shrublands in semi-arid areas

- Conduct trials around granite outcrops to burn surrounding vegetation but protecting the fringing communities
- Conduct prescribed burning trials in reserves and selected privately owned remnant vegetation to demonstrate techniques and ecological outcomes using different fire regimes and leaving unburnt control or reference areas
- Conduct trials around sections of salt lakes and drainage systems affected by various levels of salinity to determine if burning can assist with regeneration or cause further deterioration of fringing vegetation
- Develop a listing of all declining species in the region and their preferred habitats
- Assess DRF populations and threatened fauna populations in terms of how rapidly they are declining due to absence of fire or inappropriate fire management
- Develop the use of indicator bird species for monitoring responses to fire (abundance measures, time to recolonisation and successful breeding) and incorporate as standard monitoring technique for prescribed burns
- Further investigate the use of other focal or indicator species to monitor fauna responses to fire (reptiles, invertebrates, mammals)

LOW

- Conduct surveys of landholders, bush fire brigades and community members to collate any local information about fire history for various reserves and remnant vegetation in each shire of the Avon Basin. Approximate time of fire would be useful (eg <5 yrs, 5-10 yrs, 10-20 yrs, 20-30yrs, 30-50 yrs, >50 yrs)
- Assess correlations between viability, landscape position, proximity to townsites and vegetation associations by sampling various reserves to determine if viability can be predicted (salinity, weediness, grazing, disturbance).
- Develop a descriptive system to predict fire behaviour in semi-arid vegetation communities based on vegetation structure (strata and cover), dominant species, habit, litter depth and continuity of fuels in different layers and relate this to age of vegetation for different vegetation associations.
- Develop a simple visual system for estimating time since last fire using visual clues such as charring of bark, height of seedlings, diameter of sapling cohorts, depth of litter, proportion of senescent or dead shrubs

10. ECOLOGICAL MONITORING GUIDELINES

10.1 Monitoring of fire planning objectives

All fires, including prescribed ecological burns, community protection burn, fuel reduction buffer burns and wildfire, are opportunities to learn about plant responses and fire behaviour. This requires that appropriate measurements are taken to determine the effect fire is having in different vegetation communities and under different burn conditions. This monitoring is done to assess if **ecological fire management objectives** are being met. An assessment is also required to know if the **prescribed burn objectives** (ie area, intensity and timing of a prescribed burn) were achieved, which is really performance evaluation rather than monitoring. These are quite different management objectives which require a different set of measurement indicators.

Monitoring the ecological effect of fire, or its absence, should be done at the regional and landscape levels as well as the species and community levels. Different techniques are used for different scales of monitoring and it is important to determine which scale is most relevant to the monitoring task. For example, remote sensing and GIS techniques using corporate datasets can be used to measure the frequency distribution of fire age classes for each broad vegetation class and vegetation association before and after a burning season, and comparing this to a preferred or theoretical distribution. Whereas for declared rare or priority flora, changes in population demographics are measured at the quadrat level over time to determine species responses to fire.

The key to effective monitoring is to set clear management objectives and match these to indicators of change that are specific, measurable, achievable, relevant and timely (ie 'SMART'). For example, a prescribed burn objective may be to treat no less than 1,000ha and achieve fire intensities >1,000 kW/m over 50% of the area. Another prescribed burn objective for a number of burns may be to measure rates of spread and patchiness of burn in dryandra heath greater than 15 years old in the 300-400mm rainfall region. An ecological fire management objective may be to improve the rock sheoak habitat and thereby increase the population numbers for red-tailed phascogale by 20% in 10 years.

Indicators used to measure objectives must be sensitive to the changes expected. The use of surrogates as indicators for measuring ecological objectives, such as using focal species and habitat complexity scores for measuring changes in biodiversity or species richness, provide useful information **only** if the relationship between the surrogate and the objective is known and has been adequately tested in the environment being monitored.

10.2 Key evaluation questions

Most fire management actions should be monitored at some point to know if the objectives are being met and if the resources are being used in the most effective manner. This is particularly so where new management actions are being planned or where previously accepted management actions are being implemented in new areas or with untested vegetation types or species. In these cases, fire management planning should include a monitoring program which is designed to answer **key evaluation questions** to assess if the management objectives for the prescribed burn are being met. These evaluation questions need to be carefully developed and methods can then be selected that will best answer the questions.

Some examples of evaluation questions for **prescribed burn objectives** might be:

- What was the rate of spread of the fire in the different vegetation types?
- What was the fuel load before and after the fire?
- What were the weather conditions during the fire compared with predicted weather conditions (wind speed/direction, temperatures, humidity)?
- How much area was left unburnt and what area burnt out completely?
- What was the level of patchiness in the whole burn area?

- Did the areas planned to be protected remain unburnt?
- Did the area burn in the expected time frame?
- Did the reduced fuel buffers contain any outbreaks from the burn?
- Were the resources employed adequate to manage the burn as planned?

Some examples of evaluation questions for **ecological fire management objectives** might be:

- Which plant species have been reduced by more than 50% as a result of a particular prescribed burn?
- Which plant species responded to the burn by resprouting or by reseeding?
- How will the age distribution of a vegetation association change after the burn?
- Which vegetation associations have less than 25% of their total area in the juvenile seral stage?
- What are the relative changes in plant species abundance?
- How has a sub-population of declared rare flora responded to the burn?
- What is the longest time for a species to set seed after the burn?
- What are the changes in vegetation structure with time after the burn?
- What are the changes in vegetation cover after the burn?
- Has the burn reduced the weed abundance?
- How has fauna abundance been affected in the burn area?
- Have any fauna utilized the refuge areas?
- What fauna species were able to recolonise the burnt area?
- How many years after the burn before malleefowl use the area for nesting?
- How has the fire affected the habitat values of red tailed phascogales?
- How has the fire impacted on watercourses in the burn area?

10.3 Adaptive management

Monitoring is most useful when there is an open-minded approach to management and a willingness to take on new information and trial different techniques. This is the **cycle of adaptive management** ie planning, action, monitoring, evaluation and adjustment (Fig. 10.1). Evaluation involves more than just data analysis and interpretation, it involves making decisions based on that information. Effective monitoring and adaptive management at the local level encourages staff and land managers to become more involved in activities and interested in achieving the desired outcomes. It creates a sense of 'moving forward' together.

Unfortunately, monitoring is often seen as a waste of time and resources by managers (Possingham, 2001), and as unreliable by the research community. Some monitoring has been implemented for interest only, and which cannot influence management actions, ie monitoring has not been designed to measure clear management objectives. Monitoring design is also often inadequate to capture the natural variability of systems, particularly over large landscapes. The data must be robust with adequate statistical power to give managers confidence to change their management actions (Possingham, 2001). The reliability of the monitoring program becomes critically important where the impact of different fire regimes on threatened species is being assessed.

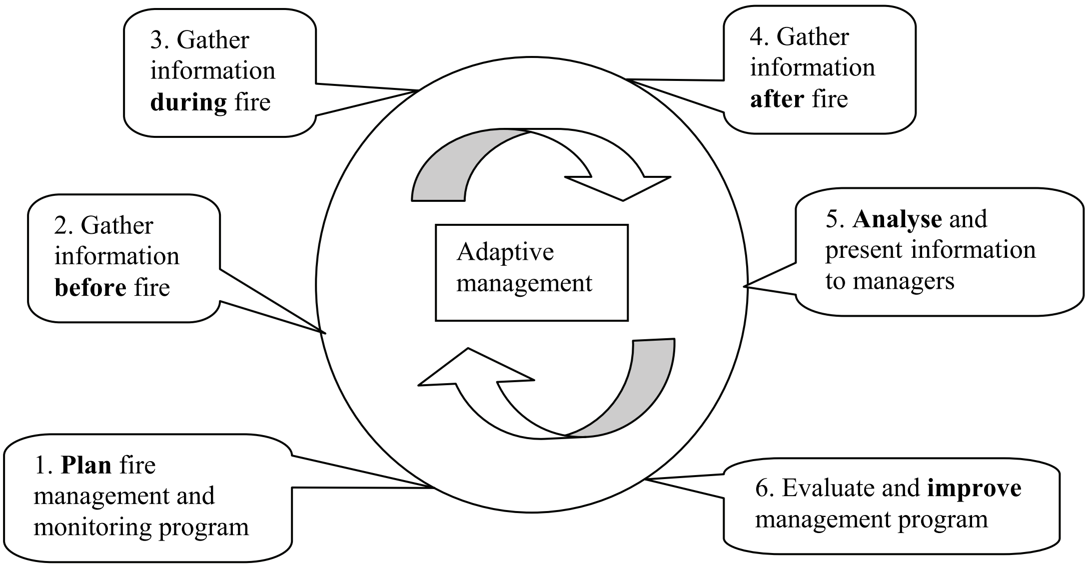


Figure 10.1 The adaptive management cycle involves planning, action, monitoring, evaluation and adjustment (after SEQ Fire and Biodiversity Consortium, 2002)

10.4 Fire research

If scientists and statisticians provide the right advice and training to operational staff to develop reliable monitoring programs then they can have more confidence in the results and use the monitoring trends and information to determine which aspects of fire management require more detailed research effort. Scientists need to ensure that:

- monitoring programs are designed to answer key evaluation questions,
- appropriate methods are selected to match the scale of the objectives,
- adequate number of samples are used to detect trends in population size
- local land managers and DEC officers are trained in plot establishment,
- data is collected, analysed and reported using appropriate techniques,
- information is interpreted in a logical way that is useful to fire managers.

This being said, it is as incumbent on managers and DEC officers to understand the importance of monitoring programs and to seek appropriate advice from scientists about good monitoring programs as it is on scientists to deliver these programs.

An excellent example of close co-operation between DEC Science, Fire Management Services and district operational staff is currently in progress in the Walpole Wilderness Area in the lower south-west forest. The Fire Mosaic Project aims to monitor the impacts of a frequent burning regime, designed to create a small grain mosaic, on a wide range of fire regime sensitive species and habitat values (Wardell-Johnson *et al*., 2006). This research and monitoring project has also attracted several university researchers and students who will add further value by asking more detailed evaluation questions and ensuring the new fire management techniques being trialled are thoroughly tested before being implemented.

Many of the suggested monitoring activities in Table 10.1 could be conducted by University undergraduate students or as Honours, Masters or PhD research projects.

This linkage with the Universities would be invaluable training for students and would have positive spin-offs for local district staff.

10.5 Priorities for monitoring fire management objectives

All prescribed burns should be evaluated to determine if the **prescribed burn objectives** were achieved and to meet DEC auditing requirements. This is fairly straight forward and can usually be assessed during or within a few weeks of the burn. Resources must be made available to ensure this level of monitoring is achieved and results are collated and reviewed on an annual basis. It is important to monitor patchiness and intensity of burns in as many different vegetation associations and fire ages as possible under different weather conditions to build up our knowledge of fire behaviour. Remote sensing techniques to measure patchiness and fire intensity are still being developed and will greatly assist monitoring of prescribed burn objectives.

However, monitoring of **ecological burn objectives** often involves repeated measures over several years and requires a long term commitment to resources. A process for prioritising monitoring effort is obviously required as available resources and skills are limited. Burns in high conservation value areas may have several significant ecological objectives, while other degraded areas may have no special conservation values and have no requirement for ecological monitoring. In addition, it is likely that several burn areas will have the same objectives, such as measuring the impact of fire on salmon gum regeneration or malleefowl nesting. So decisions need to be made at the strategic level to rationalise monitoring resources across the region, and to determine which burns and objectives have the highest priority for monitoring.

The main areas for monitoring of ecological fire management objectives are outlined in Table 10.1 below. The highest priority for monitoring at the district, community and species level will be in areas with known threatened, declining and fire regime sensitive species to ensure their survival. High priority should also be given to monitoring long term changes in floristic composition and cover in representative burnt and unburnt areas, and monitoring changes in fauna habitat suitability. Accumulation of life history attribute data and responses to fire should be an ongoing high priority to underpin the strategic fire planning processes. Indicator species should be monitored both before and after burns in permanent quadrats. Opportunistic monitoring and casual observations of fire responses for different species should be encouraged and a process to capture this information should be developed.

Monitoring of **ecosystem processes** (Table 10.1) is important but often overlooked. In many cases, these processes can be evaluated using quadrats already established for monitoring flora by taking a few additional samples or measurements.

There are many other aspects of fire management that can be monitored and endless species and communities covering vast areas. There are also many different ways to monitor the same objective, including casual observation, photopoint quadrats, flora quadrats and transects using qualitative or detailed quantitative measures. So the decisions of what to measure, where, at what scale and what method should all be related to answering the key evaluation question – why. There will inevitably be some overlap in monitoring species and communities at the different scales and these should be managed co-operatively between research, district and regional staff.

Details of monitoring methods have not been covered in these Guidelines as they are covered in other literature and guidelines (Keighery, 1994; Barker, 2001; SEQ Fire and Biodiversity Consortium, 2002; Hussey and Wallace, 2003; Coote, undated). The Fire and Biodiversity Monitoring Manual developed for local government officers and land managers in Queensland (SEQ Fire and Biodiversity Consortium, 2002) explains how to establish monitoring plots, what to measure and includes data sheets, and methods of data analysis and presentation. This manual could readily be adapted to vegetation types in the Avon Basin. It is also anticipated that a comprehensive DEC biodiversity monitoring manual with standardised methods and field sheets will be developed in future, along with associated databases and training for staff.

Table 10.1 Suggested monitoring activities to determine possible impacts of prescribed burning or absence of fire on ecological values. It is recommended that permanent quadrats and transects be established in representative areas in each vegetation association to derive long term trends. Short term (3-5 years) quadrats and transects are used to test key evaluation questions and should be established before burning and in equivalent burnt and unburnt areas.

Scale of monitoring	What to monitor?	Key evaluation question	Methods	Frequency
Avon Basin or Regional	Local flora extinctions	Are any flora taxa declining to <50% of existing population due to current fire regimes or lack of fire?	Establish permanent sample quadrats in each vegetation association and threatened community. Use existing quadrats where available. Record floristic composition of quadrats	Every 3 years when flowering
	Local fauna extinctions	Are any fauna species declining to <50% of existing population due to current fire regimes?	Establish permanent monitoring transects to record fauna species present and determine fauna population trends of indicator species. Review other survey records eg Birds Australia, WA Museum, Faunafile	Three times per year every 3 years to allow for annual migrations and breeding cycles
	Fauna habitat	Is the fauna habitat suitable to maintain the fauna in the local area with current fire regimes?	Assess habitat suitability score across local areas occupied by each indicator species. Use area and scoring appropriate to species	Every 3 years
	Juvenile stage	Which vegetation associations have less than 25% of total area in the juvenile seral stage?	Query fire history and vegetation association spatial databases and construct frequency distribution curves. Assess area of each vegetation association in juvenile stage	When fire history data is available and every 3 years
	Senescent stage	Which vegetation associations have greater than 40% of total area in senescent seral stage?	Query fire history and vegetation association spatial databases and construct frequency distribution curves. Assess area of each vegetation association in senescent stage	When fire history data is available and every 3 years
	Landscape patchiness	Are areas proposed for burning spatially distributed over the geographic range of each vegetation association?	Examine fire history maps of each vegetation association and ensure adequate spatial variation of proposed burns.	During strategic planning stage each year

	Unburnt reference areas	Are there adequate areas of long unburnt vegetation remaining in each vegetation association as reference areas?	Examine fire history maps of each vegetation association and ensure adequate areas are left as long unburnt	During strategic planning stage each year
	Resource allocation	What percentage of total area prescribed for burning is actually burnt each year?	Review fire management records and determine why any burns did not proceed or meet objectives	Annual review
Landscape or District	Floristic changes	Are current or new fire regimes causing changes in species composition?	Establish quadrats in representative areas with known fire history and record floristic composition. Record life history attributes of fire sensitive species.	Before burn when flowering, then every year for 3 years or until seed set, then every 3 years
	Vegetation structure	What are the changes in vegetation structure and cover after the burn?	Establish quadrats before the burn and measure Muir's vegetation structure and estimate cover for all plant species. Use photopoints	Before burn, 3 months and one year after burn, then every 3 years
	Senescent vegetation	What are the changes in species composition and structure in long unburnt vegetation?	Establish permanent quadrats in long unburnt vegetation in different vegetation associations and record floristic composition and structure	At time of plot establishment and every 3 years
	Natural regeneration	What species are able to regenerate naturally in the absence of fire?	Establish quadrats in unburnt vegetation and record seedling growth and resprouting. Measure diameter of larger shrubs and trees and graph diameter class distributions	At time of plot establishment and every 3 years
	Fauna habitat	How has the burn affected the fauna present?	Survey fauna in areas to be burnt and assess abundance of each species. Monitor key indicator species using trapping and sighting transects, tracks, scats and breeding records	Before burn, 3 months and one year after burn, then every year for 5 years
	Mosaic burning	Has the burn achieved the scale of patchiness and variation in intensity required to meet ecological objective?	Interpret aerial photography or remotely sensed imagery and compare with objectives for area. Conduct GPS survey of area on ground in sample areas.	Within 6 months of burn
	Unburnt patches	Are there adequate areas of long unburnt vegetation remaining in each burn area for fauna habitat?	Assess requirement for long unburnt vegetation in burn area for fauna refuge and habitat. GPS areas left unburnt after burn or	Before burn during planning stage and within 6 months of burn.

			use remote sensing techniques for larger areas	
Vegetation communities	Rock outcrops	Has sensitive fringing vegetation been protected from damaging wildfires?	Map areas of rock outcrops and sensitive vegetation before burn and survey area after burn with GPS. Establish quadrats and assess regeneration of sensitive vegetation in burnt and unburnt areas	GPS areas before burn during planning stage and within 6 months of burn. Assess regeneration in quadrats every 3 years
	Wetlands	Has wetland vegetation been protected from fire?	Map areas of wetland vegetation before burn and survey area after burn with GPS. Establish quadrats and assess regeneration of vegetation and habitat values in burnt and unburnt areas	GPS areas before burn during planning stage and within 6 months of burn. Assess regeneration and habitat values in quadrats every 3 years
	Threatened Ecological Communities	Have TECs been identified and managed according to requirements?	Identify TECs and set prescribed burn objectives for each TEC before burn and assess outcomes after burn	Review before and after burn
	Other fire regime sensitive communities	Have other fire regime sensitive communities been identified and managed according to requirements?	Identify other fire regime sensitive communities and set prescribed burn objectives before burn and assess outcomes after burn	Review before and after burn
	Isolated fragments	Have isolated fragments been managed to allow adequate recolonisation after fire?	Identify isolated fragments and prescribe mosaic burn and leave 30% unburnt. Control and monitor predator population and grazing pressure. Monitor key indicator species.	Before burn during planning stage and assess objectives within 6 months of burn.
Populations and species	Declared rare flora	How do DRF species respond to proposed fire regime or lack of fire?	Establish quadrats in burnt and unburnt areas of the population and observe type of response, measure germination and resprouting success, growth, time to first flowering and seed set and time to maximum seed set and senescence. Ensure approval to take DRF is granted.	Three years before burn is planned, 3 months after burn, then every 6 months until flowering and seed set, then every 3 years
	Priority flora	How do priority species respond to proposed fire regime or lack of fire?	Establish quadrats in burnt and unburnt areas of the population and observe type of response, measure germination and resprouting success, growth, time to first	Three years before burn is planned, 3 months after burn, then every 6 months until flowering and seed set, then every 3 years

			flowering and seed set and time to maximum seed set and senescence.	
	Threatened fauna	Have threatened fauna populations been affected by fire regime or lack of fire?	Establish fauna monitoring transects or quadrats and measure changes in abundance and age structure	Three years before burn is planned, then every 6 months for 3 years, then annually
	Declining fauna	Has the rate of decline of these fauna populations been affected by fire regime or lack of fire?	Establish fauna monitoring transects or quadrats and measure changes in abundance and age structure	Three years before burn is planned, then every 6 months for 3 years, then annually
	Fire regime sensitive flora	How do fire regime sensitive species respond to proposed fire regime or lack of fire?	Establish quadrats in burnt and unburnt areas of the population and observe type of response, measure germination and resprouting success, growth, time to first flowering and seed set and time to maximum seed set and senescence.	Three years before burn is planned, 3 months after burn, then every 6 months until flowering and seed set, then every 3 years
	Weeds	Has the area of invasive weeds reduced after the burn?	GPS and map extent and species of invasive weeds in area. Use photopoints to assess changes in cover and abundance of weeds	Year before burn is planned then every year for 3 years, then every 3 years
	Feral predators	Have feral predators reduced fauna populations after burn?	Establish fauna monitoring transects or quadrats and measure changes in abundance and age structure after burn in baited and unbaited areas. Monitor fox and cat populations using sand pads	Two years before burn is planned then every 6 months after burn for 3 years, the every 3 years
	Grazing animals	Has grazing pressure reduced regeneration success after burn?	Establish fenced and unfenced quadrats in representative areas and measure floristics, structure and cover. Estimate population of grazing animals using scats and diggings or sightings. Use photopoints to assess changes	Two years before burn is planned then every 6 months after burn for 3 years, the every 3 years
Ecosystem processes	Legume component	Has the legume composition in vegetation changed after burn?	Establish quadrats and identify leguminous species and estimate their cover	Year before burn then every year for 3 years then every 3 years
	Pollination	Has burn affected level of pollination of species?	Select indicator plant species and measure pollination success in different size fragments in burnt and unburnt areas	Three years before burn is planned then every year after flowering starts for 3 years
	Soil nutrient status	How has soil nutrient status changed over time and with	Establish quadrats in burnt and unburnt areas and sample representative areas for	Two years before burning in late summer then every year for 3 years

		burning?	total carbon and nitrogen in soil and C/N ratio	then every 3 years
	Soil fungal composition	How has soil fungal composition changed over time and with burning?	Establish quadrats in representative areas in burnt and unburnt areas and sample mycelium and fruiting bodies to identify soil fungi	Three years before burning in late autumn then every year for 3 years then every 3 years
	Soil invertebrate composition	How has soil invertebrate composition changed over time and with burning?	Establish quadrats in representative areas in burnt and unburnt areas and survey for soil invertebrates. Identify and measure abundance	Three years before burning in spring then every year for 3 years then every 3 years
	Litter depth	How has litter depth changed over time and with burning?	Establish quadrats in burnt and unburnt areas and measure litter cover and depth of intact litter and humus	Two years before burning in late summer then every year for 3 years then every 3 years
	Surface soil condition	How has surface soil condition changed over time and with burning?	Establish quadrats in burnt and unburnt areas and estimate cover of bare areas and assess surface sealing or erosion	Year before burning then every year for 3 years
	Water courses	Have any water courses been affected by the burn?	Monitor stream flows and water quality in burnt and unburnt areas for salinity and turbidity	Two years before burning then every year in winter for 3 years
	Soil erosion	Has the burn and pre-burn operations increased soil erosion?	Survey firebreaks and tracks for erosion gullies and assess level of surface soil build up in wind-prone areas	Year before burn then 6 months and one year after burn

PART TWO

FIRE MANAGEMENT GUIDELINES FOR VEGETATION CLASSES IN THE AVON BASIN

1. BROAD VEGETATION STRUCTURAL CLASSES

The 9 broad vegetation structural classes referred to in Section 7.2 of Part One are listed in Table 1.1 and described in the following sections. These classes have been developed for the fire management Guidelines and combine various vegetation associations with similar structural characteristics and dominant species. They are based on the Beard-Hopkins pre-European vegetation association mapping for the area (Beard, 1979 – 1981; Shepherd *et al.,* 2002). The areas refer to that contained within the Avon Catchment Council NRM boundary and used in the ArcGIS project developed for this project. Intersections were made with the Beard-Hopkins pre-European vegetation association mapping and the DEC corporate remnant vegetation GIS layer. The vegetation class descriptions in Table 1.1 refer to the dominant stratum in terms of height and foliage cover but may contain a wide variety of other species in understorey strata. A full list of vegetation association names and areas within these vegetation classes is provided in Appendix II. Wetlands other than salt lakes have been discussed here separately but there is no broad scale mapping of this wetland vegetation and the total area is unknown.

Table 1.1 Broad vegetation structural classes recognised in the Avon Basin for use in fire management, based on life form, structure, height and crown cover of dominant species.

Vegetation Class	**Area remaining (ha)**	**Height (m)**	**Foliage Cover %**	**Description**
Woodland	847,387	10-30	<30	Single stems tall *Eucalyptus* trees with crowns slightly separated
Low Woodland	3,209	<10	10-30	Single stems medium *Eucalyptus*, *Banksia, Acacia* or *Allocasuarina* trees with crowns slightly separated
Mallee Shrubland	697,527	2-10	30-70	Multi-stemmed *Eucalyptus* with tall shrubs and crowns touching
Shrubland	799,279	1-2	30-100	Dense thickets or shrubland of *Acacia, Melaleuca* and *Allocasuarina,* separately or together
Shrub Heath	290,908	<2	30-70	Tall and low shrubs of mixed species
Succulent Steppe	57,570	<2	10-30	Saltbush and samphire saline flats
Mosaic	104,391	various	2-70	Heterogenous mixtures of various structural classes
Rock Outcrops	42,924			Mostly bare areas with granite or greenstone outcrops and sparse vegetation
Salt Lakes and Wetlands	60,258			Mostly bare areas covered by salt lakes and clay pans. Wetland area unknown
Total	2,903,453			

2. WOODLAND

2.1 Distribution of major woodlands

The term 'woodland' describes ecosystems with widely spaced trees with their crowns not touching (canopy cover <30%). The woodland vegetation class used in these Guidelines is generally 10-30m tall and includes open and sparse woodland types. Low woodlands with trees <10m tall are discussed in the following section. Floristic diversity in eucalypt woodlands can be very high in the central wheatbelt, with annuals often making up the majority of species present. Total number of species in this area was similar to the species-rich kwongan heath communities and greater than for mallee, shrubland and rock communities (Yates and Hobbs, 1997).

The major woodland types in the Avon Basin are salmon gum (*Eucalyptus salmonophloia*), York gum (*E. loxophleba*) and wandoo (*E. wandoo*), often in association with red morrel (*E. longicornis*) and gimlet (*E. salubris*) (Fig. 2.1). Other woodland types include redwood (*E. transcontinentalis*), Kondinin blackbutt (*E. kondininensis*), yate (*E. occidentalis*), brown mallet (*E. astringens*) and powderbark wandoo (*E. accedens*). Eucalypt species may occur in pure stands or as mixtures or patchy mosaics with varying understorey components. Succulent steppe and some shrublands also contain isolated eucalypt woodland species, but they are treated as a separate vegetation class. Woodlands generally occupy valley landforms with higher inherent fertility and more favourable water absorption and holding capacity than upland soils (Fig. 2.2). Vegetation typically changes in relation to soil type and underlying geology. Clearing of eucalypt woodlands for agriculture in the south-west Botanical Province of WA has resulted in significant losses (Table 2.1). Beard and Sprenger (1984) estimated that 72% of mallet-powderbark wandoo woodlands, 97% of York gum-salmon gum-wandoo woodlands and 78% of salmon gum-gimlet woodlands have been cleared. Some eucalypt woodlands remain that have an abundance of *Gastrolobium* species which are toxic to livestock.

Major woodland vegetation associations in the Avon Basin are shown in Table 2.1. Minor woodland vegetation associations with less than 1,000ha remaining in the Avon Basin have been amalgamated into the appropriate major woodland type, based primarily on dominant and co-dominant species. Some minor mosaic woodland vegetation associations have also been included in this woodland class. These vegetation associations may well have additional areas outside the Avon Basin boundary but these have not been determined for this project. Five of the minor woodland vegetation associations (145, 948, 1025, 1059 and 1065) have greater than 70% of the remaining total area in Western Australia within the Avon Basin boundary and may need to be considered separately in the future for fire management purposes (see Part One, Table 7.1).


(a)


(b)


(c)


(d)

Figure 2.1 Examples of woodland vegetation (a) salmon gum/York gum woodland with sparse understorey; (b) York gum woodland with shrub understorey; (c) open salmon gum woodland with saltbush understorey; (d) wandoo woodland with low heath understorey.

Salmon gum woodland was originally the most widespread of the woodland species, with optimum development in the wheatbelt and extending into the eastern goldfields areas of Western Australia (Boland *et al*., 1992). However, salmon gum woodlands were preferentially cleared as they grew on heavier, more fertile soils considered indicators of good farmland (Yates and Hobbs, 1997). Following clearing for agriculture, only about 20% of the original extent of salmon gum woodlands remains in the wheatbelt (Beard and Sprenger, 1984), often near rocky areas, in isolated remnants and in roadside vegetation. Fortunately areas of uncleared salmon gum woodlands still exist to the east of the wheatbelt. Salmon gum now occurs in about 36% of all remaining vegetation in the Avon Basin as a dominant or co-dominant in woodlands with York gum and gimlet, and as a component in other vegetation associations. Salmon gum grows to 30m tall and often grows with wandoo and gimlet and sparse shrubland of *Acacia* and *Melaleuca* to 1.5m on upper slopes, or with gimlet and red morrel on lower slopes with heavier clays and a dense layer of halophytic shrubs of saltbush, samphire and herbs to 0.5m where soils are highly calcareous (McArthur, 1992).

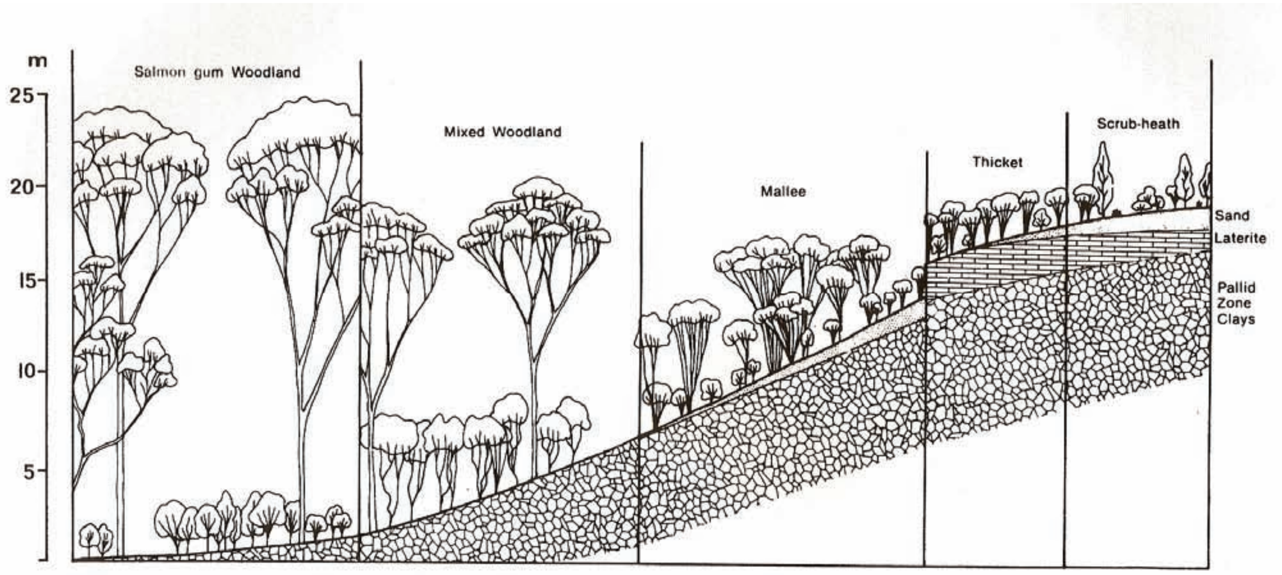


Fig 2.2 A typical catenary sequence of vegetation in the eastern Boorabin system in the eastern Coolgardie 2 IBRA sub-region showing arrangement of woodland species in valleys to scrub heath on the residual lateritic plateau (from Beard, 1981b).

York gum woodland grows on red brown loamy soils over granite and dolerite on upper slopes in the western portion of the Avon Basin. It often grows in association with jam (*Acacia acuminata*), especially where the soil is shallower and more rocky. It also grows widely on flats, near salt lakes and along drainage lines throughout the area, and is generally associated with sandier soils than salmon gum and wandoo. York gum is very widely distributed and is composed of four sub-species, the two main ones being *E. loxophleba* subsp. *loxophleba,* which dominates the western part of the wheatbelt running in a south-east to north-west direction, and *E. loxophleba* subsp. *lissophloia* which dominates the eastern part (Florabase, 2006). *E. l. loxophleba* grows as a small to medium tree to 15m tall with rough fibrous and persistent bark over most of the trunk, with smooth coppery brown bark on upper branches. *E. l. lissophloia* is generally a mallee to 12m high with smooth pink brown bark. A third subspecies grows near Lake Grace and is a mallee to 8m with rough bark on the lower trunk. York gum may also be a component in succulent steppe, and shrublands with *Melaleuca* and *Allocasuarina*.

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8m with rough bark on the lower trunk. York gum may also be a component in succulent steppe, and shrublands with *Melaleuca* and *Allocasuarina*.

Table 2.1 Areas of Beard-Hopkins pre-European vegetation and remnant vegetation for 16 major and 17 minor vegetation associations (VAs) in the woodland vegetation class in the Avon Basin. (Highlighted VAs have no area remaining but are included for completeness.)

Dominant species	Major VA	Minor VAs	Total pre-European area ha	Total Remnant Vegetation ha	Remnant vegetation %
Wandoo/powderbark	5	13	9,963	5,894	59.2
Salmon gum/gimlet	8		450,013	69,527	15.4
York gum/salmon gum	141	357	996,570	127,968	12.8
York gum/salmon gum	142		205,646	35,488	17.3
York gum	352	145, 948	342,884	19,049	5.6
Salmon gum/morrel	511	501, 929	689,254	303,161	44.0
Redwood/merrit	522	1078	124,314	12,143	9.8
Rough fruited mallee	535		24,348	20,780	85.3
Morrel/rough fruited mallee	536	491, 537	13,944	6,318	45.3
Salmon gum	936	468, 931	47,737	21,418	44.9
Wandoo	946	4	47,932	7,019	14.6
Powderbark/mallet	947	962	12,779	2,820	22.1
York gum/wandoo	1023	7	846,928	38,662	4.6
Wandoo/York gum	1049	1025, 1065	836,186	30,559	3.7
Salmon gum/morrel	1067	144, 148	19,582	12,491	63.8
Salmon gum/morrel	1068	1059	271,250	134,091	49.4
	Total		**4,939,330**	**847,388**	**17.2**

Red morrel woodland has a similar distribution to salmon gum, but grows near salt lakes on brown clay loam flats over limestone with a powdery surface soil with high salinity. It grows to 20-25m tall and may be confused with York gum having a similar rough bark on the trunk but with reddish grey smooth bark on upper branches. Gimlet is more an eastern wheatbelt species, usually growing with salmon gum on good red loamy soils or separately on slightly heavier soils. Gimlet grows to 20-25m tall and has thin reddish brown to reddish green bark with strongly fluted trunk. Both red morrel (rarely) and gimlet can occur as a mallee (Florabase, 2006).

Wandoo, **powderbark wandoo, wheatbelt wandoo** (*E. capillosa*) and **mallet woodlands** all grow to 20-25m tall with smooth barked trunks and occur in more restricted areas than salmon gum and York gum. Powderbark wandoo is the most limited in extent, growing in the western part of the Avon Basin associated with gravely soils over gritty clay on lateritic breakaways and stony ridges. Brown mallet is more widespread in the east and south of the Avon Basin growing on and below lateritic breakaways in gravely soils often in association with *Gastrolobium, Dryandra* and *Allocasuarina* thickets. In more eastern areas it may grow as a mallee. Wheatbelt wandoo grows in the north and east of the Avon Basin on red or grey sandy loam and clay on granitic breakaways, and can occur as a mallee (*E. capillosa* subsp. *polyclada*) in some regions (Florabase, 2006). Wandoo is common in the western part of the Avon Basin growing on a range of soil types and landscape positions, but is mostly associated with gravelly clay loams on laterite or granite. It occurs as pure stands or in mixtures with powderbark wandoo or mallet, and further west, grows with marri and jarrah in woodlands and forests. In these Guidelines, the term 'wandoo' includes *E. capillosa* as this was not recognised as a separate species until after the vegetation mapping had been completed (Brooker and Hopper, 1991).

Other minor woodland species include yorrel (*E. gracilis*), merrit (*E. flocktoniaei)* and ribbon barked mallee (*E. sheathiana*).

Many areas of eucalypt woodland have been cut for timber, fence posts, fuel and railway sleepers. Gimlet was heavily logged in the 1890s and 1900s and has regenerated as thick stands after harvesting and fires. Wandoo was heavily cut for railway sleepers and brown mallet harvested for tannin, tool handles and fence posts (Department of CALM, 1995). Large areas of salmon gum woodlands were cleared to supply mining timber in the goldfields over 100 years ago and dense regrowth woodlands have emerged following this massive disturbance (Daniel, 2003). In the wheatbelt, reserves with woodland vegetation were originally reserved for timber and other forest products and so there are few areas of undisturbed York gum, salmon gum and jam in existence. Salmon gum and morrel were considered to have low value for construction timber compared with that for gimlet and mallet (Beard, 1972b).

2.2 Fire management in woodlands

Fire management in woodlands is a low priority and in many cases, fire will not be required. Depending on eucalypt species and type of understorey, fire may be prescribed in eucalypt woodlands on a 20-200 year fire interval range to ensure adequate regeneration. It is likely that regeneration of understorey species and fauna habitat will determine fire regimes in the short term. Fire ecology research is a high priority and trials should be conducted across a range of woodland types.

Eucalypt woodlands cover a large proportion of remaining vegetation in the Avon Basin and have significant structural and functional importance for flora and fauna assemblages and ecosystem processes, so understanding their requirements for fire management and regeneration is essential. However, there is little information available on appropriate fire regimes for Western Australian eucalypt woodlands, other than some for salmon gum and wandoo (Burrows *et al.,* 1990; Yates *et al*., 1994; Hobbs, 2002). Much more research is needed in this area and it is incumbent on all fire managers to monitor woodland fires and prescribed burns and gather fire response data for each species to ensure their long term health and survival.

Fire management of woodlands must consider the appropriate fire regime intervals of the understorey, regeneration of the eucalypt species themselves and maintenance of the native fauna habitat. The eucalypt species present and the type of understorey, whether tall or short shrubs, succulent steppe, sedges, grasses or herbs, will vary primarily with landscape position, soil type and rainfall and to some extent by previous fire regimes for the area. The mapped vegetation associations will provide a guide as to what may be present, but on-ground assessment must be undertaken before a burn prescription is planned. Auslig (1990) has further divided woodlands into those with:

a) low trees and tall shrubs
b) low shrubs
c) hummock grasses
d) tussock grasses
e) herbaceous plants

This sub-grouping would be very useful for fire management and could be determined from Beard's descriptions of woodland vegetation associations. Future development of these Guidelines should assess the usefulness of these woodland sub-groups, particularly in determining fuel loads and fire behaviour to develop separate fire management recommendations.

Dry eucalypt woodlands typically have sparse understorey which does not usually carry hot fire. This is particularly so for woodlands with saltbush and samphire understorey as these shrubs develop foliage with low flammability due to high salt content (Hodgkinson and Griffin, 1982). Woodlands in the eastern part of the Avon Basin (ie Coolgardie IBRA sub-region) are even less fire-prone and generally only small areas will burn before going out. Litter accumulation is significantly slower in these woodland communities due to lower rainfall and slower growth rates and leaf turnover, and to the greater litter harvest by termites (Hobbs, 2002). In addition, sand often covers much of the litter and reduces continuity of litter fuels. Only in rare seasons following a period of high rainfall and lush growth of ephemerals will fire be sustained.

In Victorian grassy woodlands, low intensity patchy fire is required to maintain the diversity of ground stratum species. Fire removes taller grasses (eg *Themeda* spp.) and litter and creates bare spaces in which the smaller herbaceous species can germinate and grow (Tremont and McIntyre, 1994). Fire intervals of 1-3 years have been recommended in those grassy woodlands to create the gaps needed for recruitment (Morgan, 1998), but this frequency would eliminate most shrub and eucalypt species. On less fertile soils in Western Australian woodlands, tall clumping native grasses (eg *Themeda triandra*) are uncommon and the herbaceous stratum is generally low and open with many bare spaces, so creation of gaps for germination may not be an issue. Most ephemerals, annuals and grasses respond readily to high rainfall events, and only a few actually require fire to stimulate germination (Bell *et al*., 1982). Most native grasses and sedges are perennials and regenerate by rhizomes rather than seed, so do not need fire. In any case, grass seeds are mobile and can take advantage of other disturbance events to regenerate and are not threatened by lack of fire (T. Mcfarlane, personal communication). The openness of this low understorey of woodlands can lead to weed invasion of introduced annuals, especially after fire, and this can be detrimental to the regeneration of woody shrubs and trees. For example, strong negative correlation has been found between the survival of York gum seedlings and cover of non-native annuals in the year following a fire in York gum/jam woodland (Hobbs and Atkins, 1991). It is recommended that fire not be prescribed for these herbaceous/grassy woodlands other than for research purposes.

In the medium rainfall western part of the Avon Basin, woodlands often coexist with low to medium density shrubs from 0.5-2m tall, including *Melaleuca, Dryandra, Hakea, Gastrolobium, Acacia* and *Allocasuarina*, which greatly increase the fuel load and continuity. In lower valley positions, ground cover of short grasses and herbs adds to the fine fuel load and allows fires to carry, especially under warm dry and windy conditions (Beard 1980a, b and c; Burrows *et al*., 1990). Rates of litter accumulation are still slow compared with jarrah forests, depending on canopy density and age. At Dryandra, for wandoo and powderbark wandoo woodland, litter fuel accumulation was about 0.5tn/year over the first 10 years after fire and declined even further with canopy age (Burrows *et al*., 1987). In this environment, a high proportion of obligate seeder species are likely to be sensitive to fires less than 15 years apart. This combined with the lack of epicormic crown recovery and slow response from rootstock species, lead to the conclusion that fire frequency in the Dryandra woodlands should be in the order of 20-60 years (Department of CALM, 1995).

Smooth barked eucalypts such as salmon gum, wandoo, gimlet and mallet are particularly sensitive to fire and are easily killed by low scorch (Fig. 2.3). Salmon gum and gimlet are among the few eucalypt species in the south-west that do not form a mallee or produce coppice growth in response to fire (Gardner, 1979). Moderate intensity fire (500 – 2,000kW/m) caused significant damage to wandoo trees, killing 89% of trees less than 10cm diameter and 43% of trees in the 10-20cm diameter class, completely scorching all crowns, burning down large trees with hollow butts, and causing dry sides on most trees greater than 10cm diameter due to cambial death (Burrows *et al.,* 1990). Even low intensity fire (<500 kW/m) killed many small wandoo trees (63%) less than 10cm diameter and caused significant bole and crown damage. Clumps of saplings were particularly susceptible to fire.


(a)


(b)

Figure 2.3. Fire damage to smooth barked single stemmed eucalypt woodland trees (a) wandoo showing split bark and (b) salmon gum showing fire scar. Fire may kill the cambium layer of cells below the bark, and facilitate entry of wood decay fungal pathogens.

Although moderate intensity fire will kill small wandoo (<20cm diameter) back to the ground, they can resprout from lignotubers. Larger trees which suffered complete crown scorch sprouted coppice or epicormic shoots which mostly replaced damaged crowns 14 months after the fire (Burrows *et al.,* 1990). However, wandoo was noticeably slower at producing leaves from epicormic shoots than jarrah or marri on the same site. Coppice growth in salmon gum woodlands is poor or non-existent. Burrows *et al.,* (1990) found that few trees had epicormic shoots nine months after an intense canopy fire, and by 22 months only 50% of adult trees had survived and were resprouting. However, Beard (1979b) states that salmon gums are tender to fire easily killed and badly damaged, and that it does not regenerate from fire. Brown mallet at Dryandra is killed by 100% crown scorch and depends entirely on seedling growth for regeneration. This species takes 12 years to first flowering and is clearly sensitive to frequent fire (Burrows, unpublished data). While most woodland species have some ability as adults to survive crown scorch and resprout from epicormic or basal shoots, many individuals are killed or do not achieve full crown replacement. Research is needed into the age at which bark thickness is sufficient to protect cambial cells from fires of low to moderate intensity and the impact of these fires on adult mortality.

Seedling germination and survival are often rare or non-existent in many unburnt woodland communities (Burrows *et al.,* 1990; Yates *et al*., 1994; Hobbs, 2002) and some fire may be needed to aid regeneration. This is a particular concern in fragmented ecosystems, where the likelihood of wildfire is low (Yates *et al*., 1994). Hot fires will consume large logs and branches, where these are present, to produce an ashbed which greatly increases wandoo and salmon gum seedling growth and reduces seedling mortality (Burrows *et al.,* 1990; Yates *et al*., 1994). Low intensity fires only burn fine litter ground fuels such as leaves and twigs and do not burn at sufficiently high temperatures to produce an ash-bed effect. Fire also stimulates a mass seed release of all canopy stored seed which acts to satiate predators, mainly ants, and produce a mass seedling regeneration which increases the likelihood that some seedlings will survive drought and escape grazing by herbivores. This seed release is best stimulated by hot fire in late summer and autumn when soil and litter fuels are sufficiently dry to carry fire to scorch the crown (Burrows *et al.,* 1990).

For salmon gum woodlands in particular, burning to stimulate seedling recruitment presents a difficult challenge (Yates *et al*., 1994). Adult trees may need to be killed to reduce competition for soil stored water before seedlings will successfully establish (Yates *et al*., 1994). However, if the adult trees are killed by fire and regeneration fails due to inadequate rainfall and high temperatures, the structure of the community could be altered for possibly hundreds of years. In this dry environment where rainfall is low and less predictable, most seedlings do not survive their first summer and successful recruitment may depend entirely on having above annual rainfall years following fire. However, it is a high risk strategy to attempt a prescribed burn in the summer–autumn of one year and hope for above annual average rainfall in the following year. The incidence of above annual rainfall years should be considered when planning prescribed burning for woodland regeneration and long range weather forecasts should be consulted. Other disturbances which remove the adult tree canopy, such as storm damage, drought and flooding can also lead to successful regeneration of salmon gum and may have a significant role.

The impact of low and moderate intensity fires on production of tree and log hollows for fauna is an important issue for many woodland mammals and birds (see Section 3.2 in Part One). Moderate intensity fire during autumn felled large wandoo trees (>60cm diameter) and consumed large logs and branches, whereas low intensity fire felled large trees but did not consume coarse woody debris (Burrows *et al.,* 1990). Severe storms, drought, fire and flood produced many large standing and fallen dead trees in salmon gum woodlands, whereas there were no standing dead trees in the undisturbed sites. Salmon gum and wandoo woodlands produce the majority of habitat hollows in the wheatbelt, particularly for parrots and cockatoos (Saunders, 1979; Long, 1990), and are being destroyed by all causes faster than they are being created. It has been estimated that it may take up to 200 years for suitable hollows to form in wandoo for various parrot species (Stoneman *et al*., 1997), and without successful recruitment of young trees these habitats will be lost as mature trees die.

Other major woodland species, including York gum, Kondinin blackbutt and red morrel have persistent rough and deeply fissured bark up most of their trunks which is likely to confer a greater level of fire resistance (Boland *et al*., 1992; Florabase, 2006). These species often occupy areas around drainage lines and near salt lakes which are less likely to experience fire and have samphire and saltbush understorey which does not carry fire. They also occur as a mallee form in some areas which might suggest they have been exposed to more frequent fire and sprouted from basal shoots. York gum occurs around granite rocks with jam, and occurs as a mallee in mallee shrublands where fire is more frequent than in woodlands.

Where there is a shrubby understorey, then the fire requirements of these species and their associated fauna may determine the appropriate fire interval, season and intensity. Where the shrubs are succulent samphires or saltbush, then generally no fire should be prescribed as it is unlikely to carry. Often these communities also have *Melaleuca* shrubs or thickets and occupy moist drainage lines which will only carry fire if the soil and fine litter are sufficiently dry. On lower to upper slopes, where the shrub layer may be denser and contains more fine fuels, fire can be prescribed to regenerate these shrub species or to manage fauna habitat according to the life attributes of the most fire sensitive species. As many of these species are obligate seeders, then fire intervals in the range 15-60 years are likely to be appropriate. However, the impact of fire on standing woodland trees must be considered, so that overstorey canopy and structure are not permanently altered. The fire regime may include applying moderate intensity autumn burns every 50-100 years to regenerate a proportion of the woodland trees, and milder late autumn burns every 15-30 years to regenerate the obligate seeder shrubs.

3. LOW WOODLAND

3.1 Distribution of low woodlands

Low woodland in these Guidelines includes single stemmed trees less than 10m high with canopies not touching (10-30% canopy cover), and low open woodlands with less than 10% canopy cover. Several low woodland vegetation associations have been combined for these Guidelines where their individual area was less than 1,000ha (Table 3.1). The major low woodland species is rock sheoak (*Allocasuarina huegeliana*) with some jam (*Acacia acuminata*) and York gum. This class includes a low woodland mosaic vegetation association (VA3041) of the same species around granite rocks and a small area of York gum and cypress pine low woodland (VA256). The *Banksia* low woodland (VA949) has been amalgamated with the *Banksia/Xylomelum* shrub heath (see Table 6.1) as there were only 420ha remaining and they contain similar dominant species. Over 80% of the low woodland has been cleared, even though it mainly occurs around granite outcrops.

Table 3.1 Areas of major and minor vegetation associations in the low woodland vegetation class remaining in the Avon Basin.

Dominant species	Major VA	Minor VAs	Total pre-European area ha	Total Remnant Vegetation ha	Remnant vegetation %
Rock sheoak/jam/York gum	25	256, 1005, 1041, 3041	16,260	3,209	19.7

Rock sheoak/jam/York gum low woodlands are usually associated with deeper soil pockets around granite rocks receiving run-off from the outcrop in the Avon Wheatbelt bioregion. They are less than 10m tall with York gum growing only as a small tree or mallee. Rock sheoak usually occurs in coarse sandy soil, and stands vary from open to very dense with *Leptospermum erubescens*, *Santalum spicatum* or *Oxylobium* in the shrub layer and tussocks of *Lepidosperma* and some perennial grasses and ephemerals. Sometimes the ground is bare and covered with sheoak needles and twigs. Jam grows more in loamy soils as open stands usually 5-10m tall with scattered York gum to 15m, but these species all intermingle. This rock sheoak/jam/York gum vegetation normally grades into adjacent York gum woodland (Beard, 1980b).

3.2 Fire management in low woodlands

Low woodlands of rock sheoak and jam are fire prone but regenerate well following fire if grazing pressure is managed. However, the frequency of fire events is likely to be low, in the order of 30-50 years, following above average rainfall years. Due to the proximity to rock outcrops, fire management of low woodlands should be determined more by the fire sensitive refugial species that may be found in this habitat than by the dominant species.

Rock sheoak/jam low woodland is likely to be fire prone although it may have some protection from being around granite outcrops which generally experience a lower fire frequency (Hopper, 2000). Both dominant species are killed by 100% crown scorch and take 5 years to reach first flowering. Rock sheoak is an obligate seeder, while jam stores seed in the soil. Rock sheoak may, however, regenerate well in the absence of fire. Muir (1985) observed that a stand near Quairading that had not been burnt for 63 years had a high proportion of seedlings and young saplings along with some dead and senescing older trees. Mass germination is stimulated by fire in this species and highest seedling density and highest survival rates were recorded following a fire in 1987 at the Chiddarcooping Nature Reserve, north-east of Merridin (Yates *et al*., 2003a). There was a rapid thinning of seedlings in the first 4-5 years following germination. By the time of another fire in 2000, only 13.7% of seedlings from the first fire were alive and all were killed by the second fire.

Germination of jam in the absence of fire is not recorded in the literature but has been observed (personal observation) and should be investigated further. Jam occurs either as single stemmed small trees or as tall shrubs with up to 6-8 stems arising from the ground (Maslin, 2001), which suggests that it also has the ability to coppice after fire. Thickets of rock sheoak and jam often adjoin the low woodlands and it is often not feasible to separate these communities for fire management. The rock sheoak/jam low woodlands may well be the mature seral stage of the thicket formation, having escaped some fire through being close to granite outcrops. Alternatively, jam and rock sheoak may grow to greater heights in the run-off zones around rock outcrops due to greater availability of moisture. Fire management of these woodlands is likely to be determined more by the presence of fire sensitive refugial species around the granite rock habitat than by the fire requirements of the dominant species.

4. MALLEE SHRUBLAND

4.1 Distribution of mallee shrublands

Mallee shrubland is dominated by *Eucalyptus* species with a multi-stemmed habit usually less than 10m tall, which coppice or resprout after fire or drought. While significant areas of mallee shrubland remain across southern Australia more or less intact, only about 30% of the pre-European mallee shrubland remains in the Avon Basin (Table 4.1). Most of the area of remaining mallee shrubland occurs in the Mallee2 IBRA sub-region (MAL2), with about 80,000ha in the Coolgardie2 IBRA sub-region (COO2) and about 22,000ha in the Avon Wheatbelt1 IBRA sub-region (AW1). Most of the mallee species reach 3-5m tall (whipstick mallee) in the eastern areas but can grow to 8-10m tall (bull mallee) in better conditions in the western areas. Height of stand also varies with age and many mallee communities could form low woodlands if left long unburnt (Beard, 1980c). In more optimal conditions many mallee species can form a single stemmed small to medium tree to 12m. Canopy cover is typically 10-30% but this varies and can be sparse (2-10%) or dense (70-100%). Understorey shrubs are usually >1m tall and can range from very open to very dense.

Some species of *Eucalyptus* never grow as mallees (eg *E. salubris* and *E. salmonophloia*) while other species including *E. oleosa* and *E. eremophila* may occur as either trees or mallees. Species that only appear in the mallee form such as *E. pleurocarpa* and *E. cylindriflora* occur in the drier regions or where there is dense scrub or thicket, which may indicate that fire has been a factor in the development of this type of growth (Gardner, 1979). It could also indicate an adaptation to aridity and periodic drought.

In the Avon Basin, mallee is most often found on the edge of the sandplain or just below lateritic breakaways on deeper soils (see Fig. 2.2), transitional between the sandplain heathland on the old plateau and woodlands in the valleys. Mallee communities in the eastern areas occur on red sandy soils whereas the yellow sandy soils tend to carry *Acacia* thickets (Beard, 1972a). York gum may form a mallee community or low woodland around granite rock outcrops in western and northern areas of the Avon Basin. Mallee vegetation is well represented in MAL2 where it occupies the bulk of the Hyden vegetation system. Many of the shrub species are common to the adjoining heathland or woodland, depending on soil type, with more heathland species on the sandier soils and woodland species on clayey soils. In AW1, mallee understorey can have high species richness comparable to that for kwongan heathland (Beard, 1980b). Floristic diversity of mallee *Eucalyptus* species in Western Australia is higher than in the south-east of Australia, possibly due to the mosaic of relictual soils and climatic changes which resulted in genetic isolation and speciation (Hill, 1990 in Bradstock and Cohn, 2002).

Patches of mallee may also occur within the adjoining heathland on sandplains and included in the shrubland class (eg VA1024), or in woodland communities as woodland/mallee shrubland mosaic units (see Section 8 below). Tallerack (*E. pleurocarpa*) mallee heath shrubland (VA47) has been included here as mallee shrubland because of the height and dominance of the eucalypt species. Note: Tallerack was previously the common name for *E. tetragona* (Gardner, 1979) but this is no longer recognised as a species nor is tallerack used as a common name in Florabase, while *E. oleosa*, which has a number of sub-species, is sometimes also known as red mallee (Brooker and Kleinig, 1996)

The main *Eucalyptus* species in mallee shrublands in the MAL2 and COO2 IBRA sub-regions are tall sand mallee (*E. eremophila*), silver mallet (*E. falcata*), blue mallet (*E. gardneri*), tallerack (*E. pleurocarpa*), redheart (*E. decipiens*), white mallee (*E. cylindriflora*), open fruited mallee (*E. annulata*), gooseberry mallee (*E. calycogona*), mirret (*E. celastroides*) and giant mallee (*E. oleosa*). Some of the dominant large shrubs in these vegetation associations include *Banksia media, Hakea multilineata, Melaleuca uncinata, Callitris roei* and *Calothamnus quadrifidus*. Further east and near salt lake systems, chenopods and saltbush may appear in the understorey.

Table 4.1 Areas of Beard-Hopkins pre-European vegetation and remnant vegetation for 7 major and 2 minor vegetation associations (VAs) in the mallee shrubland class for the Avon Basin.

Dominant species	Major VA	Minor VAs	Total pre-European area ha	Total Remnant Vegetation ha	Remnant vegetation %
Tallerack	47		9,364	7,163	76.5
Tall sand mallee	519		1,213,792	466,463	38.4
Redwood/black marlock	960	934	220,703	23,133	10.3
York gum/ribbon gum	1055	1058	136,170	14,037	10.3
York gum/ribbon gum	1063		171,323	154,859	90.4
Tall sand mallee/black marlock	1075		174,470	29,606	17.0
Morrel/ribbon gum	1081		15,148	2,266	15.0
	Total		**1,940,970**	**697,527**	**35.5**

In the AW1 IBRA sub-region, York gum (*E. loxophleba*) and ribbon gum (*E. sheathiana*) dominate the mallee shrublands with a range of other species including narrow-leaved red mallee (*E. foecunda*), stiff-leaved mallee (*E. rigidula*), black marlock (*E. redunca*), redwood (*E. transcontinentalis*) as well as tall sand mallee and white mallee. Some of the dominant large shrubs in these vegetation associations include *Melaleuca uncinata, M. hamulosa, Callitris columellaris, Allocasuarina acutivalvis* and *Acacia acuminata.*

Variation in the floristics and structure of mallee communities, especially in the understorey, has lead to a diverse and distinctive array of vertebrate species. Compared with adjacent woodland and heath communities, mallee has higher reptile species diversity, similar mammal diversity and lower bird and amphibian diversity (Bradstock and Cohn, 2002). However, other than some passerine bird species most are not mallee specialists.

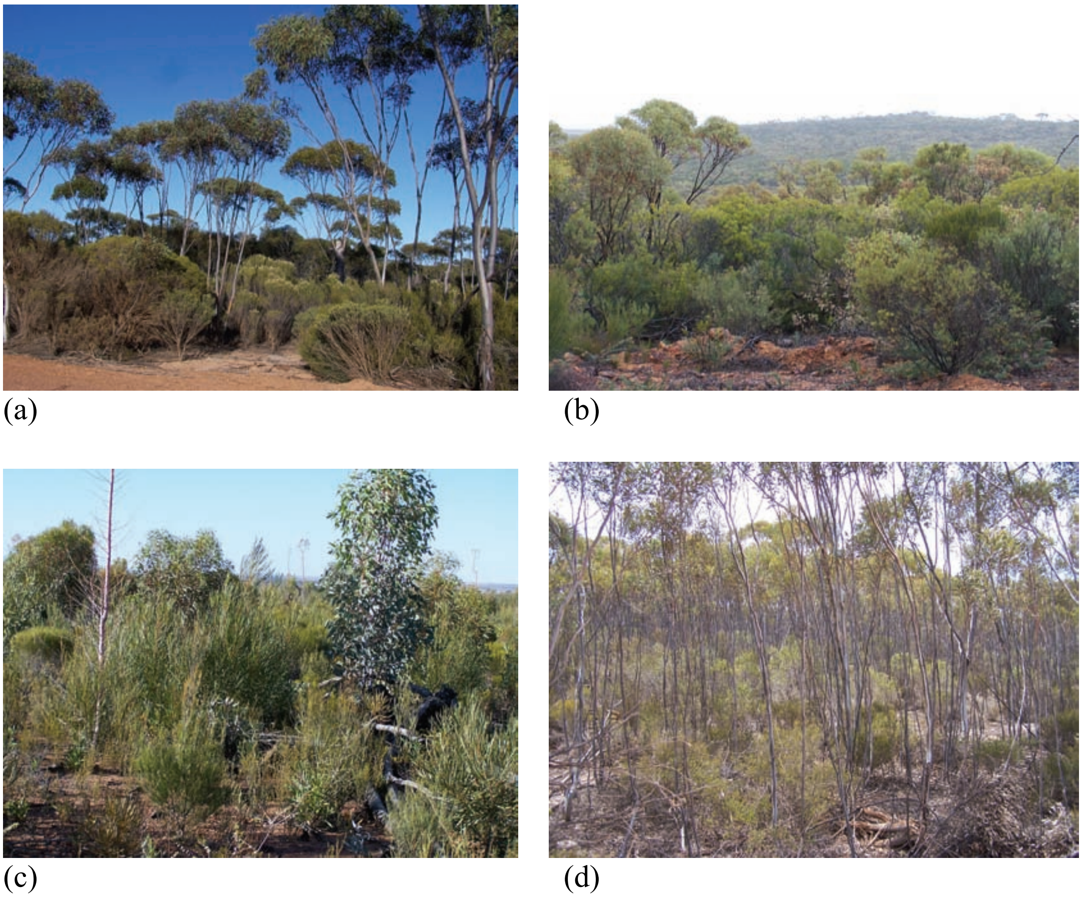
4.2 Fire management in mallee shrublands

Eucalypt mallee shrubland is fire prone and regenerates well after fire. Fire return interval can be less than 5 years after above annual average rainfall and cause local extinction of fire sensitive species. A tolerable fire interval range of 15-50 years is recommended, depending on the key shrub and heath species present. Where mallee species dominate the range could be 20-80 years given the longevity of these species. Fire spread can be rapid when fuels are dry and burn very large areas. Cooler mosaic burning, wind driven buffers, discontinuous fuels and diversity of landform should be used to create finer scale mosaics. Patches of dense canopy cover in long unburnt vegetation are required by malleefowl and hollow nesting birds.

Eucalypt species within mallee communities are generally well adapted to respond to fire by coppicing and it is likely that fire, and to some extent drought, has maintained the mallee habit for many thousands of years (Fig. 4.1). Fuels in mallee shrublands are often discontinuous and may take decades to build up where crown cover is sparse. Suspended materials, including bark strands, dead leaves and twigs accumulate among the stem bases and up to 1m above the ground and are clustered around individual plants. In the south-east of Australia, surface fuel load in mallee shrublands from eucalypt litter can vary from 5-15tn/ha while shrub litter loads are often only 1.5tn/ha and partially covered with sand. In some cases it may take more than 100 years to reach a stage when fire will carry (Bradstock and Cohn, 2002). However, in more favourable areas where the shrub component is more significant, areas of mallee shrubland have re-burnt 5-10 years after the previous fire (Bradstock and Cohn, 2002).

In true mallee shrublands, eucalypt litter is the main contributor to fuel loads, and these build up fairly regularly from year to year. Additional to this is the fuel loading from shrubs which varies from place to place, and from the annuals, grasses and ephemerals which can proliferate following periods of above average rainfall. It is this additional flush of surface fuels which provides not only the increase in fuel loading but the spatial continuity of fuels to enable fire to carry, and enable very large areas to burn out in a day (<100,000ha). In some extreme cases where above average rainfall follows a fire, mallee shrubland can re-burn within three years due to the flush of flammable ephemerals (Bradstock and Cohn, 2002). Prediction of peak flammability in mallee shrublands must therefore consider both the time since last fire for the build up of eucalypt litter and also the stochastic nature of ephemeral fuels.

Ignition source for most wildfires in mallee shrublands is lightning during summer storms. Where these storms are dry, multiple ignitions can occur and on days with extreme or very high fire danger rating, fires can develop rapidly and spread at faster rates than in other eucalypt dominated communities, such as woodlands and forests (Bradstock and Cohn, 2002). Total fuels in tallerack mallee (*Eucalyptus pleurocarpa*) heath shrubland in the Stirling Ranges increased up to 10 years after fire then remained fairly constant at about 13tn/ha, while litter and dead fuels continued to increase up to at least 20 years (McCaw, 1998). Dead fuels in mallee shrublands are also typically drier than in eucalypt forests due to the relative shallowness of the fuel bed and greater exposure to the sun (McCaw, 1998). Although fuel loadings in mallee shrubland are less than in woodland and forests, rates of spread are often 3-6 fold greater due to the more open canopy and greater surface wind speed. Because of the discontinuous nature of the mallee fuels, a higher threshold of wind speed must be exceeded before fires are likely to spread (McCaw, 1998). Fire spread is also strongly affected by litter moisture content, which can be used to predict whether or not fires will spread or go out (McCaw, 1998). Below 8% moisture content of the shallow litter, fire will spread freely and rate of fire spread increases linearly with wind speed. The loose strands of bark and leaves around mallee stems are thought to contribute to prolific spotting and forward spread during wildfires.



(a) (b) (c) (d)

Figure 4.1 Examples of mallee shrubland communities (a) long unburnt mallee with broombush thicket; (b) midslope mallee with mixed shrubland; (c) regrowth mallee shrubland after fire; (d) whipstick mallee regrowth after fire.

In remote areas, access and fire suppression resources are limited and fire size can be very large and continue burning until there is a change in the weather cycle. In fragmented mallee shrublands surrounded by agricultural landscape, ignition may come from various sources (eg lightning, escaped stubble fires, machinery and arson) but the size of fires is smaller. Both lack of fire and too frequent fire in fragmented reserves and remnants may be leading to local extinction of species.

Mallee populations have regenerated with little mortality from one fire every 5-10 years (Bradstock and Cohn, 2002). More frequent fire caused significantly higher mortality after autumn burns than after spring burns, and it has been suggested that autumn burning forces out-of-phase growth and physiological stress (Noble, 1989). Spring burning was also more favourable for seedling growth of mallee eucalypt species, than either autumn or winter burning. Noble (1982) suggested that successful seedling establishment may only need to occur once every few centuries, since eucalypts may live for around 200 years.

Although mallee eucalypts and a range of shrub species can survive fire every 5-10 years by resprouting, other understorey species are killed by fire and require more time between fires to replenish seed stores. Understorey species such as *Callitris roei, Hakea multilineata* and *Banksia media* are obligate seeders and are killed by 100% crown scorch and are totally dependent on successful seedling regeneration to avoid local extinctions. Other species have long juvenile periods, such as *Melaleuca uncinata* (6 years), *H. multilineata* (5 years) and *B. media* (4 years) and will be detrimentally affected by frequent fire, although some also have the capacity to resprout from basal shoots (N. Burrows, unpublished data). Individuals of these species require at least 10-15 years between fires. In tallerack (*Eucalyptus pleurocarpa*) mallee heath shrubland in the Stirling Ranges, about 20% of species are fire sensitive, some of which have primary juvenile periods of 7 years or longer, so that fire intervals of 14 years may be required, yet fire return periods as short as 5-8 years have been recorded (McCaw, 1998).

Callitris species are particularly sensitive to fire and may only occur where the previous fire frequency has been relatively infrequent, such as around rocky outcrops, on breakaways, sand dunes, near salt lakes or drainage lines. These are usually long-lived species and can therefore provide some information about previous fire history of an area. It is not known whether *C. roei* and *C. columellaris* are able to regenerate from seed without fire, or to survive fire after they reach a certain age (as with *C. verrucosa*), but they are likely to be key fire species for which more information is required.

Mallee vegetation provides a range of habitat resources for fauna including food, shelter and nest sites. Effects of fire regimes have been studied for some species, particularly birds. Some opportunist species are common in recently burnt vegetation, while others that become more numerous in the 1-10 years post-fire are those which are generally common and widespread, also using other vegetation types such as heath or woodland. In the intermediate stage after fire (10-30 years), birds endemic to mallee may be favoured as the vegetation becomes taller and thickets become denser with abundant litter, including Gilbert's whistler and southern scrub robin. Those that prefer older vegetation (>30 years) include the endangered malleefowl and hollow nesting birds such as the striated pardalote and the regent parrot (Bradstock and Cohn, 2002).

In one Victorian study (Meredith, 1984), bird abundance peaked at 15 years post-fire and then declined, while species richness continued to increase until 60 years post-fire in mallee shrubland. That study found that 13 out of 16 mallee endemic bird species were only recorded in mallee vegetation that was older than 17 years post-fire. In other words, if that vegetation was burnt completely on a 15 year cycle to maximize productivity, then 40% of the bird species would be lost. Young vegetation was simple and dense with little structural diversity, whereas in older vegetation there were four layers, including trees, shrubs, litter and dead and fallen branches which provided greater diversity of resources and nesting opportunities. Although productivity of the vegetation declined after about 15 years, this was coupled with greater biodiversity and species richness.

Adult malleefowl have a home range of about 4 km^2 and juveniles may disperse and forage across a range of several km (Benshemesh, 1990), but it is not know if this is the optimal burn patch size. Optimal patch size may also vary in different vegetation types due to differences in foraging quality. Highest density of active mounds around Ongerup in the great southern occurred at a 140ha site surrounded by cropland that had been isolated for over 20 years (Harold and Dennings, 1998). Dense canopy cover has been shown to be the most important feature associated with high breeding densities while the abundance of acacias, an important food source, was poorly correlated suggesting that availability of food was not limiting populations (Benshemesh, 1992). Recolonisation of malleefowl into areas burnt by wildfire is very slow and it may take 30 years in some areas before the habitat is suitable. Widescale burning on a 10-25 years cycle will eventually eliminate malleefowl from an area (Benshemesh, 2000). While large scale fires are deleterious to malleefowl populations, the effect of fire can be mitigated if patchy mosaic burning is used (Benshemesh, 2000). Radio tracking studies showed that birds survived in relatively small unburnt patches (about 1/10th the average home range size) by using burnt patches for foraging and unburnt patches for roosting, nesting and daytime shelter. Ten years after the patchy burn breeding success was similar to levels before the burn. Patchy aboriginal burning of spinifex may have also protected the mulga habitat used by malleefowl in central Australia from wildfire. Effective fox, dingo and cat control after burning may improve this rate of recolonisation and requires further research.

5. SHRUBLAND

5.1 Distribution of shrublands

Most of the pre-European shrubland occupied the north-eastern half of the Avon Basin, with very little mapped in the Avon Wheatbelt 2 (AW2) or Mallee 2 (MAL2) IBRA sub-regions. The remaining shrubland still occupies almost half (46%) of the original pre-European extent in the Avon Basin, being a dominant vegetation class in the largely uncleared Coolgardie 2 (COO2) IBRA sub-region. These tall shrublands form either closed shrubland or 'thicket', or open shrubland, or 'scrub' as described by Beard (1972a) (Table 5.1). Broombush thicket is a single layered very dense shrub community with low species diversity (Beard, 1972b). The broombush habit consists of numerous steeply ascending branches from near the base with no main leader, which develop rounded crowns with dense foliage of small needle like leaves.

The characteristic genera in the shrublands are *Acacia, Melaleuca* and *Allocasuarina* either in relatively pure stands or as mixed alliances (Fig. 5.1). The same species may also occur in woodlands, mallee shrublands and shrub heath, particularly in transition zones between these formations. *Allocasuarina* tends to dominate where annual rainfall exceeds 325mm and *Acacia* dominates below that figure with mixed dominance in between (Beard, 1984).

Acacia thickets replace heathland on the central sandplain habitat in the drier semi-arid regions to the east (Beard, 1972a).

Table 5.1 Areas of Beard-Hopkins pre-European vegetation and remnant vegetation for 15 major and 12 minor vegetation associations (VAs) in the shrubland vegetation class in the Avon Basin. (Highlighted VAs have no area remaining but are included for completeness.)

Dominant species	Major VA	Minor VAs	Total pre-European area ha	Total Remnant Vegetation ha	Remnant vegetation %
Acacia/sheoak thicket	36		300,249	65,615	21.9
Melaleuca thicket	37		6,371	1,785	28.0
Melaleuca scrub	41		13,772	4,988	36.2
Acacia thicket	435	413	529,999	75,004	14.2
Mixed acacia thicket	437	18, 19, 39, 59, 420, 436, 483, 555	139,429	32,311	23.2
A. quadrimarginea thicket	520	202	26,881	3,156	11.7
A. brachystachya scrub	538	40	128,660	21,135	16.4
Tammar thicket	551		163,805	44,453	27.1
Sheoak/calothamnus thicket	552	516	13,116	12,444	94.9
Jam/rock sheoak thicket	954		6,502	1,043	16.0
Tammar thicket	956		25,556	2,744	10.7
Mallee/sheoak thicket	1024		593,089	58,312	9.8
Broom bush thicket	1053		12,706	1,722	13.6
Acacia/tammar thicket	1056		21,073	3,098	14.7
Acacia/sheoak/melaleuca thicket	1413		1,219,852	471,468	38.6
	Total		**3,201,060**	**799,278**	**25.0**

The dominant species varies with soil type and landform. *Melaleuca* thickets form on valley bottoms around saline lakes and drainage systems, and in winter wet places with sandy clay soils. *Allocasuarina* and *Acacia* thickets and scrub form on the residual lateritic sandplain plateau on yellow sand with ironstone gravel at varying depths. *Acacia* thickets form on deeper yellow sands on the broad interfluve lateritic sandplains, with *A. resinomarginea* the dominant species on the deepest sands and *A. neurophylla* on the shallow sands near the breakaways, with a mixture of other species of *Acacia, Allocasuarina* and other genera in between. *Allocasuarina* shrublands form on the edge of the sandplain near the breakaway on shallow sandy gravel with a higher proportion of gravel in the upper layers, or on shallow granitic soil around rock outcrops. *Allocasuarina campestris* (tammar sheoak), *A. corniculata* and *A. acutivalvis* dominate this habitat on the sandplains in the western regions of the Avon Basin, but are replaced by *Acacia neurophylla* in the drier eastern regions. On shallower granitic soils on the mid-slope below the breakaways, *Acacia* forms more open shrubland with a diverse understorey (Beard, 1972a). In eastern areas, open *Acacia* scrub may also form around salt lakes or claypans on low red brown sandy ridges (Beard, 1979b). *Acacia quadrimarginea* and *Allocasuarina* thickets also form on ironstone and greenstone (basaltic) ridges in the east of the Avon Basin (Beard, 1972b).

(a) (b)

Figure 5.1 Examples of shrubland (a) *Allocasuarina* and *Acacia* dominated shrubland and (b) *Melaleuca* dominated shrubland on a lower slope adjoining a saline creekline.

The structure of shrubland varies with dominant species and time since fire. Shrubland is generally greater than 1m tall, other than following a fire, and may reach 4-5m tall if long unburnt. Shrublands are often devoid of understorey shrubs. Some shrubland may have scattered emergent trees, eg *Melaleuca uncinata* shrubland with scattered York gum, or *Allocasuarina campestris* shrubland with scattered wandoo. As well, woodlands of salmon gum and gimlet on lower slopes with deeper clay loam soils may have thickets of low *Acacia* shrubland. Thickets of *Acacia, Allocasuarina* and occasionally *Melaleuca* may also occur in heathland as mosaic patches or transitional zones on the sandplain soils.

The main shrubland vegetation association in the Avon Basin is VA1413 (mainly in COO2 and AW1 sub-regions with some in MAL2) which consists of mixed *Acacia, Allocasuarina* and *Melaleuca* alliances. The dominant species in this alliance includes *Allocasuarina campestris, Allocasuarina acutivalvis, Allocasuarina corniculata, Acacia ligustrina, Acacia resinomarginea, Acacia brachyphylla, Melaleuca uncinata* and *Melaleuca cordata* as well as *Callitris canescens, Eucalyptus leptopoda, E. burracoppinensis, E. loxophleba, E. redunca, Calothamnus quadrifidus* and *Grevillea paradoxa*

The next most widespread vegetation association is VA435 which is *Acacia* dominated thicket mostly in the COO2 sub-region. The main species in this shrubland are *Acacia acuminata, Acacia neurophylla, A. beauverdiana, A. resinomarginea, Brachychiton gregorii, Melaleuca uncinata, Allocasuarina corniculata, Calothamnus quadrifidu*s and *E. loxophleba.* The main species in VA36, which is mostly in the AW1 sub-region, are *Acacia stereophylla, A. signata, Allocasuarina acutivalvis, Grevillea paradoxa, Hakea falcata, Melaleuca cordata* and *Leptospermum erubescens.*

Melaleuca dominated shrublands (Boree) are not very extensive and include *M. lateriflora* in the most saline areas, with *M. scabra, M. thyoides, M. uncinata* and *M. cuticularis* in less saline areas. Some *Melaleuca* shrublands may have samphire understorey but these have been included in the succulent steppe vegetation class. Due to excessive land clearing in the wheatbelt region, ground water levels have risen and increased in salinity causing the decimation of many *Melaleuca* shrublands (Beard, 1972b).

5.2 Fire management in shrublands

***Acacia, Melaleuca* and *Allocasuarina* dominated shrublands are well adapted to regenerate after fire by seed, and some can germinate well in the absence of fire. Minimum age to flowering for most species is 4-6 years, and senescence age is possibly around 50-60 years, so tolerable fire interval range may be 15-50 years. Mosaic burning should be used at the landscape and smaller patch scale to avoid widespread wildfires. *Melaleuca* shrublands are likely to require less frequent fire and impacts of increasing salinity need to be considered.**

Shrublands, including thickets and the more open scrub, are very flammable and well adapted to regenerate after fire by seed and coppice growth. Fires ignited by lightning in summer storms are relatively common in wheatbelt shrublands and have probably been a factor in determining the structure of this vegetation for millions of years. In the semi-arid areas, these shrublands may not have been regularly burnt for hunting by aborigines as the impenetrable nature of the thickets probably discouraged aboriginal use (Beard, 1972a). Indeed, Beard (1972a) comments that in the Jackson area where it is sparsely settled, most sandplains show no evidence of having been burnt at all and the thickets are evidently very old with a marked scarcity of burn patterns in the aerial photography. The general lack of understorey shrubs, especially in older shrublands, may limit the spread of fire.

A number of short-lived pioneer species appears after a fire in *Acacia* thicket shrublands which give way to the climax species within five years (Beard, 1972a). In the drier areas where fire appears to be uncommon in dense thickets, the seed of the pioneer species must remain viable in the ground for many years. Pioneers species include the fast growing small tree *Codonocarpus cotinifolius* (native poplar)*, Grevillea excelsior, G. didymobotrya, Cyanostegia aungustifolia, Hemigenia sp., Keraudrenia integrifolia, Lachnostachys eriobotrya* and *Dampiera eriocephala.*

Large areas of shrubland have been burnt by wildfire from 1994 to 2005 in the Avon Basin and it useful to examine some of this data (Table 5.2). Most of these wildfires have been in the COO 2 IBRA sub-region but this data refers to the four IBRA sub-regions contained within the Avon Basin boundary. It can be seen from this table that the fire cycle calculated over the last 11 years for shrubland vegetation associations in the Avon Basin ranges from 5 to 1,887 years. If we estimate the minimum tolerable fire interval is 15 years and the maximum tolerable fire interval is 50 years, then the Fire Cycle [(15+50)/2] is 32.5 years. The largest shrubland association (VA1413) is half this figure (17 years) and this tends to dominate the overall average value. Another large vegetation association (VA435) has had nearly 22% of the total area burnt each year, some areas burnt repeatedly, and the calculated fire cycle is only 5 years, which is cause for concern. The database indicates that 96,745ha were burnt in VA435 in 2001 alone which only has a total area of 74,926ha, so either there were several fires in the same area in the same year or the fire scar database contains some errors. Most of the other shrubland associations have calculated fire cycles mush greater than 32.5 years, which would suggest they are well into senescence, but these are only calculated on the last 11 years data. Accurate fire scar data determined from satellite imagery for at the last 30-40 years is urgently required so that longer term trends and more accurate estimates of actual fire cycles can be determined.

Very little of the *Melaleuca* shrubland (VA37 and VA41) has been burnt in the last 11 years and it is likely that fire is naturally infrequent in this community due to its low position in the landscape and having longer periods of moist soil. Regeneration of these shrublands may be required periodically to prevent senescence, although many species have some capacity to regenerate from seed in the inter-fire period. The ability of seeds to germinate in an increasingly saline soil environment will need to be assessed.

Table 5.2 Total and average areas burnt by wildfire and prescribed burning from 1994-2005 for each major shrubland vegetation association in the Avon Basin boundary, and calculated fire cycles. Data are from the DEC corporate data bases.

Vegetation Association No.	[a] **Total area remaining (ha)**	[b] **Area burnt 1994-2005**	[c] **Ave area burnt/yr (ha)** [b/11]	**Ave area burnt/yr (%)** [c/a*100]	**Fire cycle (yrs) (calculated)** [a/c]
36	65,615	7,083	644	1.0	102
37	1,785	0	0	0	-
41	4,988	0	0	0	-
435	74,926	179,331	16,303	21.8	5
437	32,062	5,566	506	1.6	63
520	3,156	1,721	156	5.0	20
538	21,135	10,740	976	4.6	22
551	44,453	259	24	0.1	1,887
552	12,439	1,134	103	0.8	121
954	1,043	204	19	1.8	56
956	2,744	233	21	0.8	129
1024	58,312	4,840	440	0.8	133
1053	1,722	47	4	0.2	406
1056	3,098	351	32	1.0	97
1413	471,468	307,509	27,955	5.9	17
Total	**798,946**	**519,018**	**47,183**	**5.9**	**17**

All *Acacias* have hard coated seeds that remain viable in the soil for up to 50 years (Auld, 1986) and are mostly available for regeneration after fire or other environmental disturbance. A few *Acacia* species (eg *A. acuminata, A. aneura*) also have the ability to resprout but there is little information available on this subject. Ants probably bury a significant proportion of *Acacia* seeds and other seeds with an attached aril, or lipid-rich elaiosome, and a reasonably hot fire is required to stimulate germination. Wind and water movement of soil also buries a proportion of seed which helps reduce seed predation. The proportion of dormant seeds that are stimulated to germinate, remain dormant or are killed by heat shock depends on the amount of heat transferred through the soil during the fire and the depth of seed burial (Auld, 1987). Successful germination may only occur following an occasional high rainfall season when ground fuels accumulate and are burnt by a hot fire.

Hard coated seeds may therefore persist for many years in the soil and only a proportion of the total seed bank may be stimulated to germinate in any one fire.

Optimal field conditions for stimulating seed germination of shrubland species are largely unknown at this stage. Atkins and Hobbs (1995) examined germination responses of three *Acacia* species from salmon gum woodland to different oven temperatures and found one species (*A. hemiteles*) had a similarly high germination rate with no heat treatment or up to 100°C but was severely inhibited by 150°C. Another species (*A. colletioides*) had low germination rates for 0 to100°C heat treatments with highest germination rates at 150°C. Fire heterogeneity may therefore be an important mechanism promoting the maintenance of species diversity in shrublands (Atkins and Hobbs, 1995). For 35 eastern Australian Fabaceae heathland species, the temperature of the fire required for breaking dormancy and optimal germination of hard coated seed varied with species, and was found to range from 95-100°C (Auld and O'Connell, 1991). Temperatures above 120°C were lethal to seeds of all species examined.

Considerable variation in soil temperatures over short distances was found during management fires at Durokoppin Nature Reserve in the central wheatbelt during autumn, with soil surface temperatures attaining 770°C in some quadrats (Atkins and Hobbs, 1995). Significantly greater temperatures were found at the soil surface where litter was present or where additional litter was added (665°C) than for bare soil (560°C). Some species had higher seedling numbers where litter was present before the fire (eg *Allocasuarina acutivalvis* and *A. campestris*) while other species had higher seedling numbers where there was bare soil before the fire (eg *Leucopogon hamulosa* and *Verticordia chrysantha*). Again these results suggest differential responses of species to high temperatures. These soil surface temperatures are much greater than the 120°C found to be lethal by Auld and O'Connell (1991) and suggest that most surface seeds would have been killed by these autumn fires. However, temperatures at 2cm below the soil surface at Durokoppin were significantly less than either surface or canopy temperatures in most quadrats and this probably allows buried seeds to survive the fire front. Fires that consume 6-20tn/ha of fine ground fuel are required to generate the range of temperatures (95-100°C) needed for optimal germination at 1-3cm soil depth where most seeds are thought to be stored (Bradstock and Auld, 1995).

Optimal temperatures for release and germination of canopy stored seed (eg *Melaleuca, Eucalyptus, Allocasuarina*) are largely unknown and will depend on height of seed capsules above the ground and amount of litter and heat transfer upwards to the canopy. Some capsules and cones of serotinous species may be consumed by hot fires (Lamont and Barker, 1988), but seed release is often greater following a hot fire than a mild fire (Enright and Lamont, 1989).

6. SHRUB HEATH

6.1 Distribution of shrub heath

Most of the original shrub heath in the Coolgardie sub-regions remains intact, being 63.9% of VA1148 (Table 6.1). There are also significant areas remaining of VA380 and VA2048 in MAL2, but most of the original shrub heath in the Avon Wheatbelt IBRA sub-regions has been cleared and only about 4% remains. Shrub heath communities are largely confined to areas with at least 200mm winter rainfall per year, greater than that required for the taller and denser *Acacia* and *Allocasuarina* shrublands.

Heathlands are heterogenous communities without clear species dominance and, particularly in south-western Australia, are species rich with a high proportion of endemic species (Fig. 6.1). Shrub heath communities are generally treeless with an open upper layer of taller shrubs with Proteaceae species or scattered small mallee eucalypts less than 2m tall, and a closed lower layer of ericoid shrubs with mainly Myrtaceae species. Mallee eucalypt species are generally fewer and less common in northern than southern heathlands and where these are present communities are referred to as mallee heath (Beard, 1984). Where mallee eucalypts are more developed and dominant, the community is referred to as mallee shrubland. True heath communities consist of a single closed layer of ericoid shrubs generally less than 1m tall, but these occupy very small areas in the Avon Basin (VA49, 952 and 1147) and have been combined into this shrub heath vegetation class. These latter communities are usually dominated by *Hakea, Dryandra* and *Xanthorrhoea*. A small area of *Banksia* low woodland (VA949) has also been combined with the shrub heath vegetation class.

Table 6.1 Areas of Beard-Hopkins pre-European vegetation and remnant vegetation for 5 major and 3 minor vegetation associations (VAs) in the shrub heath vegetation class in the Avon Basin.

Dominant species	**Major VA**	**Minor VAs**	**Total pre-European area ha**	**Total Remnant Vegetation ha**	**Remnant vegetation %**
Scrub heath on sandplain	380		32,541	13,672	42.0
Banksia/ Xylomelum heath	694	949	154,113	5,285	3.4
Scrub heath SE Avon	1147	49, 952	41,748	2,740	6.6
Scrub heath Coolgardie	1148		195,575	124,900	63.9
Scrub heath Mallee	2048		305,701	144,311	47.2
	Total		**729,678**	**290,908**	**39.9**


(a)


(b)


(c)


(d)

Figure 6.1 Examples of shrub heath communities (a) in patches on heavy laterite within wandoo woodland; (b) with *Dryandra* and *Callitris*; (c) with *Xanthorrhoea* and Myrtaceae species on poor white sand; (d) with *Dryandra* over shallow granite rock.

Shrub heath communities generally grow on the upland lateritic sandplains which are highly leached sand over ironstone duricrust and partly weathered rock or clay. The degree of dissection of the residual lateritic plateau landscape varies greatly over the Avon Basin, so that areas of sandplain on the interfluves can be quite small and discontinuous. Sandplains of yellow loamy sand also occur on low ground near the Avon River and its tributaries. Laterite gravel may occur at the surface or at varying depths under the sand, and there are large variations in habitat in close proximity. Slight variations in soils and landform produce subtle differences in vegetation floristics and structure. These soils are very infertile and have poor water holding characteristics, and many plant species possess specialised root modifications to survive in this difficult environment.

The *Banksia-Xylomelum* alliance (VA694) which has various species of *Banksia* conspicuous in the upper layer grows on deep yellow sand or loamy sand, while *Banksia* low woodland (VA949) grows on leached sands in the higher rainfall areas. Where there is more ironstone gravel in the soil profile, then species of *Dryandra, Grevillea* and *Hakea* are more characteristic elements (VA49 and VA952) (Beard, 1984). Proteaceae species tend to be more numerous on areas with sand or clay over laterite, while Myrtaceae species are more numerous on deep yellow or white sand, with *Leptospermum erubescens* a common dominant species. Some larger shrub species of *Acacia, Allocasuarina* and *Melaleuca* that are dominant in shrublands may also occur in shrub heath communities, but they are more sparsely distributed.

A number of *Eucalyptus* mallee species occur in shrub heath as scattered emergents, but do not dominate the vegetation as in mallee shrubland (see above). For example, shrubland with tallerack (*E. pleurocarpa*) could be considered either mallee shrubland or shrub heath depending on the height and dominance of the upper mallee layer, which in some cases may be related to time since fire. Other species found in shrub heath include *E. foecunda, E. redunca, E. albida, E. burracoppinensis, E. leptopoda, E. macrocarpa, E. incrassata, E. plenissima* and *E. drummondii.*

6.2 Fire management in shrub heath

Shrub heath communities are highly flammable and may re-burn within 5-8 years before first flowering and seed set of slower maturing shrubs. Fires can ignite in any season due to rapid drying of fine fuels and travel rapidly burning out large areas. Floristic and structural diversity is high in shrub heath communities and prescribed burning should aim to use this heterogeneity to create a patchy mosaic at the local scale. Tolerable fire interval ranges for flora species are estimated from limited data to be around 15-30 years.

Shrub heath vegetation is highly flammable due to their well aerated fine fuels, a tendency for dead foliage to persist on plants and the direct exposure of fuels to wind and solar radiation without a tree canopy to retain surface moisture (Keith *et al*., 2002). Due to the large amount of elevated dead fine fuels, and rapid drying after rain, fine fuels may reach low moisture contents in relatively mild conditions. Litter fuels in canopy gaps consistently dried to a minimum moisture content of 5% or less in mild summer conditions when air temperatures were in the range 25-30°C (McCaw, 1997). These communities therefore remain fire prone throughout much of the year, unlike forest fuels which have deeper litter layers that remain moist for most of the year, or grasslands which have strong seasonal curing patterns.

Fuel loads in shrub heaths typically reach equilibrium within 10-15 years after fire but may continue to accumulate for more than 30 years on productive sites (Keith *et al*., 2002). Fuel loads at equilibrium range from 15tn/ha on poor sites prone to seasonal drought to 30tn/ha on more productive sites. The proportion of dead fuels in the total fuel load continues to increase with age, and while total fuel loads may reach equilibrium at 20 years, the flammability of this fuel continues to increase.

Fuel continuity has a major effect on fire behaviour and this depends on which layer of fuel dominates the spread process (Keith *et al*., 2002). Under marginal conditions for fire spread, flames will only be sustained in areas of more continuous fuels. Each layer contributes differently to fire spread and even within layers there are species differences, such as leaf size and shape and litter accumulation characteristics. Litter cover and load is greatest under tall shrubs and mallee, but dense vegetation and ground cover leads to more intense and continuous fires, especially at higher wind speeds. Less dense vegetation still ignites, but only at higher wind speeds.

Few shrub heath communities have continuous fuel in either vertical or horizontal directions, with considerable variation at a patch scale. This heterogeneity in vegetation combines with variation in wind speed to produce variation in fire severity at the local and landscape scale (Atkins and Hobbs, 1995). Under low to moderate winds (10-20km/hr) some unburnt or partially burnt patches remained. However, under intense weather conditions, as occurs in most summers in Western Australia, large areas of shrub heath can be burnt completely and little unburnt dead or live fuels remain.

Forward rates of fire spread are generally greater (3-6 fold) in shrub heath than in open eucalypt forest where the forest canopy greatly reduces wind speeds, but less than in grasslands (Keith *et al*., 2002). Spread rates of around 2.1m/s and fireline intensities of 35MW/m have been recorded over several hours under severe weather conditions, with greater values expected during peak fire behaviour ((Keith *et al*., 2002). In shrub heath with tall emergent shrubs and eucalypt mallees, flames may extend into and spread through the crown layer. This is more likely where shrubs have fibrous or long ribbon bark and abundant dead foliage accumulated on stems, as found more commonly in mallee shrublands. Crowning may occur when the forward rate of spread and fireline intensity reach 0.4m/s and 8.5MW/m respectively (McCaw, 1997). In experimental fires in mallee heath shrubland, fires spread freely in all plots where moisture content of the shallow litter was below 8% regardless of wind speed (McCaw, 1998).

Shrub heath communities may re-burn 5-8 years following a fire when fuel loads and continuity of fuels are sufficient to sustain fire spread (McCaw, 1998). This is less than the time required for many species to reach first flowering or replenish seed stores. Juvenile periods of most shrub heath species are unknown, but first flowering for a few representative species is around 4-7 years after fire, and even longer in drier regions and in times of drought (N. Burrows, unpublished data). Thus inter-fire periods need to be at least 15-21years (ie 2-3 times the longest juvenile period) to ensure adequate regeneration of all species in the community. Maximum longevity and seed viability of these species is also largely unknown so maximum tolerable fire interval can only be estimated from a few known species to be 30-50 years (Keith *et al*., 2002).

Impacts of inappropriate fire regimes on heath dependent fauna species have mostly been studied in the eastern states, but there have been some studies of bird habitats in Western Australia (Keith *et al*., 2002), which were discussed earlier in Part One Section 3.2.3. Fires can cause substantial fauna mortality rates, especially where the shrub heath landscape offers few refuge sites, such as moist creeklines or rock outcrops. Predation levels and starvation after fire can also be high leading to the temporary disappearance of some species. Recovery of vegetation and habitat attributes post-fire will vary with site productivity and this will largely determine the rate of recolonization of fauna species that depend on this vegetation for foraging, nesting and protection from predators.

Prescribed burning should aim to produce a patchy mosaic at the local scale wherever possible, using heterogeneity of vegetation and landscape discontinuities, to provide refuges for fauna during and after fire. Burning during mild weather and using late afternoon ignition will reduce the severity and extent of the burn. Wind driven burn buffers should also be used periodically to break up the areas of continuous vegetation where there are few natural discontinuities.

7. SUCCULENT STEPPE

7.1 Distribution of succulent steppe

Most of the remaining succulent steppe is found in the AW1 sub-region (35,194ha), with the main vegetation associations being VA325 and VA951 (Table 7.1). The main vegetation associations in the COO2 sub-region are VA147 and VA314, which are also found in AW1. Only relatively small areas of succulent steppe remain in the AW2 (2,824ha) and MAL2 (4,592ha) sub-regions. Most of the area with saltbush remains, (VA221, VA314 and VA325). Succulent steppe with sparse or open woodland and thickets of *Melaleuca* now covers about 26% of the original pre-European extent, while succulent steppe with thickets of *Melaleuca* or samphire have only about 12% of the original extent, possibly due to the impact of grazing and increasing salinisation. Overall, about 30% of this vegetation class remains although much of this is under threat of further degradation and loss from rising saline groundwater.

There are four mosaic vegetation associations with open woodland (mostly salmon gum) and succulent steppe, and one with *Melaleuca* scrub and samphire which could have been included in this vegetation class but have been dealt with under the mosaic vegetation class below. Swamp sheoak (*Casuarina obesa*) woodland (VA950) occupies only a small area in AW2 (190ha) and has been included in this vegetation class as it occupies a similar habitat to succulent steppe.

Succulent steppe refers to saltbush and samphire communities consisting of semi-succulent and succulent low shrubs with 30-70% cover which occur in saline areas (Fig. 7.1). Saltbush and samphire communities may mix or occur singly and may form a ground layer only or be associated with *Melaleuca* shrubs and/or open *Eucalyptus* woodland (Beard, 1980b). Samphire flats also grow at some distance from the parent lake growing on poor soils of high alkalinity (Datson, 2002). In the Avon Basin, this vegetation class is most commonly associated with salt lake chains and major saline drainage lines and is therefore quite restricted in extent. The salt lakes themselves are seasonally flooded and dry in summer being covered with salt crystals and usually devoid of vegetation, although some salt lakes are permanent. Around these lakes, and covering extensive flats of saline mud, are samphire communities which are less prone to inundation. Surrounding these samphire communities, often on sandy rises, are thickets of *Melaleuca* grading into open or sparse woodland with species of saltbush, samphire or *Melaleuca* in the understorey. Samphires also grow in and around claypans which are often freshwater sumps associated with larger lakes and usually elevated above them (Datson, 2002).

The most common *Melaleuca* species include *M. hamulosa, M. uncinata, M. lateriflora* and *M. thyoides. Callitris canescens, C. columellaris, Hakea preissii, Acacia multispicata* and *A. acuminata* may grow with *M. hamulosa* on nearby gypseous dunes. *Melaleuca* thickets often grow to 4-6m tall with dense canopies which exclude most understorey species. More open stands can be seen to 6-7m tall which may be reaching senescence and these have a limited number of understorey species (Beard, 1980c).

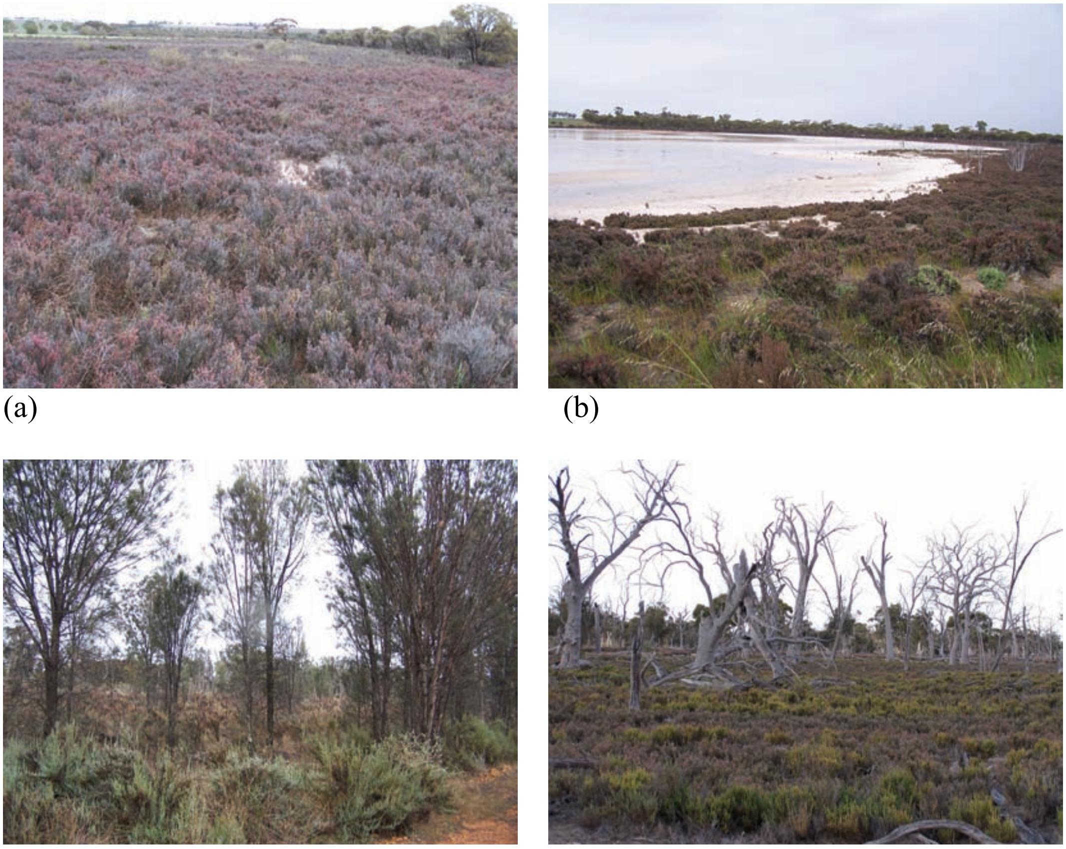
Samphire species are small compact semi-succulent shrubs from 0.2-1.0m tall. They grow in different niches around saline wetlands, some preferring well drained soils (eg *H. indica* subsp. *bidens*) and others (eg *H. halocnemoides* subsp. *caudata*) only found on waterlogged saline clays (Datson, 2002). Species growing on lake 'beaches' are often quite different to those found growing on adjacent aeolian dunes or in claypans. Other common samphire species include *Halosarcia doleiformis, H. finbriata, H. pergranulata,* Samphire seeds germinate freely after heavy rain when the soil is wet with relatively fresh water. The major threats to samphire communities are prolonged inundation or drought, or damage by vehicles or animals (Datson, 2002). Samphires have been grazed indiscriminately by sheep, particularly in times of drought, and have caused much damage to this brittle vegetation. Changes in hydrology following widespread clearing for agriculture have caused the decline and destruction of many salt lake vegetation communities. Often the *Melaleuca* thickets have died leaving only the succulent samphire species which have either survived or colonised the salt lake margins.

Table 7.1 Areas of Beard-Hopkins pre-European vegetation and remnant vegetation for 10 major and 6 minor vegetation associations (VAs) in the succulent steppe vegetation class in the Avon Basin.

Dominant species	Major VA	Minor VAs	Total pre-European area ha	Total Remnant Vegetation ha	Remnant vegetation %
Acacia/ saltbush	147	1071	36,248	12,616	34.8
Saltbush	221		3,144	3,144	100
York gum/ saltbush	314		6,391	6,394	100
Saltbush/ samphire	325		7,920	7,464	94.2
York gum/ *Melaleuca*/samphire	631		11,812	3,916	33.2
York gum/ *Melaleuca*/samphire	951		27,508	8,441	30.7
Melaleuca/samphire	953		9,457	1,432	15.1
Yorrel/ *Melaleuca*/samphire	959		13,093	4,003	30.6
Melaleuca/samphire	988	392, 950, 1080	54,126	3,671	6.8
York gum/ *Melaleuca*/samphire	1062	356, 676	29,867	6,488	21.7
	Total		**199,566**	**57,569**	**28.8**

Saltbush species generally require reasonably well drained soils but will grow well on salt affected land in the 200-500mm annual rainfall region. Common saltbush species include *Atriplex paludosa*, *Enchylaena tomentosa* and bluebush *Maireana brevifolia*, the latter species requiring well drained soil that is only marginally saline, and grows with *Eucalyptus kondininensis* and *E. longicornis* woodland (Runciman and Malcolm, 1989). Chenopods are usually obligate seeders with low flammability, no effective post-fire seed bank and short range seed dispersal. The seeds do not have thick endocarp walls to protect the embryo from lethal temperatures during fires (Hodgkinson and Griffin, 1982).

The *Acacia* scrub with saltbush understorey community (VA147 and VA1071) occurs on sandy soils near salt lakes in the COO2 and AW1 sub-regions in the north and north-east of the Avon Basin. The main species are *Acacia ramulosa* var. *linophylla, A. tetragonophylla, Callitris huegelii, Hakea preissii, Grevillea sarissa* and *Eremophila* spp., along with saltbush species *Atriplex paludosa* and *A. nummularia,* succulents and ephemerals (Beard, 1972a). The *Acacia* species also occur as open shrubland on banded ironstone ranges and on granite but there have quite different understorey shrubs (Beard, 1972a).



(a) (b)

(c) (d)

Figure 7.1 Succulent steppe communities associated with a saline environment (a) samphire flat; (b) samphire around a salt lake; (c) open *Allocasuarina* woodland with samphire and saltbush understorey on upper bank of salt lake; (d) dead wandoo woodland with samphire understorey severely affected by rising saline groundwater.

Open or sparse woodland species with succulent steppe include *Eucalyptus loxophleba* on sand dunes around salt lakes, and *E. kondininensis, E. salubris, E. salmonophloia, E. longicornis* and *E. gracilis* on saline flats and valleys. These woodland species may occur as patchy denser stands within the succulent steppe which are included in the mosaic vegetation class below. Understorey is usually *Melaleuca uncinata* or *M. hamulosa* with *Hakea preissii, Santalum acuminatum* and various saltbush and samphire species (Beard, 1980b, 1980c). Some grass species (eg *Austrostipa*) and ephemerals may be common on sandy dunes around salt lakes, often with introduced grasses.

7.2 Fire management in succulent steppe

Succulent steppe vegetation communities generally have low flammability and will not carry fire unless there is an abundance of grasses and ephemerals after heavy rain. Some adjoining open woodland and shrubland vegetation may burn patchily during a fire, but there is no strong reason to prescribe fire for these communities. The adjoining salt lake systems may provide a natural barrier for fire and can be used for this purpose in fire management planning.

Fire is not required for seed germination in semi-succulent vegetation communities of samphire and saltbush. This vegetation will generally not carry fire due to the moist soil conditions, high percentage of bare soil between shrubs and the high water and salt content of the foliage which results in low flammability (Hodgkinson and Griffin, 1982). Other vegetation types with succulent steppe understorey, such as *Melaleuca* thickets, *Acacia* open scrub and *Eucalyptus* open woodland may need some fire very occasionally to regenerate the upper canopy species, but the open nature of this canopy and the semi-succulent understorey will in most cases prevent fires from spreading, and will promote a natural mosaic burn pattern. Only after heavy rains when there is abundant growth of grasses and ephemerals on surrounding salt lake dunes and sandy ridges will there be sufficient continuity of ground fuels to carry a fire.

It is therefore recommended that fire is not introduced into these vegetation communities and it is unlikely that fire will need to be actively suppressed. Rather, these salt lake systems can act as natural barriers to fire. Burning away from the salt lake systems into surrounding vegetation may provide a useful strategic firebreak during prescribed burn and wildfire suppression operations. There are numerous small and large reserves with salt lakes and succulent steppe surrounded by agricultural land. In most cases, these can be left unburnt unless there is other vegetation in the reserve that requires fire for regeneration. It is recommended that any fires around salt lake systems be closely observed to understand how fire behaves when approaching from adjoining woodlands and shrublands.

8. MOSAIC VEGETATION

8.1 Distribution of mosaic vegetation

Mosaic vegetation communities are a heterogenous vegetation class comprising mixtures of several vegetation associations at a scale that is too small for broad scale mapping. They generally occur in broad valley systems between woodland, mallee shrubland and succulent steppe or between woodland and shrubland (Fig. 8.1). Most of the remaining areas are found in the central AW1 and MAL 2 sub-regions (Table 8.1). Two vegetation associations (VA214 and VA941) occur in the COO2 sub-region, the former covering 100% of the original extent. The most heavily cleared vegetation community (VA131) is salmon gum and gimlet medium woodland with redwood mallee and shrubland where only 5.2% remains. Other heavily cleared mosaic vegetation associations include the shrub heath with tammar sheoak in the south-east Avon (VA955 and VA2047) with 7.8% remaining, and salmon gum and morrel medium woodland with tall sand mallee shrubland (VA486 and VA1200) with only 8% remaining. Open and sparse salmon gum woodland with succulent steppe (VA1061, VA1079 and VA1098) are moderately well represented.

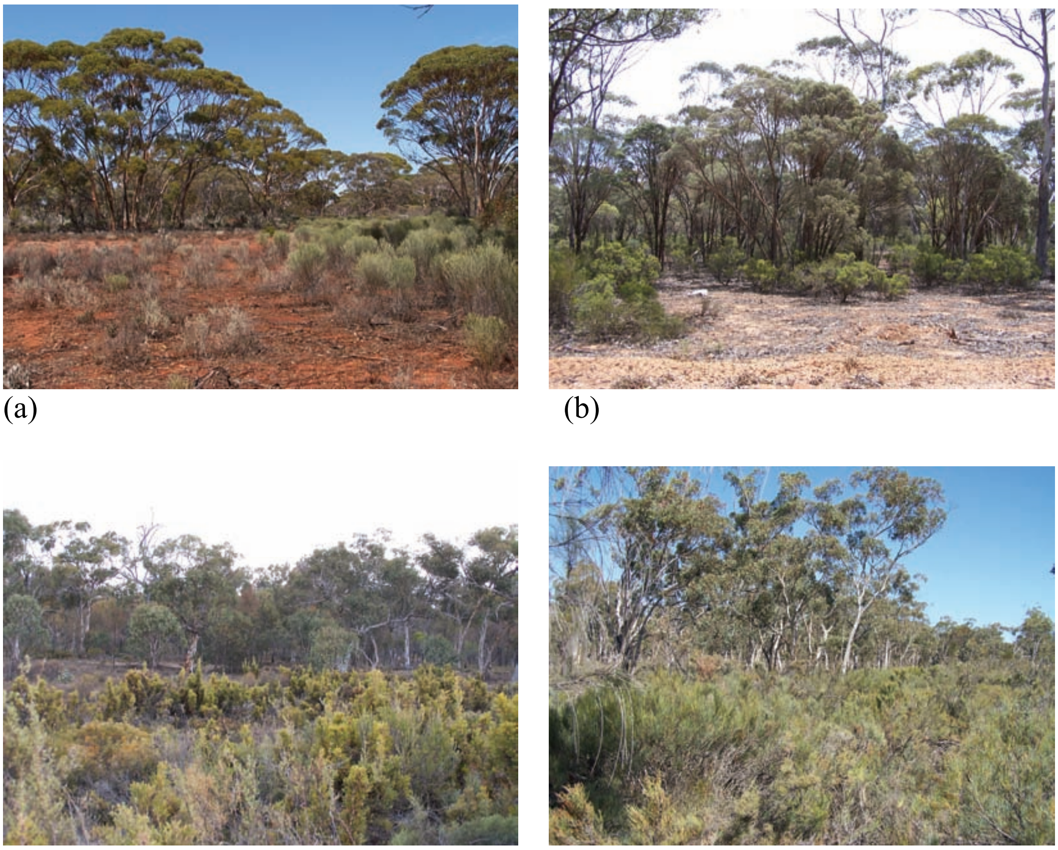
Table 8.1 Areas of Beard-Hopkins pre-European vegetation and remnant vegetation for 11 major and 7 minor vegetation associations (VAs) in the mosaic vegetation class in the Avon Basin.

Dominant species	Major VA	Minor VAs	Total pre-European area ha	Total Remnant Vegetation ha	Remnant vegetation %
Salmon gum/ gimlet/ redwood mallee	131		171,466	8,932	5.2
Salmon gum/ morrel/redwood mallee	941		34,257	14,536	42.4
Salmon gum/ redwood mallee	945	486, 942, 1076, 1094	176,850	22,428	12.7
Shrub heath/ tammar thicket	955	2047	132,023	10,359	7.8
Shrub heath/ tammar thicket	961		27,389	4,276	15.6
Melaleuca/ samphire	1048		13,815	2,373	17.2
Salmon gum/ gimlet/ York gum mallee	1057		145,312	13,588	9.4
Salmon gum/ yorrell/ saltbush	1061	214	58,462	12,567	21.5
Salmon gum/ morrel/saltbush	1079		10,119	3,881	38.4
Salmon gum/ morrel/samphire	1098	966	20,756	3,349	16.1
Salmon gum/ morrel/ mallee	1200		102,557	8,102	7.9
	Total		**893,006**	**104,391**	**17.6**

8.2 Fire management in mosaic vegetation

Mosaic vegetation units should be burned according to the most fire sensitive type of vegetation within them, aiming to maintain the structural diversity. Open salmon gum woodlands with saltbush and samphire will not readily burn, but may need to be regenerated by fire at some time. Salmon gum with mallee shrubland is more flammable and care is needed not to kill the sensitive woodland species. Fire should be encouraged to burn in a mosaic according to the differential flammability of the vegetation patches.

These mosaic vegetation associations could be amalgamated into their respective vegetation classes using just the dominant two or three species, but the distribution of the vegetation communities within them is more clumped or patchy and fire is likely to behave somewhat differently. This may not be a problem as fire will tend to burn in a more mosaic pattern which is desirable. It is quite possible that the patchy nature of this vegetation has resulted from previous mosaic burning patterns. At this stage they have been mapped separately but this may be reviewed in the future. The mosaic vegetation associations generally form an intermediate zone between other major vegetation associations where the boundaries of different vegetation types intermingle and probably expand and contract with environmental conditions and competitive factors. Fire management should consider the needs of all components and aim to maintain this more complex structural diversity and prescribe for the most fire sensitive vegetation unit within the mosaic.



(a) (b) (c) (d)

Figure 8.1 Examples of mosaic vegetation communities showing patches of distinctly different vegetation structures that are too small to map other than at a local scale (a) salmon gum woodland with low shrubland of saltbush and *Acacia*; (b) Woodland interspersed with mallee shrubland ; (c, d) wandoo woodland with small patches of *Dryandra* shrub heath.

Mosaic vegetation associations with a succulent steppe understorey have a low requirement for fire and will not normally carry fire. Fire management in salmon gum open woodland with succulent steppe would be as for salmon gum woodlands, while management of fire in *Melaleuca* shrubland with succulent steppe would be as for succulent steppe. Salmon gum woodland with mallee shrubland is more complex and depends on the mix of species present. Protection of the fire sensitive salmon gum and gimlet woodlands areas should be a high priority, except where regeneration of these species has been prescribed. Shrub heath with sheoak shrubland should be considered for fire management along with other sheoak shrublands where the tammar sheoak dominates, or as shrub heath where this community dominates.

Adequate field assessment of these vegetation communities is essential before fire planning is initiated. Monitoring of fire behaviour during prescribed burning and wildfires will provide valuable evidence for a range of vegetation communities and for the interface between different communities.

9. ROCK OUTCROPS

9.1 Distribution of rock outcrops

Rock outcrops (VA128) have been mapped as bare areas by Beard in the pre-European vegetation mapping and include granite boulders and pavements as well as ironstone ridges, but may also include some other bare areas such as gravel pits and lateritic breakaways which were difficult to differentiate in aerial photography. They occur throughout the Avon Basin and are generally only small in size as individual units but together cover 136,209ha. The area mapped as remnant vegetation within this vegetation association is 42,924ha but it is not known how much of the original area mapped as rock outcrop included fringing or associated vegetation so it is not possible to determine how much has been cleared. The Coolgardie IBRA sub-region has about 15,494ha, Avon Wheatbelt 1 has 14,255ha, Mallee 2 has 11,682ha and Avon Wheatbelt 2 has 1,490ha of remaining rock outcrop vegetation. Associated vegetation grows both on the outcrops in crevices and where soil has accumulated, between rock outcrops and boulders, and as fringing vegetation which could not be mapped separately (Fig. 9.1).


(a)


(b)

Figure 9.1 Granite rock outcrops and their fringing vegetation which receive increased water flows (a) run-off area below granite with rock sheoak (*Allocasuarina huegeliana*); (b) dense sheoak run-off vegetation below granite before shrub heath community on shallow granite.

Many nature reserves in the wheatbelt were originally kept as water reserves where water was harvested from the sloping rock face of granite outcrops (Laing and Hauck, 1997). These water reserves have fortuitously conserved the best of the granite outcrops (Beard, 1972b). The value of granite outcrops as sources of water was recognised by early explorers and public water supplies were later developed from runoff at about 200 locations in the wheatbelt. Bores, wells and soaks were also developed in land adjacent to many outcrops to utilise the runoff water (Laing and Hauck, 1997). Aboriginal people valued gnammas or rock holes for the fresh water and fauna that was attracted to it. The granite domes provided a focal point in the landscape for cultural and ceremonial activities (Bindon, 1997). Two important food trees the kurrajong (*Brachychiton gregorii*) and quandong (*Santalum acuminatum*) are commonly found around granite outcrops and lizard traps were used to trap lizards on granite surfaces.

Granite outcrops provide a wide range of microhabitats over short distances that are not present in the surrounding landscape and this often results in structurally diverse and species-rich vegetation (Yates *et al.,* 2003a). Outcrops in the south-west have high species turnover within and between rocks and very high levels of endemism and rarity. Protection of these rare and threatened species and communities is a major challenge for conservation and knowledge of their response to fire is paramount. The impacts of fragmentation, grazing and weed infestation are beginning to be understood, but the interactive effects of these and fire management are critical for the survival of many threatened species both on and off the conservation reserve.

In the north-west of the Avon Basin, sandplains are dominated by outcropping granite domes with *Allocasuarina huegeliana, Allocasuarina campestris, Acacia acuminata, Melaleuca radula, Eucalyptus loxophleba, Thryptomene australis* and *Verticordia preissii* as surrounding vegetation in the runoff zone or where there are soil pockets within the rock outcrop. On rock slabs there is *Borya nitida, Lepidosperma* spp., annual grasses, geophytes and herbs in moss mats, and *Kunzea pulchella* and *Calytrix angulata* in rock crevices (Beard, 1979a).

In the south-central parts of the Avon Basin, large domes of granite and gneiss are a common feature, with much bare rock surrounded by shallow stony and gritty sandy soils and pockets of deep soil. Around the outcrop may be shrubland or low woodland of *Allocasuarina huegeliana* and *Acacia acuminata* on shallow sandy soils receiving runoff, with *Leptospermum erubescens, Dodonaea viscosa, Gastrolobium spinosum, Hakea petiolaris* and *Hibbertia hemignosta* in the understorey. On large rock outcrops (eg Hyden Rock) pools of water fill in winter and support ephemeral life forms. A number of species grow in crevices and shoulders where some soil accumulates and may be confined to this habitat including *Acacia lasiocalyx, Calothamnus quadrifidus, Daviesia benthamii* subsp. *acanthoclona*, *Kunzea pulchella, Melaleuca radula, M. lateriflora* and *Thryptomene australis.* Within some granite complexes there may also be shrub heath and thickets (Beard, 1972b, 1980b).

In the drier north-east parts of the Avon Basin, there are numerous large bare areas of outcropping granite where the duricrust has been eroded, typically on the middle slopes. Where soil has accumulated on rock outcrops there are some characteristic species including *Thryptomene australis, Melaleuca fulgens, M. uncinata, Grevillea paniculata, A. multispicata, A. lasiocalyx, Eucalyptus caesia, E. crucis, E. websteriana, Santalum acuminatum, Diplolaena microcephala, Hakea preissii* These may be surrounded by *Eremophila oldfieldii, E. clarkei,* and *Dodonaea inaequifolia* as well as *Acacia brachystachya* and *A. tetragonophylla* open shrubland (Beard, 1972a, 1979b, 1980a).

Banded ironstone and greenstone ranges in the eastern areas of the Avon Basin form abrupt rocky hills with little soil. Some of these have dense thickets of *Allocasuarina campestris, Allocasuarina acutivalvis, Acacia quadrimarginea, Santalum spicatum, Calothamnus asper, Hakea* spp., *Dryandra arborea*, *Callitris preissii* and other shrubland or shrub heath species which vary in composition from one range to another (Beard, 1972a, 1979b). Many cryptogams, including mosses, liverworts, fungi, lichens and algae are present on most if not all granite outcrops and are an important component of the biodiversity as well as being critical in the functioning and break down of rock crystals into soil particles on which all else depends. There are common and rare species of cryptogams, but we have poor knowledge about their taxonomy and distribution. Most species are vulnerable to disturbance, including vehicles driving over the rock surfaces and footprints (George, 2000). In extreme fire events they can be scorched from the surface of rocks.

Granite outcrops also provide habitat for fauna and invertebrate species, some being endemic to this environment, others able to use the many sheltered niches, water holes and various surrounding vegetation types (Withers and Edward, 1997) (Fig. 9.2). A number of frogs, snakes, lizards, spiders and scorpions are mainly found in granite outcrops including the ornate dragon lizard, Gould's hooded snake, Stimson's python, various skinks and geckos and about 50 species of aquatic invertebrates (Pinder *et al*., 2000). Some frog species use the rock pools for the egg and tadpole stages and shelter in rock crevices. Many bird species are attracted to granite outcrops by the diversity of flowering species in the surrounding vegetation and the availability of fresh water in rock holes.

Figure 9.2 Rock pools are an extremely important water source for fauna as well as a microhabitat for aquatic invertebrate and frogs.

9.2 Fire management of rock outcrops

Vegetation surrounding granite rock outcrops and ironstone ranges are thought to be less prone to fire, and may provide refugia for many fire sensitive species. These areas will burn when fuel loads are sufficient and frequent fire can result in local extinctions of many endemic and rare obligate seeders. Fire regimes must consider the fire responses of the most sensitive species and be implemented with great caution. Vegetation surrounding localised rock outcrop communities should be managed to reduce the incidence of wildfires.

The high levels of species richness and endemism in granite outcrop communities may be due to the heterogenous microhabitats and the incidence of major disturbances such as sheet flooding, soil erosion, storm damage and drought. In addition, fire is a significant disturbance event in the wheatbelt, with fire regimes varying with location, degree of fragmentation, surrounding farming practices, rainfall, fuel loads and length of dry season (Yates *et al.,* 2003a). Granite outcrops and ironstone ridges may provide natural firebreaks, depending on size and extent of bare rock, and may be burnt less frequently than surrounding vegetation, thus providing a refuge for more fire sensitive species (Yates *et al*., 2003b). Granite outcrops typically have a high proportion of obligate seeders which may indicate that the flora has evolved under relatively long fire intervals (Hopper, 2000). There may also be less dependence on fire for seedling germination in granite rock flora as has been shown with the rare endemic species *Verticordia staminosa* ssp. *staminosa* (Yates *et al*., 2003b) compared with congeneric species in surrounding shrub heath or woodland vegetation.

However, fire can also be frequent. At Chiddarcooping Nature Reserve, fires occurred in 1975, 1978, 1987 and 2000 and were initiated by lightning strike other than the 1975 fire which started in adjoining farmland (Hopper, 2000; Yates *et al*., 2003a). This reserve has high biodiversity conservation value with possibly 600 species of vascular plants and several rare, endemic and relictual species (Hopper, 2000). At this reserve, 75% of all species were obligate seeders, including a high proportion of annuals. There were dramatic differences within and between species in seedling recruitment following the last two fires.

For the obligate seeder *Allocasuarina huegeliana,* seedling densities were high (55 seedlings/m^2) after the 1987 fire but rapidly thinned in the first 4-5 years and by the 2000 fire, only 13.7% of seedlings were alive. For *Hakea petiolaris* ssp. *trichophylla,* seedling density was much lower after the 1987 fire (3 seedlings/ m^2) but survival rates were higher so that by the 2000 fire, 78% of seedlings were alive. However, all plants of both species which recruited following the 1987 fire were killed by the 2000 fire (Yates *et al.*, 2003a). Following this fire, seedling densities were again high for *Allocasuarina huegeliana,* although less than after the 1987 fire, while seedling densities for *Hakea petiolaris* ssp. *trichophylla* were very low. This, combined with the 100% mortality of seedlings following the 2000 fire, resulted in almost complete regeneration failure for *H. petiolaris* ssp. *trichophylla* leading to likely elimination of this species from patches where it was previously abundant, and replacement by *Acacia lasiocalyx (*Yates *et al*., 2003a). The low seedling mortality rates may have been due in part to the low winter rainfall following the 2000 fire. Patches of *H. petiolaris* ssp. *trichophylla* that escaped one or both fires are likely to retain some plants that may aid recolonisation and buffer against local extinction from this reserve. These large differences in seedling recruitment following repeated fires demonstrate that fire can have a significant impact on floristic composition, particularly for obligate seeder species.

For resprouter *Eucalyptus* mallee species, seedling density following the 1987 fire at Chiddarcooping Nature Reserve was greater for *E. petrea* (52 seedlings/ m^2) than for *E. crucis* ssp. *lanceolata* (7 seedlings/m^2) or *E. caesia* ssp. *magna* (4 seedlings/ m^2) (Yates *et al.*, 2003a). However, there was rapid mortality of seedlings in the first 2-3 years and only 0.4%, 0% and 2.3% of seedlings were alive by the 2000 fire for these species, respectively. A small number of recruits of *E. petrea* and *E. caesia* ssp. *magna* survived the 2000 fire as well as all adults studied. There was no decline in the number of resprouting stems for these three species in the first year following the fires in 1987 and 2000 and although there was a rapid decline in the number of resprouting stems in the three years after the 1987 fire, there were still more stems per adult plant following the 2000 fire. This demonstrates the greater survival capacity of resprouters compared with obligate seeders in the face of repeated fires even 13 years apart in a low rainfall area (300mm annual rainfall).

At Chiddarcooping Nature Reserve, there was a high number of native annual species (49 spp) and geophytes (11 spp) as well as 17 species of exotic annuals. These responded to the fires and contributed to the greatest species diversity in the first year following fire when canopy cover was much reduced and resources were more available (Yates *et al*., 2003a). Species numbers for geophytes were highest in the fourth year after the 1987 fire, while one orchid species (*Cyanicula ashbyae*) was only recorded in the first year following the 1987 fire. There was a significant turnover of geophyte species, six orchids and one sundew recorded before 2000 were not recoded after the fire, and several only appeared for one year.

Many granite outcrop reserves are infested with introduced annual grasses which can greatly increase the fine fuel loads during summer. This will increase the intensity of the fire and provide continuity of ground fuels that might otherwise not be present, so that larger areas of surrounding vegetation are burnt. In many cases, this means that the whole reserve burns rather than small patches, which can destroy the last remaining refuge in the farming landscape for many native animals. More intense fire may carry further up into protected rock shelves and crevices where fire sensitive species would otherwise escape fire (Fig. 9.3). Fire also stimulates annual weeds to germinate and increase in area, further exacerbating the problem. For example, a granite endemic, *Pimelea graniticola* is being threatened by invasion of weeds at Yilliminning Rock (Piggott and Sage, 1997). Therefore, it is important that weed management is implemented prior to burning granite outcrop vegetation, particularly those with known high conservation value.

(a) (b)

Figure 9.3 Intense fires can penetrate vegetation on granite rock outcrops, (a) vegetation in rock crevices, shelves and valleys may still be burnt; (b)*Kunzea pulchella* only grows high on granite rocks where it may escape fire.

Surveys of granite outcrops have suggested that the age of some fringing communities, including woodlands of *Allocasuarina huegeliana* and *Acacia lasiocalyx* to be 80-100 years (Chapman and Newby, 1987). Granite outcrop communities on Middle Island in the Recherche Archipelago escaped fire for 150 years and there was no loss of species when the area was eventually burnt. Some of these species were fire ephemerals (eg *Alogyne hakeifolia*) that only appeared for a few years following the fire, which suggests that they remained as viable seed in the soil seed bank for a very long time (Weston, 1985). This information is important in that it demonstrates that it is not essential to prescribe fire for regeneration as soon as species show signs of senescence, as they can continue to survive for many years as soil borne seeds. Obviously, knowledge of seed viability in the soil seed bank is critical for this course of action.

In many cases, rock outcrop communities are surrounded by more flammable and less fire sensitive vegetation such as low woodland of rock sheoak and jam, shrublands or heathland. To avoid serious damage to the more fire sensitive vegetation in rock outcrop communities, it is recommended that some fire be introduced more frequently to the surrounding vegetation and burnt away from the rock outcrop to provide a greater level of fire protection to the rock outcrop itself. This should be done under mild weather conditions in spring or late autumn with the appropriate wind direction. Obviously only one side of a rock outcrop could be burnt in one operation under these conditions. There may then be a requirement for very occasional fire to regenerate some of the species within the rock outcrop community. A draft DEC Guideline for managing fire in granite outcrops in south-west forests has been recently developed and should be consulted before planning fire in these areas (Burrows, 2007).

10. SALT LAKES AND WETLANDS

10.1 Distribution of salt lakes and wetlands

Salt lake systems (VA125) are a common feature of the Avon Basin and have been mapped in the Beard-Hopkins pre-European vegetation mapping as bare areas. The original pre-European area for salt lakes in the Avon basin mapped by Beard was 256,437ha. The area currently mapped as remnant vegetation for salt lakes is 59,725ha but it is not known what the actual extent of fringing vegetation was around the salt lakes when first mapped so it cannot be determined how much has been cleared or lost as a result of increasing salinisation. The majority of the salt lake area with remnant vegetation is in the Coolgardie sub-region (35,778ha), while Avon Wheatbelt 1 sub-region has 17,487ha. There is only a small area of claypans (VA 1271) mapped as bare areas (1,095ha) with 532ha of remnant vegetation in the Avon Basin. The description of the vegetation surrounding salt lakes and fire management guidelines has been included in that for succulent steppe (Section 7).

Wetlands other than salt lakes, claypans, succulent steppe or *Melaleuca* thicket do not appear to have been mapped separately by Beard-Hopkins in the Avon Basin and so have not been analysed in the same way as other vegetation classes. They are generally small isolated occurrences such as run-off areas below rock outcrops, perched fresh or brackish wetlands or deep fresh pools along the tributaries of the Avon River. A few larger lake systems are still fresh or brackish but are under imminent threat of salinisation eg Lake Toolibin and Lake Bryde.

Biodiversity is generally greater in fresh water wetlands than in saline wetlands, with greater species richness of plants, invertebrates and birds in particular (Halse *et al.,* 1993; Lyons *et al.,*2004). This decline in biodiversity with increasing salinisation is mainly due to the increased salinity levels and/or to increased water levels and waterlogging which kills the fringing vegetation. The presence of dead trees and shrubs surrounding many wheatbelt wetlands is evidence of the effects of increasing salinity and waterlogging since land clearing. For example, Lake Noonying which has only recently become saline has a dense stand of recently dead swamp sheoak (*Casuarina obesa*) on the lake bed with live saplings just above and below the water mark. Often more salt tolerant shrubs replace the original vegetation.

Lake Toolibin is brackish wooded wetland and still retains dense thickets of swamp sheoak and *Melaleuca strobophylla,* which is regarded as a critically endangered ecological community. In more southern areas, flat topped yate (*Eucalyptus occidentalis*) grows in fresher wetlands in a similar position but there are few areas remaining. Lake Bryde is a wetland of national significance and has a low shrub cover over the lake bed of the rare species *Muehlenbeckia horrida* subsp. *abdita* with scattered *Tecticornia verrucosa,* which is also listed as a critically endangered ecological community. Flooded gum (*Eucalyptus rudis*) also prefers fresh to brackish water and grows mainly to the west and south of the Avon Basin.

The vegetation associations that may contain swamp sheoak are VA950 and VA988 and the total area remaining of these is about 5,789ha, found only in the Avon Wheatbelt IBRA region (Avon Catchment Council, 2005). The vegetation associations that may contain flat topped yate are VA131 and VA931 in the Mallee sub-region, with a total area of about 9,546ha. The actual areas of fresh wetlands within these vegetation associations would be much less than these total areas. One of these areas (VA988) has been included under the succulent steppe vegetation class above.

Other fringing vegetation species found around fresh to brackish wetlands include *Baumea articulata, Halosarcia pergranulata*, *Juncus pallidus, Lepidosperma tenue, Melaleuca lateriflora, Melaleuca rhaphiophylla, Melaleuca strobophylla, Melaleuca teretofolia, Muehlenbeckia sp.*, *Sarcocornia blackiana, Tecticornia verrucosa* and *Typha domingensis*. Smaller perched clay wetlands and soaks may have additional species but they have not been accessed for these guidelines. Vegetation associated with rock pools (eg Yorkrakine rock pools) and run-off seepages have been discussed earlier under the rock outcrops section.

10.2 Fire management of wetlands

Wetland vegetation will burn if sufficiently dry, but often contains flora species that are sensitive to fire and generally should be protected. Fringing vegetation also provides important refuge, feeding and breeding sites for birds and frogs which feed on aquatic invertebrate. Inappropriate fire may increase the threat to this vegetation posed by salinisation and increased water levels.

The fringing or riparian vegetation associated with fresh and brackish wetlands is more flammable than succulent steppe communities around salt lake systems. Some wildfires have been known to burn through the crowns of wetland vegetation while there is still free standing water below. Many of the *Melaleuca* species including *Melaleuca rhaphiophylla* have the ability to respond to fire by resprouting as does swamp sheoak and flat topped yate, but the most appropriate fire regimes are not well known.

With a drying climate and increasing salinisation there is a greater risk of fire carrying into this fringing vegetation as it will contain greater amounts of fine and coarse dead material. It may be critical to protect these vegetation communities from fire, particularly as some are listed as critically endangered due to the threat of salinity. However, there may also be a role for fire in regenerating some of these communities beyond the zone of rising saline water tables and this needs to be carefully and urgently researched. Seedlings of some species may have a lower tolerance to increasing salinity levels than adult plants, and fail to regenerate after fire.

Freshwater wetlands, seepage areas and rock pools in the wheatbelt are extremely valuable refuges for a range of aquatic invertebrate, fauna and birdlife. Retaining healthy fringing vegetation, particularly sedges and rushes, is critical for many nesting birds and frogs. For example, Lake Toolibin is very important breeding area for freckled duck, Australasian bittern and purple swamphen, as well as a wide range of other water birds. These important fauna habitats should be protected from fire if a prescribed burn is planned in the surrounding area.

11. 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. Pp 119-146. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Armstrong R 2004 Ecologically based fire management- Integration of multiple land use objectives. 11th Annual Conference and Inaugural Bushfire CRC Conference, 7-9th October, 2004. Perth.

Atkins L and Hobbs RJ 1995 Measurement and effects of fire heterogeneity in south-west Australian wheatbelt vegetation. CALMScience Supplement 4, 67-76.

Auld TD 1986 Population dynamics of the shrub *Acacia suaveolens* (Sm.) Willd.: dispersal and the dynamics of the soil seed bank. Aust. J. Ecol. 11, 235-254.

Auld TD 1987 Population dynamics of the shrub *Acacia suaveolens* (Sm.) Willd.: survivorship throughout the life cycle, a synthesis. Aust. J. Ecol. 12, 139-151.

Auld TD and O'Connell MA 1991 Predicting patterns of post-fire germination in 35 eastern Australian Fabaceae. Aust. J. Ecol. 16, 53-70.

AUSLIG 1990 Atlas of Australian resources: Vegetation. Aust. Gov. Printing Service, Canberra.

Avon Catchment Council 2005 The Avon natural resource management strategy. The regional natural resource management strategy for the Avon River basin. Avon Catchment Council.

Barker P 2001 A technical manual for vegetation monitoring. Resource Management and Conservation, Department of primary Industries, Water and Environment, Hobart.

Barrett S, Broomhall G, Comer S, Freebury G and Grant M 2004 Draft fire management strategy for the Stirling Range National Park. Department of Conservation and Land Management. Unpublished Report.

Beard JS 1969 The vegetation of the Boorabbin and Lake Johnston areas, WA. Proc. Linn. Soc. NSW 93, 239-269.

Beard JS 1972 (a) The vegetation of the Jackson area Western Australia. Vegmap Publications, Sydney.

Beard JS 1972 (b) The vegetation of the Hyden area Western Australia. Vegmap Publications, Perth.

Beard JS 1973 The vegetation of the Ravensthorpe area, Western Australia. Vegmap Publications, Perth.

Beard JS 1979 (a) The vegetation of the Moora and Hill River areas Western Australia. Vegmap Publications, Perth.

Beard JS 1979 (b) The vegetation of the Southern Cross area Western Australia. Second edition. Vegmap Publications, Perth.

Beard JS 1980 (a) The vegetation of the Bencubbin area Western Australia. Vegmap Publications, Perth.

Beard JS 1980 (b) The vegetation of the Kellerberrin area Western Australia. Vegmap Publications, Perth.

Beard JS 1980 (c) The vegetation of the Corrigin area Western Australia. Vegmap Publications, Perth.

Beard JS 1980 (d) The vegetation of the Dumbleyung area Western Australia. Vegmap Publications, Perth.

Beard JS 1981(a) The vegetation of the Newdegate and Bremer Bay areas Western Australia. Second edition. Vegmap Publications, Perth.

Beard JS 1984 Biogeography of the Kwongan. *In* Kwongan - Plant life of the Sandplain. Eds JS Pate and JS Beard, pp 1-26. University of Western Australia Press, Nedlands.

Beard JS and Sprenger BS 1984 Geographical data from the vegetation survey of Western Australia. Vegmap Publications, Perth.

Beecham BC 2003 Avon Wheatbelt 1 (AW1 – Ancient Drainage sub-region). *In* A biodiversity audit of Western Australia's 53 biogeographical subregions in 2002. Pp 7-35. Department of Conservation and Land Management, Perth.

Beecham BC and Danks A 2003 Mallee 2 (MAL2 – Western Mallee subregion). *In* A biodiversity audit of Western Australia's 53 biogeographical subregions in 2002. Pp 435-465. Department of Conservation and Land Management, Perth.

Bell DT, Hopkins AJM and Pate JS 1984 Fire in the Kwongan. *In* Kwongan – Plant life of the Sandplain. Eds JS Pate and JS Beard, pp 178-204. University of Western Australia Press, Nedlands.

Bell DT, Van der Moezel PG, Delfs JC and Loneragan WA 1987 Northern sandplain Kwongan: Effect of fire on *Hakea obliqua* and *Beaufortia elegans* population structure. J. Roy. Soc. WA, 69(4) 139-142.

Benshemesh J 1990 Management of malleefowl with regard to fire. *In* The Mallee Lands: a conservation perspective. Eds JC Noble, PJ Joss and GK Jones. Chp 32, 206-211. CSIRO Australia.

Benshemesh J 1992 The conservation ecology of malleefowl, with particular regard to fire. PhD Thesis. Monash University, Clayton Victoria.

Benshemesh J 2000 National Recovery Plan for Malleefowl. Department of Environment and Heritage. South Australia.

Bindon P 1997 Aboriginal people and granite domes. *In* Proceed. Granite Outcrops Symposium Sep 14-15, 1996. J. Roy. Soc. WA 80 (3), 173-179.

Boland DJ, Brooker MIH, Chippendale GM, Hall N, Hyland BPK, Johnston RD, Kleinig DA and Turner JD 1992 Forest trees of Australia. Pp 410-411. CSIRO Publishing, Collingwood.

Bradley AJ 1995 Red-tailed Phascogale. *In* The Mammals of Australia. Ed R Strahan. Pp 102-103. Australian Museum. Reed New Holland Publishers, Sydney.

Bradstock RA and Auld TD 1995 Soil temperatures during experimental bushfires in relation to fire intensity: consequences for legume germination and fire management in south-eastern Australia. J. App. Ecol. 34, 76-84.

Bradstock RA and Cohn JS 2002 Fire regimes and biodiversity in semi-arid mallee ecosystems. *In* Flammable Australia - The fire regimes and biodiversity of a continent. Pp 238-258. Eds RA Bradstock, JE Williams and AM Gill. Cambridge University Press, Cambridge.

Brooker L and Lefroy T 2004 Testing approaches to landscape design in cropping lands. Habitat neighbourhoods for conserving viable populations of birds. CSIRO Report commissioned by Land and Water Australia. CSIRO, Wembley, Western Australia.

Brooker L, Atkins L and Ingram J 2001 Enhancing biodiversity values in agricultural lands. Morbinning sub-catchment and surrounds. CSIRO report commissioned by Greening Australia. CSIRO, Western Australia.

Brooker MG 1998 Fire and birds in a Western Australian heathland. Emu 98, 276-287.

Brooker MG and Rowley I 1991 Impact of wildfire on the nesting behaviour of birds in heathland. Wildl. Res. 18, 249-263.

Brooker MIH and Hopper DS 1991 A taxonomic revision of *Eucalyptus wandoo*, *E. redunca* and allied species (Eucalyptus series Levispermae Maiden - Myrtaceae) in Western Australia. Nuytsia 8, 41.

Brooker MIH and Kleinig DA 1996 Eucalypts – An illustrated guide to identification. Reed Books Australia, Melbourne.

Brown K and Brooks K 2002 Bushland weeds. A practical guide to their management. Environmental Weeds Action Network (Inc), Greenwood, Western Australia.

Burbidge AH 2003 Birds and fire in the Mediterranean climate of south-west Western Australia. *In* Fire in ecosystems of south-west Western Australia: impacts and management. Pp 321-347. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Burbidge AH, Blyth J, Danks A, Gillen K and Newbey B 1997 Western ground parrot Interim Recovery Plan 1996-1999. IRP No. 6. Department of Conservation and Land Management, Perth.

Burrows N 2005 Burning rocks. Landscope Special Fire Edition 2, 26-33. Department of Conservation and Land Management, Perth.

Burrows N (in prep.) Evidence-based approaches to fire management in south-west Australian forest landscapes.

Burrows N 2007 Draft Fire Management Guideline No. E5. Granite Outcrops. Department of Environment and Conservation, Perth.

Burrows N and Abbott I 2003 Fire in south-west Western Australia: synthesis of current knowledge, management implications and new research directions. *In* Fire in ecosystems of south-west Western Australia: impacts and management. Pp 437-452. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Burrows ND and Armstrong RA 2003 Managing bushfires in a biodiversity hotspot. Paper presented to the 3rd International Wildland Fire Conference, October 2003.

Burrows N 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: impacts and management. Pp 225-268. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Burrows ND, McCaw WL and Maisey KG 1987 Planning for fire management in Dryandra Forest. *In* Nature Conservation; The Role Of Remnants Of Native Vegetation. Eds DA Saunders, GW Arnold, AA Burbidge and AJM Hopkins. Surrey Beatty and Sons Ltd.

Burrows N, Gardiner G, Ward B and Robinson A 1990 Regeneration of *Eucalyptus wandoo* following fire. Aust. For. 53 (4), 248-258.

Chapman A and Newby KR 1987 A biological survey of the Fitzgerald area, Western Australia. Report to the Australian Heritage Commission, Canberra.

Christensen PES 1980 The biology of *Bettongia pencillata* Gray, 1837, and *Macropus eugenii* (Desmarest, 1817) in relation to fire. Bulletin 91, Forests Department of Western Australia.

Christensen P and Maisey K 1987 The use of fire as a management tool in fauna conservation reserves. *In* Nature Conservation: The role of remnants of native vegetation. Eds DA Saunders, GW Arnold, AA Burbidge and AJM Hopkins. Pp 323-329. Surrey Beatty & Sons, Chipping North.

Christensen P, Liddelow G and Hearn R 2005 Assessment of vertebrate fauna prior to disturbance – timber harvesting and prescribed burning in the forests of Western Australia: The Forest Fauna Distribution Information System. Unpublished report to the Department of Environment and Conservation, Perth. October 2005. Report plus appendices, 522pp.

Christensen PES, Wardell-Johnson G and Kimber P 1985 Birds and fire in southwestern forests. *In* Birds of Eucalypt forests and woodlands: Ecology, Conservation , Management. Eds A Keast, HF Recher, H Ford and D Saunders. Pp 291-299. Royal Australasian Ornithologists Union and Surrey Beatty & Sons.

Coates A 1990 Vegetation survey of the Lake Campion Nature Reserve (No. 24789) and Reserve No. 21759. Department of Conservation and Land Management. Unpublished Report.

Coote M undated Monitoring and evaluating biodiversity conservation projects. Unpublished report, Bushcare Office, Department of Environment and Conservation, Kensington.

Cowan M, Graham G and McKenzie N 2003 Coolgardie 2 (COO2 – Southern Cross subregion). *In* A biodiversity audit of Western Australia's 53 biogeographical sub-regions. Pp 143-155. Department of Conservation and Land Management, Perth.

Daniel G 2003 Draft Wildfire Threat Analysis and fire prevention plan for Crown lands between Coolgardie and Southern Cross. Department of Conservation and Land Management. Perth.

Datson B 2002 Samphires in Western Australia: A field guide to Chenopodiaceae tribe Salicornieae. Department of Conservation and Land Management. Perth.

DEH 2003 Australian Vegetation Attribute Manual: National Vegetation Information System, Version 6.0. Executive steering committee for Australian vegetation information (ESCAVI). Department of the Environment and Heritage, Canberra.

Dell B, Malajczuk N, Grove TS and Thompson G 1990 Ectomycorrhiza formation in Eucalypts. IV Ectomycorrhizas in the sporocarps of the hypogeous fungi *Mesophellia* and *Castorium* in eucalypt forests of Western Australia. New Phytol. 114, 449-456.

Dell J, Harold G, Kitchener DJ, Morris KD and Muir BG 1979 Biological survey of the Western Australian wheatbelt Part 7: Yomaning Nature Reserve, Part 8: Wilroy Nature Reserve, and Part 9: Marchagee Nature Reserve. Records of the Western Australian Museum Supplement No. 8.

Department of Conservation and Land Management 1995 Dryandra Woodland Management Plan. Department of Conservation and Land Management. Management Plan No. 30. Perth.

Department of Conservation and Land Management 2004 Towards a biodiversity conservation strategy for Western Australia. Government of Western Australia.

Dixon K and Barrett R 2003 Defining the role of fire in south-west Western Australian plants. *In* Fire in Ecosystems of south-west Western Australia: impacts and management. Pp 205-223. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Dixon KW, Roche S and Pate JS 1995 The promotive effect of smoke derived from burnt native vegetation on seed germination of Western Australian plants. Oecologia 101, 185-192.

Enright NJ and Lamont BB 1989 Seed banks, fire season, safe sites and seedling recruitment in five co-occurring *Banksia* species. J. Ecol. 77, 1111-1122.

Enright NJ and Lamont BB 1992 Recruitment variability in the resprouting shrub *Banksia attenuata* and non-sprouting congeners in the northern sandplain heaths of southwestern Australia. Acta Oecol. 13, 727-741.

Fire Ecology Working group 2003 Developing an ecological burning strategy – a practitioner's manual. Department of Natural Resources and Environment, Victoria.

Fire Ecology Working Group 2004 Guidelines and procedures for ecological burning on public land in Victoria 2004. Department of Sustainability and Environment, Victoria.

Florabase 2006 Department of Environment and Conservation, Perth.

Ford HA 1989 Ecology of Birds. An Australian Perspective. Pp 74-75. Surrey Beatty & SonsCipping Norton, NSW.

Freudenberger D 2001 Bush for the birds: Biodiversity enhancement guidelines for the Saltshaker Project, Boorowa, NSW. Consultancy report to Greening Australia, ACT & SE NSW, Inc. CSIRO Sustainable Ecosystems, Canberra.

Friend GR 1993 Impact of fire on small vertebrates in mallee woodlands and heathlands of temperate Australia: a review. Biol. Cons. 65, 99-114.

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: impacts and management. Pp 363-380. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Friend JA 1995 Family Myrmecobiidae: Numbat. *In* The Mammals of Australia. Ed R Strahan. Pp 159-162. Australian Museum. Reed New Holland Publishers, Sydney.

Gardner CA 1979 Eucalypts of Western Australia. Bulletin 4013. Ed TEH Aplin. Western Australian Department of Agriculture, Perth.

Garnett ST and Crowley GM 2000 The action plan for Australian birds. Environment Australia, Canberra.

George AS 2000 Cryptogams on granites. *In* Proceed. Management of Granite Outcrops Symposium. Hyden, April 16-18 1999. J. Roy. Soc. WA, 83 (3), 127-129.

Gibson N, Keighery GJ, Lyons MN and Webb A 2004 Terrestrial flora and vegetation of the Western Australian wheatbelt. *In* A biodiversity survey of the Western Australian agricultural zone. Eds GJ Keighery, SA Halse, NL McKenzie and MS Harvey. Pp 139-189. Records of the Western Australian Museum Supplement No. 67.

Gill AM and Nicholls AO 1989 Monitoring fire-prone flora in reserves for nature conservation. *In* Fire management on nature conservation lands. Eds N Burrows, L McCaw and G Friend. Pp 137-151. WA Department of Conservation and Land Management, Perth.

Grealish G and Wagnon J 1995 Land resources of the Bencubbin area. Land Resources Series No, 12. Agriculture Western Australia.

Hallam SJ 1975 Fire and hearth: a study of aboriginal usage and European usurpation in south-western Australia. Australian Institute of Aboriginal Studies, Canberra

Harold G and Dennings S 1998 The first five years. Malleefowl Preservation Group, Ongerup, WA. Pp97.

Hill KD 1990 Biogeography of the mallee eucalypts. *In* The mallee lands: a conservation perspective Eds JC Noble, PJ Ross and GK Jones, pp 16-20. CSIRO Melbourne.

Hilton RN, Malajczuk N and Pearce MH 1989 Larger fungi of the Jarrah forest: an ecological and taxonomic survey. *In* The Jarrah Forest: A complex Mediterranean ecosystem. Eds B Dell, JJ Havel and N Malajczuk. Pp 89-109. Kluwer, Dordrecht.

Hobbs RJ 2002 Fire regimes and their effects in Australian temperate woodlands. *In* Flammable Australia - The fire regimes and biodiversity of a continent. Eds RA Bradstock, JE Williams and AM Gill. Chp 13 pp 305-326. Cambridge University Press, Cambridge.

Hobbs RJ and Atkins L 1991 Interactions between annuals and woody perennials in a Western Australian wheatbelt reserve. J. Veg. Sc. 2, 643-654.

Hobbs RJ, Arnold GW and Leone J 1992 Weed control and rehabilitation of woodland in the Western Australian wheatbelt. Unpublished report to World Wildlife Fund. CSIRO, Division of Wildlife and Ecology, Midland, W.A.

Hodgkinson KC and Griffin GF 1982 Adaption of shrub species to fires in the arid zone. *In* Evolution of the flora and fauna of arid Australia. Eds WR Barker and PJM Greenslade. Paper 15, 145-151. Peacock publications, Frewville, South Australia.

Hopkins AJM 1985 Fire in the woodlands and associated formations of the semi-arid region of south-western Australia. *In* Fire Ecology and Management in Western Australian Ecosystems. Ed J Ford. Proc. Symposium May 1985. WAIT Environmental Studies group Report 14, 83-90.

Hopper SD 2000 Creation of conservation reserves and managing fire on granite outcrops – a case study of Chiddarcooping nature reserve in the Western Australian wheatbelt. J. Roy. Soc. W.A. 83, 173-186.

Hopper SD 2003 An evolutionary perspective on south-west Western Australian landscapes, biodiversity and fire: a review and management implications. *In* Fire in ecosystems of south-west Western Australia: impacts and management. Pp 9-35. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Hussey BMJ 1999 How to manage your wandoo woodlands. Land for Wildlife. Department of Conservation and Land Management, Perth.

Hussey BMJ and Wallace KJ 2003 Managing your bushland. Department of Conservation and Land Management, Western Australia.

Hussey P and Baxter A 2006 The use of fire in small remnants. Land for Wildlife Notes No. 17. Department of Conservation and Land Management, Western Australia.

Huston M 2003 Understanding the effects of fire and other mortality-causing disturbances on species diversity. *In* Fire in ecosystems of south-west Western Australia: impacts and management. Pp 37-70. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Jenkins S 1996 Native vegetation on farms survey 1996. Resource management technical report 164. Agriculture Western Australia.

Johnson CN 1997 Fire and habitat management for a mycophagous marsupial: the Tasmanian bettong *Bettongia gaimardi*. Aust. J. Ecol. 22, 101-105.

Johnson EA and Van Wagner CE 1984 The theory and use of two fire history models. Can. J. For. Res. 15, 214-219.

Keighery BJ 1994 Bushland plant survey. A guide to plant community survey for the community. Wildflower Society of Western Australia (Inc.), Nedlands.

Keighery GJ 2004 State salinity strategy biological survey of the Western Australian wheatbelt: background. *In* A biodiversity survey of the Western Australian agricultural zone. Eds GJ Keighery, SA Halse, NL McKenzie and MS Harvey. Pp 1-6. Records of the Western Australian Museum Supplement No. 67.

Keighery GJ, Halse SA, McKenzie NL and Harvey MS 2004 Eds A biodiversity survey of the Western Australian agricultural zone. Records of the Western Australian Museum Supplement No. 67.

Keith DA 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 DA, Williams JE and Woinarski JCZ 2002 Fire management and biodiversity conservation: key approaches and principles. *In* Flammable Australia - The fire regimes and biodiversity of a continent. Eds RA Bradstock, JE Williams and AM Gill. Chp 17 pp 410-425. Cambridge University Press, Cambridge.

Laing IAF and Hauck EJ 1997 Water harvesting from granite outcrops in Western Australia. *In* Proceedings Granite Outcrops Symposium Sep 14-15, 1996. J. Roy. Soc. WA 80 (3), 181-184.

Lambeck RJ 1997 Focal species: A multi-species umbrella for nature conservation. Conserv. Biol. 11, 849-856.

Lambeck RJ 1998 Nature conservation at the landscape scale – adequacy of habitat. *In* Dongolocking pilot planning project for remnant vegetation. Phase 1 Final Report. Ed KJ Wallace. Pp 59-74. Department of Conservation and Land Management, Perth.

Lambeck RJ 1999 Landscape planning for biodiversity conservation in agricultural regions. A case study from the wheatbelt of Western Australia. CSIRO Wildlife and Ecology. Biodiversity Technical Paper No. 2. Department of Environment and Heritage, Canberra.

Lamont BB 1991 Canopy seed storage and release – what's in a name. Oikos 60, 266-268.

Lamont BB and Barker MJ 1988 Seed bank dynamics of a serotinous fire-sensitive *Banksia* species. Aust. J. Bot. 36, 193-203.

Leigh JH and Holgate MD 1979 The responses of the understorey of forests and woodlands of the Southern Tablelands to grazing and burning. Aust. J. Ecol. 4, 25-45.

Leonard SWJ and Kirkpatrick JB 2004 Effects of grazing management and environmental factors on native grassland and grassy woodland, Northern Midlands, Tasmania. Aust. J. Bot. 52, 529-542.

Long JL 1990 The breeding biology of four species of parrots in the south of Western Australia. Agriculture Protection Board Technical Series 6, Perth.

Lyons MN, Gibson N, Keighery GJ and Lyons SD 2004 Wetland flora and vegetation of the Western Australian wheatbelt. *In* A biodiversity survey of the Western Australian agricultural zone. Eds GJ Keighery, SA Halse, NL McKenzie and MS Harvey. Pp 39-89. Records of the Western Australian Museum Supplement No. 67.

Main BY 1995 Survival of trapdoor spiders during and after fire. CALMScience Supplement 4, 214-215.

Malajczuk N and Hingston FJ 1981 Ectomycorrhizae associated with Jarrah. Aust. J. Bot. 29, 453-462.

Maslin B 2001 Coordinator WATTLE Acacias of Australia CD and User Guide. ABRS and CALM. CSIRO Publishing, Collingwood.

Mattiske EM and Havel JJ 1998 Regional Forest Agreement, vegetation complexes – Perth, Pinjarra, Collie, Busselton/Augusta, Pemberton, Mt Barker – Western Australia. 1:250,000 maps. Department of Conservation and Land Management, Western Australia and Department of Environment and Heritage, Canberra.

Mattiske EM and Havel JJ 2004 Delineation of landscape conservation units in southwest region of Western Australia. Report prepared for the department of Conservation and Land Management, Western Australia.

McArthur WM 1992 Land resources of the Kellerberrin region. Technical Report 134. Department of Agriculture, Western Australia, South Perth.

McCarthy MA, Gill AM and Bradstock RA 2001 Theoretical fire-interval distributions. Internat. J. Wildl. Fire 10, 73-77.

McCaw WL 1997 Predicting fire spread in Western Australian mallee-heath shrubland. PhD Thesis, University of New South Wales.

McCaw L 1998 Research as a basis for fire management in mallee-heath shrublands of south-western Australia. Third International Conference on Forest Fire Research / 14th Conference on Fire and Forest Meteorology. Ed DX Viegas. 16-20th Nov., 1998. Coimbra, Portugal. Pp 2335-2348.

Coote M (undated) Monitoring and evaluating biodiversity conservation projects. Unpublished report. Department of Conservation and Land Management, Perth.

McCaw WL and Smith RH 1992 Seed release from *Banksia baxteri* and *Hakea crassifolia* following scrub-rolling and burning. J. Roy. Soc. WA 75, 47-50.

McCluskey P, Davies M, Beecham B and Bone B 2003 Draft fire management strategy Katanning district eastern reserves 2003-2013. Department of Conservation and Land Management.

McMahonARG, Carr GW, Bedggood SE, Hill RJ and Pritchard AM 1994 Prescribed fire and control of coast wattle (*Acacia sophorae* (Labill.) RBr) invasion in coastal heath south-west Victoria. Biodiversity Series, No. 8. Proceedings of Conference held 8-9th October, 1994. Footscray, Melbourne. Environment Australia.

Meredith C 1984 Fire and Birds: the results of two studies and their relevance to fuel reduction burning. *In* Fighting Fire with Fire. Ed EHM Ealey. Graduate School Environmental Science, Monash University, Melbourne. Pp 193-202.

Milberg P and Lamont BB 1997 Seed/cotyledon size and nutrient content play a major role in early performance of species on nutrient-poor soils. New Phytologist 137, 665-672.

Mollemans F 1993 A simplified key for assessing the ecological significance of on-farm bush remnants in the wheatbelt. Miscellaneous publication 20/93, Department of Agriculture, Western Australia.

Molnar CD, Fletcher D and Parsons RF 1989 Relationships between heath and *Leptospermum laevigatum* scrub at Sandringham, Victoria. Proceedings of the Royal Society of Victoria 101, 78-87.

Moore J and Wheeler J 2002 Southern weeds and their control. Bulletin No. 4558/02. Department of Agriculture of Western Australia, South Perth.

Morgan JW 1998 Importance of canopy gaps for recruitment of some forbs in *Themeda triandra* dominated grasslands of south-eastern Australia. Aust. J. Bot. 46, 609-627.

Mueller C 2001 Review of fire operations in forest regions managed by the Department of Conservation and Land Management. Department of Conservation and Land Management, Perth.

Mueller C (in prep.) Wildfire Threat Analysis of the Avon Basin.

Muir BG 1978 Some nature reserves of the Wongan-Ballidu shire. Western Australian Department of Fisheries and Wildlife. Unpublished report.

Muir BG 1985 Fire exclusion: A baseline for change? *In* Fire Ecology and Management in Western Australian Ecosystems. Ed J Ford. Proc. Symposium May 1985. WAIT Environmental Studies group Report 14, 119-128.

Muir BG, Chapman A, Dell J and Kitchener DJ 1978 Biological survey of the Western Australian wheatbelt Part 6: Durokoppin and Kodj Kodjin Nature Reserves. Records of the Western Australian Museum Supplement No. 7.

Myers N, Mittermeier RA, Mittermeier CG, da Fonseca GAB and Kent J 2000 Biodiversity hotspots for conservation priorities. Nature 403, 853-858.

Newby B 1999 Birds on Farms Project in Western Australia. Western Australian Bird Notes Supplement No. 5. Birds Australia, Floreat, Western Australia.

NLWRA 2001 Australian native vegetation assessment 2001. Compiled by M Cofinas and C. Creighton. National Land and Water Resources Audit, Land and Water Australia, Canberra.

Noble IR 1982 The significance of fire in the biology and evolutionary ecology of mallee *Eucalyptus* populations. *In* Evolution of the flora and fauna of arid Australia. Eds WR Barker and PJM Greenslade. Paper 16, 153-159. Peacock publications, Frewville, South Australia.

Noble JC 1989 Fire regimes and their influence on herbage and mallee coppice dynamics. *In* Mediterranean landscapes in Australia – mallee ecosystems and their management. Eds JC Noble and RA Bradstock. Chp 11, 168-180. CSIRO, Australia.

Noble IR and Slatyer RO 1981 Concepts and models of succession in vascular plant communities subject to recurrent fire, *In* Fire and the Australian Biota. Eds AM Gill, RH Groves and IR Noble. Pp 311-335. Australian Academy of Science, Canberra.

Pate JS and Dixon KW 1982 Tuberous, Cormous and Bulbous Plants. University of Western Australia Press, Nedlands WA.

Pigott JP and Sage LW 1997 Remnant vegetation, priority flora and wed invasions at Yilliminning Rock, Narrogin, Western Australia. *In* Proceedings Granite Outcrops Symposium. Sep 14-15, 1996. J. Roy. Soc. WA 80 (3), 201-208.

Pinder AM, Halse SA, Shiel RJ and McRae JM 2000 Granite outcrop pools in south-western Australia: foci of diversification and refugia for aquatic invertebrates. *In* Proceedings Management of Granite Outcrops Symposium. Hyden, April 16-18 1999. J. Roy. Soc. WA, 83 (3), 149-156.

Podger FD, James SH and Mulcahy MJ 1996 Review of dieback in Western Australia. Report to the Western Australian Minister for the environment Vol. 1 – Report and Recommendations. Western Australian dieback review panel.

Possingham HP 2001 The business of biodiversity: Applying decision theory principles to nature conservation. The Ecology Centre, Issue 9. University of Queensland.

Recher HF, Allen D and Gowing G 1985 The impact of wildfire on birds in an intensively logged forest. *In* Birds of Eucalypt forests and woodlands: Ecology, Conservation , Management. Eds A Keast, HF Recher, H Ford and D Saunders. Pp 283-290. Royal Australasian Ornithologists Union and Surrey Beatty & Sons.

Richardson DM, Van Wilgen BW, Le Maitre DC, Higgins KB and Forsyth GG 1994 A computer-based system for fire management in the mountains of the Cape Province, South Africa. Intern. J. Wildl. Fire 4(1), 17-32.

Rippey ME and Hobbs RJ 2003 The effects of fire and quokkas (*Setonix brachyurus*) on the vegetation of Rottnest Island, Western Australia. J. Roy. Soc. West. Australia 86, 49-60.

Robinson RM and Bougher NL 2003 The response 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. Pp 269-289. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Rose R, Wiltshire G and Lang S 1999 The importance and application of spatial patterns in the management of fire regimes for the protection of life and property and the conservation of biodiversity. Australian Bushfire Conference, July 1999, Albury.

Runicman and Malcolm 1989 Forage shrubs and grasses for revegetating saltland. Bulletin 4153. Western Australian Department of Agriculture, Perth.

Saftrom R 1995 Conservation values of small reserves in the central wheatbelt of Western Australia: A framework for evaluating the conservation values of small reserves. Unpublished Report prepared for the Department of Conservation and Land Management, Western Australia. Environs Consulting Pty Ltd.

Sage LW, Blankendaal PA, Moylett A and Agar K 2004 The occurrence and impact of *Phytophthora cinnamomi* in the central-western Avon Wheatbelt bioregion of Western Australia. J. Roy. Soc. WA 87, 15-18.

Saunders DA 1977 The effect of agricultural clearing on the breeding success of the white-tailed black cockatoo. Emu 77, 180-184.

Saunders DA 1979 The availability of tree hollows for use as nest sites by white-tailed black cockatoos. Aust. Wildl. Res. 6, 205-216.

Saunders DA and Ingram JA 1995 Birds of south-western Australia. Surrey Beatty & Sons, Sydney.

Saunders DA, Smith GT, Ingram JA and Forrester RI 2002 Changes in a remnant of salmon gum *Eucalyptus salmonophloia* and York gum *E. loxophleba* woodland, 1978 to 1997. Implications for woodland conservation in the wheat-sheep regions of Australia. Biol. Cons. 110(2), 245-256.

SEQ Fire and Biodiversity Consortium 2002 Fire and biodiversity monitoring manual. A guide for local government field officers and land managers. South East Queensland Fire and Biodiversity Consortium, Brisbane.

Shearer BL and Tippett JT 1989 Jarrah dieback: the dynamics and management of *Phytophthora cinnamomi* in the jarrah (*Eucalyptus marginata*) forest of south-western Australia. Research Bulletin 3. Department of Conservation and Land Management, Perth.

Shearer BL, Crane CE and Cochrane A 2004 Quantification of the susceptibility of the native flora of the south-west botanical province, Western Australia, to *Phytophthora cinnamomi*. Aust. J. Bot. 52, 435-443.

Shepherd DP, Beeston GR and Hopkins AJM 2002 Native Vegetation in Western Australia. Technical Report 249. Department of Agriculture, Western Australia, South Perth.

Smith GT 1995 Species richness, habitat and conservation of scorpions in the Western Australian wheatbelt. Records of the Western Australian Museum Supplement 52, 55-66.

Stoneman GL, Rayner ME and Bradshaw FJ 1997 Size and age parameters of nest trees used by four species of parrot and one species of cockatoo in south-west Australia: Critique. Emu 97, 94-96.

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, May 1999. Fire Ecology Working Group. Pp 30-38. Natural Resources and Environment, Victoria.

Tolhurst KG and Friend GR 2003 An objective basis for ecological fire management in Victoria. *In* Variability in fire interval. How important is it and how do we manage for it? Eds TV Walshe and JE Williams. Proceedings of workshop, 26th September 2003, Albany Western Australia. University of Western Australia, Crawley, WA.

Trémont RM and McIntyre S 1994 Natural grassy vegetation and native forbs in temperate Australia: structure, dynamics and life histories. Aust. J. Bot. 42, 641-658.

Van Heurck P and Abbott I 2003 Fire and terrestrial invertebrates in south-west Western Australia. *In* Fire in ecosystems of south-west Western Australia: impacts and management. Pp 291-319. Eds I Abbott and N Burrows. Backhuys Publishers, Leiden, The Netherlands.

Van Wilgen BW, Richardson DM and Seydack AHW 1994 Managing fynbos for biodiversity: Constraints and options in a fire-prone environment. S. Afr. J. Science 90, 322-329.

Wardell-Johnson G, Burrows N and Shu L 2006 Emergent patterns in the mosaic of patch burning vary with the fire environment and landscape context in south-western Australia. *In* Proceedings - Life in a fire-prone environment: Translating science into practice. Paper 42, Bushfire Conference 2006, Brisbane 6-9th June 2006.

Watson P 2001 The role and use of fire for biodiversity conservation in south-east Queensland: Fire management guidelines derived from ecological research. SEQ Fire and Biodiversity Consortium, Brisbane.

Watson J, Freudenberger D and Paull D 2003 An assessment of the focal-species approach for conserving birds in variegated landscapes in southeastern Australia. Conserv. Biol. 15(5), 1364-1373.

Wellington AB and Noble IR 1985 Post-fire recruitment and mortality in a population of the mallee *Eucalyptus incrassata* in semi-arid, southeastern Australia. J. Ecol. 73, 645-656.

Weston AS 1985 Fire – and persistence of the flora on Middle Island, a south-western Australian offshore island. *In* Fire ecology and management in Western Australian ecosystems. Ed J Ford. WAIT Environmental Studies Group Report 14, 111-118. WAIT Perth.

Whelan RJ, Rodgerson L, Dickman CR and Sutherlands EF 2002 Critical life cycles of plants and animals: developing a process-based understanding of population changes in fire-prone landscapes. *In* Flammable Australia - The fire regimes and biodiversity of a continent. Eds RA Bradstock, JE Williams and AM Gill. Chp. 17 pp 94-124. Cambridge University Press, Cambridge.

Williams JE, Whelan RJ and Gill AM 1994 Fire and environmental heterogeneity in southern temperate forest ecosystems: Implications for management. Aust. J. Bot. 42, 125-137.

Wills RT 1993 The ecological impact of *Phytophthora cinnamomi* in the Stirling Range National Park, Western Australia. Aust. J. Ecol. 18, 145-159.

Withers PC and Edward DH 1997 Terrestrial fauna of granite outcrops in Western Australia. *In* Proceed. Granite Outcrops Symposium. Sep 14-15, 1996. J. Roy. Soc. WA 80 (3), 159-166.

Woodman Environmental Consulting 2002 Environmental impact assessment of fuel reduction buffers on the nature conservation values in nature reserves and crown land for fire management. Unpublished Report to the Department of Conservation and Land Management, Western Australia.

Yates CJ and Broadhurst LM 2002 Assessing limitations on population growth in two critically endangered *Acacia* taxa. Biol. Conserv. 108, 13-26.

Yates CJ and Hobbs RJ 1997 Temperate woodlands: a review of their status, processes threatening their persistence and techniques for restoration. Aust. J. Bot. 45, 949-973.

Yates CJ and Ladd PG 2004 Relative importance of reproductive biology and establishment ecology for persistence of a rare shrub in a fragmented landscape. Cons. Biol. 19(1), 239-249.

Yates CJ, Hobbs RJ and Bell RW 1994 Landscape-scale disturbances and regeneration in semi-arid woodlands of south-western Australia. Pacific Cons. Biol. 1, 214-221.

Yates CJ, Norton DA and Hobbs RJ 2000 Grazing effects on plant cover, soil and microclimate in fragmented woodlands in south-western Australia: implications for restoration. Austral Ecol. 25, 36-47.

Yates CJ, Hopper SD, Brown A and van Leeuwin S 2003 (a) Impact of two wildfires on endemic granite outcrop vegetation in Western Australia. J. Veg. Sc. 14, 185-194.

Yates CJ, Abbott I, Hopper SD and Coates DJ 2003 (b) Fire as a determinant of rarity in the south-west Western Australian global biodiversity hotspot. *In* Fire in ecosystems of south-west Western Australia: Impacts and Management. Eds I Abbott and N Burrows. Pp 395-420. Backhuys Publishers, Leiden, The Netherlands.

PART THREE

APPENDIX I

VEGETATION ASSOCIATIONS IN THE AVON BASIN

Table 1. A list of all Beard-Hopkins pre-European vegetation associations present in the Avon Basin. Only those associations found in the AW1, AW2, MAL2, and COO2 IBRA sub-regions and within the Avon NRM boundary are included in these Fire Management Guidelines. (See Part I, Section 7.2)

Vegetation Association	Beard-Hopkins Description
4	Medium woodland; marri & wandoo
5	Medium woodland; wandoo & powderbark (*Eucalytpus accedens*)
7	Medium woodland; York gum (*Eucalyptus loxophleba*) & wandoo
8	Medium woodland; salmon gum & gimlet
13	Medium open woodland; wandoo
18	Low woodland; mulga (*Acacia aneura*)
19	Low woodland; mulga between sandridges
25	Low woodland; *Allocasuarina huegeliana* & York gum
36	Shrublands; thicket, *Acacia-Allocasuarina* alliance
37	Shrublands; teatree thicket
39	Shrublands; mulga scrub
40	Shrublands; *Acacia* scrub
41	Shrublands; teatree scrub
47	Shrublands; tallerack mallee-heath
49	Shrublands; mixed heath
59	Grasslands
125	Bare areas; salt lakes
128	Bare areas; rock outcrops
129	Bare areas; drift sand
131	Mosaic: Medium woodland; salmon gum & gimlet / Shrublands; mallee scrub, redwood (*E.transcontinentalis*) and black marlock

141	Medium woodland; York gum, salmon gum & gimlet
142	Medium woodland; York gum & salmon gum
144	Medium woodland; wandoo, salmon gum, morrel, gimlet and rough fruited mallee
145	Mosaic: Medium woodland; York gum & salmon gum / Shrublands; thicket, *Acacia-Allocasuarina-Melaleuca* alliance
147	Succulent steppe with scrub; *Acacia* species over saltbush
148	Medium woodland; gimlet
202	Shrublands; mulga & *Acacia quadrimarginea* scrub
214	Mosaic: Medium woodland; goldfield eucalypts / Succulent steppe with open low woodland; *Myoporum* over saltbush
221	Succulent steppe; saltbush
256	Low woodland; York gum
314	Succulent steppe with open woodland; York gum over saltbush
325	Succulent steppe; saltbush & samphire
352	Medium woodland; York gum
356	Succulent steppe with open woodland; eucalypts over saltbush
357	Medium woodland over scrub; York gum over bowgada & jam (*Acacia acuminata*)
380	Shrublands; scrub-heath on sandplain
392	Shrublands; *Melaleuca thyioides* thicket
413	Shrublands; *Acacia neurophylla* & *Acacia* species thicket
420	Shrublands; bowgada & jam scrub
435	Shrublands; *Acacia neurophylla*, *A. beauverdiana* & *A. resinomarginea* thicket
436	Shrublands; mixed *Acacia* thickets in thickets of *Acacia-Allocasuarina-Melaleuca* alliance
437	Shrublands; mixed *Acacia* thicket on sandplain
468	Medium woodland; salmon gum & goldfields blackbutt
483	Hummock grasslands
486	Mosaic: Medium woodland; salmon gum & red mallee / Shrublands; mallee scrub *Eucalyptus eremophila*
491	Medium woodland; morrel & Dundas blackbutt (*E. dundasii*)
501	Medium woodland; goldfields blackbutt
511	Medium woodland; salmon gum & morrel
516	Shrublands; mallee scrub, black marlock
519	Shrublands; mallee scrub (*Eucalyptus eremophila*)
520	Shrublands; *Acacia quadrimarginea* thicket
522	Medium woodland; redwood (*Eucalyptus transcontinentalis*) & merrit (*E. floctoniae*)

535	Medium woodland; rough fruited mallee on greenstone hills
536	Medium woodland; morrel & rough fruited mallee *(Eucalyptus corrugata*)
537	Medium woodland; morrel (*Eucalyptus longicornis*)
538	Shrublands; *Acacia brachystachya* scrub
551	Shrublands; *Allocasuarina campestris* thicket
552	Shrublands; *Allocasuarina acutivalvus* & *Calothamnus* (also *Melaleuca*) thicket on greenstone hills
555	Hummock grasslands, mallee steppe, red mallee over spinifex, *Triodia scariosa*
631	Succulent steppe with woodland and thicket; York gum over *Melaleuca thyoides* & samphire
676	Succulent steppe; samphire
694	Shrublands; scrub-heath on yellow sandplain *Banksia-Xylomelum* alliance in the Geraldton Sandplain & Avon Wheatbelt Regions
929	Low forest; moort (*Eucalytpus platypus*)
931	Medium woodland; yate
934	Shrublands; mallee scrub (*Eucalyptus nutans*)
936	Medium woodland; salmon gum
941	Mosaic: Medium woodland; salmon gum & morrel / Shrublands; mallee scrub, redwood
942	Mosaic: Medium woodland; yate / Shrublands; mallee scrub
945	Mosaic: Medium woodland; salmon gum / Shrublands; mallee scrub, redwood & black marlock
946	Medium woodland; wandoo
947	Medium woodland; powderbark & mallet
948	Medium woodland; York gum & river gum
949	Low woodland; *Banksia*
950	Medium woodland; *Casuarina obesa*
951	Succulent steppe with sparse woodland & thicket; York gum & Kondinin blackbutt over teatree thicket & samphire
952	Shrublands; *Dryandra* heath
953	Succulent steppe with thicket; teatree over samphire (m5)
954	Shrublands; thicket, jam and *Allocasuarina huegeliana*
955	Mosaic: Shrublands; scrub-heath (South-East Avon) / Shrublands; *Allocasuarina campestris* thicket
956	Shrublands; *Allocasuarina campestris* thicket with scattered wandoo
959	Succulent steppe with sparse woodland & thicket; yorrell & Kondinin blackbutt over teatree & samphire
960	Shrublands; mallee scrub, redwood (*E. transcontinentalis*) and black marlock
961	Mosaic: Shrublands; scrub-heath (South East Avon)/ Shrublands; *Allocasuarina acutivalvis* thicket
962	Medium woodland; mallet (*Eucalytpus astringens*)

966	Succulent steppe with sparse woodland & thicket; salmon gum & morrel over teatree & samphire
988	Succulent steppe with thicket; *Melaleuca thyoides* over samphire
1005	Low woodland; *Allocasuarina huegeliana*
1023	Medium woodland; York gum, wandoo & salmon gum
1024	Shrublands; mallee & *Allocasuarina* thicket
1025	Mosaic: Medium woodland; York gum, salmon gum & morrel / Succulent steppe; saltbush & samphire
1041	Low woodland; *Allocasuarina huegeliana* & jam
1048	Mosaic: Shrublands; *Melaleuca* patchy scrub / Succulent steppe; samphire
1049	Medium woodland; wandoo, York gum, salmon gum, morrel and gimlet
1053	Shrublands; *Melaleuca uncinata* thicket with scattered York gum
1055	Shrublands; York gum & *Eucalyptus sheathiana* mallee scrub
1056	Shrublands; thicket, *Acacia* and *Allocasuarina campestris*
1057	Mosaic: Shrublands; Medium woodland; salmon gum & gimlet / York gum & *Eucalyptus sheathiana* mallee scrub
1058	Shrublands; York gum & *Eucalyptus gonglocarpa* mallee scrub
1059	Mosaic: Medium woodland; salmon gum & gimlet / Shrublands; mallee *Eucalyptus longicornis* & *E. sheathiana* scrub
1061	Mosaic: Medium sparse woodland; salmon gum & yorrell / Succulent steppe; saltbush & samphire
1062	Succulent steppe with open woodland & thicket; York gum over *Melaleuca thyoides* & samphire
1063	Medium low woodland; York gum & cypress pine (*Callitris columellaris*)
1065	Mosaic: Shrublands;Medium woodland; wandoo & gimlet / York gum & *Eucalyptus sheathiana* mallee scrub
1067	Medium woodland; salmon gum, morrel, gimlet and rough fruited mallee
1068	Medium woodland; salmon gum, morrel, gimlet and *E. sheathiana*
1071	Succulent steppe with scrub; acacia species over saltbush & bluebush
1075	Shrublands; mallee scrub, *E. eremophila*, and black marlock
1076	Mosaic: Medium woodland; salmon gum & morrel/ Shrublands; mallee scrub, *E. eremophila* & bloodwood (*E. dichromophloia*)
1078	Medium woodland; salmon gum, redwood, merrit, gimlet and *E. sheathiana*
1079	Mosaic: Medium open woodland; salmon gum & morrel / Succulent steppe; saltbush
1080	Succulent steppe with mallee & thickets; Mallee and *Melaleuca uncinata* thickets on salt flats
1081	Shrublands; mallee scrub, *E. longicornis* and *E. sheathiana*
1094	Mosaic: Medium woodland; York gum & salmon gum / Shrublands; mallee scrub *Eucalyptus eremophila* & black marlock
1098	Mosaic: Medium sparse woodland; salmon gum & morrel / Succulent steppe; samphire
1147	Shrublands; scrub-heath in the South-East Avon Wheatbelt Region
1148	Shrublands; scrub-heath in the Coolgardie Region

1200	Mosaic: Medium woodland; salmon gum & morrel / Shrublands; mallee scrub *E. eremophila* and black marlock
1271	Bare areas; claypans
1413	Shrublands; *Acacia, Allocasuarina* & *Melaleuca* thicket
2047	Shrublands; tamma & *Dryandra* thicket
2048	Shrublands; scrub-heath in the Mallee Region
3041	Mosaic: Low woodland; *Allocasuarina huegeliana* & jam around granite rocks

APPENDIX II

VEGETATION CLASSES IN THE AVON BASIN

Table 1. List of all vegetation classes and associations in the Avon Basin, including minor associations that have been amalgamated into the major associations for these Fire Management Guidelines. A total of 56 minor vegetation associations, each having less than 1,000ha of remaining vegetation in the Avon Basin, were amalgamated into major vegetation associations.

Vegetation Class	Major Vegetation Associations	Minor Amalgamated Vegetation Associations	Total Pre-European Area (ha)	Total Remnant Area (ha)	% Remaining
Low woodland	25	256, 1005, 1041, 3041	16,260	3,209	19.7
Low woodland Total			16,260	3,209	
Mallee shrubland	47		9,364	7,163	76.5
Mallee shrubland	519		1,213,792	466,463	38.4
Mallee shrubland	960	934	220,703	23,133	10.5
Mallee shrubland	1055	1058	136,170	14,037	10.3
Mallee shrubland	1063		171,323	154,859	90.4
Mallee shrubland	1075		174,470	29,606	17.0
Mallee shrubland	1081		15,148	2,266	15.0
Mallee shrubland Total			1,940,969	697,527	
Mosaic	131		171,466	8,932	5.2
Mosaic	941		34,257	14,536	42.4
Mosaic	945	486, 942, 1076, 1094	176,850	22,428	12.7
Mosaic	955	2047	132,023	10,359	7.8
Mosaic	961		27,389	4,276	15.6
Mosaic	1048		13,815	2,373	17.2
Mosaic	1057		145,312	13,588	9.4
Mosaic	1061	214	58,462	12,567	21.5
Mosaic	1079		10,119	3,881	38.4
Mosaic	1098	966	20,756	3,349	16.1
Mosaic	1200		102,557	8,102	7.9

Mosaic Total			893,006	104,391	
Rock Outcrops	128	129	136,209	42,924	31.5
Rock Outcrops Total			136,209	42,924	
Salt lakes	125	1271	257,532	60,258	23.4
Salt lakes Total			257,532	60,258	
Shrub heath	380		32,541	13,672	42.0
Shrub heath	694	949	154,113	5,285	3.4
Shrub heath	1147	49, 952	41,748	2,740	6.6
Shrub heath	1148		195,575	124,900	63.9
Shrub heath	2048		305,701	144,311	47.2
Shrub heath Total			729,678	290,908	
Shrubland	36		300,249	65,615	21.9
Shrubland	37		6,371	1,785	28.0
Shrubland	41		13,772	4,988	36.2
Shrubland	435	413	529,999	75,004	14.2
Shrubland	437	18, 19, 39, 59, 420, 436, 483, 555	139,429	32,311	23.2
Shrubland	520	202	26,881	3,156	11.7
Shrubland	538	40	128,660	21,135	16.4
Shrubland	551		163,805	44,453	27.1
Shrubland	552	516	13,116	12,444	94.9
Shrubland	954		6,502	1,043	16.0
Shrubland	956		25,556	2,744	10.7
Shrubland	1024		593,089	58,312	9.8
Shrubland	1053		12,706	1,722	13.6
Shrubland	1056		21,073	3,098	14.7
Shrubland	1413		1,219,852	471,468	38.6
Shrubland Total			3,201,061	799,279	
Succulent steppe	147	1071	36,248	12,616	34.8
Succulent steppe	221		3,144	3,144	100.0
Succulent steppe	314		6,391	6,394	100.1
Succulent steppe	325		7,920	7,464	94.2
Succulent steppe	631		11,812	3,916	33.2
Succulent steppe	951		27,508	8,441	30.7
Succulent steppe	953		9,457	1,432	15.1

Succulent steppe	959		13,093	4,003	30.6
Succulent steppe	988	392, 950, 1080	54,126	3,671	6.8
Succulent steppe	1062	356, 676	29,867	6,488	21.7
Succulent steppe Total			199,565	57,570	
Woodland	5	13	9,963	5,894	59.2
Woodland	8		450,013	69,527	15.4
Woodland	141	357	996,570	127,968	12.8
Woodland	142		205,646	35,488	17.3
Woodland	352	145, 948	342,884	19,049	5.6
Woodland	511	501, 929	689,254	303,161	44.0
Woodland	522	1078	124,314	12,143	9.8
Woodland	535		24,348	20,780	85.3
Woodland	536	491, 537	13,944	6,318	45.3
Woodland	936	468, 931	47,737	21,418	44.9
Woodland	946	4	47,932	7,019	14.6
Woodland	947	962	12,779	2,820	22.1
Woodland	1023	7	846,928	38,662	4.6
Woodland	1049	1025, 1065	836,186	30,559	3.7
Woodland	1067	144, 148	19,582	12,491	63.8
Woodland	1068	1059	271,250	134,091	49.4
Woodland Total			4,939,330	847,387	
Total			**12,313,611**	**2,903,451**	**23.6**

APPENDIX III

VEGETATION ASSOCIATIONS IN THE IBRA SUB-REGIONS OF THE AVON BASIN

Table 2. List of all vegetation associations in the Avon Wheatbelt 1 IBRA sub-region, showing pre-European and remnant vegetation areas, and proportion remaining. Vegetation Associations with less than 10% remaining in the AW1 are shown in light grey, and those with less than 500ha remaining in the AW1 are shown dark grey.

Vegetation Association AW1	Pre-European Area (hectares)	Remnant Area (hectares)	% remaining
7	172	0	0.06
8	353,795	33,712	9.53
36	299,309	64,746	21.63
37	317	9	2.89
125	61,538	17,487	28.42
128	32,931	14,255	43.29
131	69,590	3,278	4.71
141	173,319	10,073	5.81
142	143,320	33,406	23.31
145	7,944	322	4.06
147	4,210	3,500	83.15
221	2,966	2,968	100.07
314	1,170	1,171	100.07
325	7,914	7,464	94.31
352	2,587	151	5.84
356	3,319	958	28.88
357	1	1	100.07
392	185	24	12.80
413	375	78	20.81
435	7,253	1,552	21.40
511	95,821	11,463	11.96
519	12,031	4,904	40.76
536	11,169	3,946	35.33
538	712	52	7.29
551	133,649	25,641	19.19
552	10,645	10,470	98.35
631	11,805	3,916	33.17
676	4,775	327	6.86
694	71,076	2,095	2.95
936	180	2	1.29
945	35,262	4,011	11.37
946	5,825	1,284	22.04
949	109	13	11.87
950	138	4	2.62
951	27,205	8,438	31.02

954	6,192	1,037	16.74
955	84,808	5,390	6.36
956	25,537	2,744	10.74
959	4,851	844	17.40
960	8,702	860	9.88
961	1,676	303	18.10
988	26,683	927	3.47
1023	398,723	18,233	4.57
1024	527,771	50,992	9.66
1025	1,733	26	1.49
1041	2,234	569	25.46
1048	11,115	1,854	16.68
1049	577,540	19,670	3.41
1053	10,665	1,707	16.01
1055	126,779	13,792	10.88
1056	21,063	3,098	14.71
1057	145,277	13,588	9.35
1058	9,359	244	2.61
1059	2,260	13	0.58
1061	42,722	12,494	29.24
1062	19,096	4,604	24.11
1063	478	403	84.31
1065	863	448	51.90
1067	6,029	4,142	68.70
1068	74,915	33,573	44.82
1080	351	49	13.98
1081	15,142	2,266	14.96
1147	7,042	325	4.62
1148	17	17	99.93
1271	837	495	59.14
1413	507,619	128,688	25.35
2048	624	82	13.22
3041	1,010	210	20.81
Total Area	**4,262,332**	**595,410**	**13.97**

Table 2. List of all vegetation associations in the Avon Wheatbelt 2 IBRA sub-region, showing pre-European and remnant vegetation areas, and proportion remaining. Vegetation Associations with less than 10% remaining in the AW2 are shown in light grey, and those with less than 500ha remaining in the AW2 are shown dark grey.

Vegetation Association AW2	Pre-European (hectares)	Remnant Area (hectares)	% Remaining
4	5,099	677	13.28
5	9,716	5,832	60.02
7	2,626	307	11.71
8	2,701	308	11.39
13	239	62	25.84
25	8,311	953	11.46
37	3,285	486	14.78
49	196	64	32.61
125	2,600	84	3.23
128	5,766	1,490	25.84
131	15	0	1.05
142	39,235	1,067	2.72
352	330,417	18,461	5.59
392	5	0	0.00
511	531	34	6.40
551	5,741	857	14.93
676	196	10	5.02
694	78,767	2,770	3.52
936	427	63	14.71
946	36,187	4,272	11.80
947	12,707	2,816	22.17
948	1,440	115	7.98
949	4,040	407	10.08
950	359	186	51.97
951	284	3	1.08
952	494	302	61.17
953	1,889	143	7.58
954	305	6	2.10
955	29,083	2,414	8.30
956	2	0	0.01
959	10	1	5.65
962	62	3	5.60
988	22,765	2,449	10.76
1023	386,278	17,995	4.66
1024	60,852	4,806	7.90
1025	185	6	3.44
1041	270	54	20.12
1048	2,689	519	19.31
1049	255,239	10,409	4.08
1053	2,033	15	0.73

1080	3,597	32	0.88
1147	33,986	2,048	6.03
2047	1,462	945	64.64
3041	2,932	631	21.52
Total Area	**1,355,022**	**84,101**	**6.21**

Table 3. List of all vegetation associations in the Mallee 2 IBRA sub-region, showing pre-European and remnant vegetation areas, and proportion remaining. Vegetation Associations with less than 10% remaining in the MAL2 are shown in light grey, and those with less than 500ha remaining in the MAL2 are shown dark grey.

Vegetation Association MAL2	Pre-European (hectares)	Remnant Area (hectares)	% remaining
8	41,291	2,905	7.04
25	15	0	1.83
37	2,765	1,290	46.67
41	13,771	4,988	36.22
47	9,368	7,163	76.46
59	25	3	13.03
125	84,026	6,376	7.59
128	27,975	11,682	41.76
129	37	2	5.99
131	101,802	5,654	5.55
141	191	22	11.55
142	1,519	448	29.51
380	32,539	13,672	42.02
486	18	18	99.88
511	134,115	43,244	32.24
516	29	5	17.83
519	1,121,628	381,175	33.98
522	412	411	99.88
551	521	245	47.04
552	574	96	16.64
676	2,015	144	7.14
929	227	181	79.66
931	2,216	649	29.27
934	259	89	34.48
936	37,258	18,592	49.90
941	23,443	3,714	15.84
942	36	1	1.55
945	141,346	18,386	13.01
953	7,564	1,289	17.04
955	16,584	1,611	9.71
959	8,225	3,159	38.41
960	211,662	22,184	10.48
961	25,701	3,973	15.46
966	7,085	379	5.35

1005	155	3	2.21
1023	58,586	2,126	3.63
1068	37	35	96.06
1075	174,417	29,606	16.97
1076	11	11	100.05
1079	10,115	3,881	38.36
1094	172	1	0.80
1098	13,664	2,970	21.73
1200	102,519	8,102	7.90
1271	207	37	17.64
1413	4,448	3,823	85.96
2048	300,828	139,852	46.49
Total Area	**2,721,397**	**744,197**	**27.35**

Table 4. List of all vegetation associations in the Coolgardie 2 IBRA sub-region, showing pre-European and remnant vegetation areas, and proportion remaining. Vegetation Associations with less than 10% remaining in the COO2 are shown in light grey, and those with less than 500ha remaining in the COO2 are shown dark grey.

Vegetation Association COO2	Pre-European (hectares)	Remnant Area (hectares)	% Remaining
8	52,122	32,602	62.55
18	15,717	0	0.00
19	475	0	0.00
36	871	870	99.85
39	139	139	100.06
40	7	0	0.00
125	108,236	35,778	33.06
128	69,495	15,495	22.30
141	822,944	117,870	14.32
142	21,442	567	2.64
144	3,991	116	2.91
147	31,275	8,754	27.99
148	320	0	0.00
202	1,845	0	0.00
214	15,714	73	0.47
221	176	177	100.07
256	1,321	788	59.66
314	5,220	5,223	100.07
352	229	0	0.00
357	1	1	100.07
420	107	107	100.07
435	522,370	73,374	14.05
436	1,061	0	0.00
437	110,195	32,062	29.10
468	354	0	0.00
483	28	0	0.00

491	64	64	99.83
501	68	68	100.00
511	458,522	248,171	54.12
519	80,455	80,384	99.91
520	25,036	3,156	12.61
522	123,145	10,975	8.91
535	24,348	20,780	85.35
536	2,008	1,768	88.05
537	701	540	76.97
538	127,941	21,083	16.48
551	23,822	17,710	74.35
552	1,874	1,873	99.94
555	11,683	0	0.00
676	445	445	99.90
936	7,313	2,113	28.89
941	10,831	10,822	99.92
946	787	786	99.93
1024	4,019	2,514	62.55
1063	170,845	154,456	90.41
1067	9,243	8,233	89.07
1068	194,076	100,469	51.77
1071	763	361	47.36
1078	758	757	99.89
1148	195,558	124,883	63.86
1271	52	0	0.79
1413	707,729	338,957	47.89
2048	4,380	4,376	99.92
Total Area	**3,972,119**	**1,479,742**	**37.25**

APPENDIX IV

KEY FLORA INDICATOR SPECIES IN THE IBRA SUB-REGIONS

Key fire response indicator species have been selected that are likely to be sensitive to frequent burning. These are species that have a long juvenile period or time to first flowering, and consequently a long minimum tolerable fire interval (see Part 1, Section 4.1). Obligate seeder species are most sensitive to fire-related decline as these species are often killed by fire and depend on seed germination after the fire. Species that store seed in the soil may also be prone to fire-related decline if they require fire or smoke to germinate seeds, have poor seed viability and are subjected to long fire intervals. This Appendix has selected a small proportion of obligate seeder species with long juvenile periods for each vegetation class that are likely to be sensitive to frequent fire. It also includes some of the common species that need to be considered to ensure structural integrity of the vegetation type, and some species important for fauna habitat. A separate list of rare and priority flora is provided in Appendix V, along with limited fire response data.

These lists should be used with caution, as there are many more species about which we know very little in terms of fire response that may be adversely impacted by considering only the requirements of indicator species. Flora species sensitive to lack of fire, long fire intervals or to inappropriate fire season have not been included here as there is insufficient knowledge of maximum tolerable fire intervals or effect of fire season or intensity for a wide range of species to compile useful guidelines. There is also very little information about how long seeds remain viable in the soil or in canopy stored seed capsules. This is an important area that requires considerable research.

Species data has been selected from the Beard-Hopkins pre-European and remnant vegetation corporate databases, clipped to the IBRA sub-regional boundaries; fire response data and juvenile period data are taken from unpublished databases compiled by Neil Burrows (with permission).

AVON WHEATBELT 1 (AW1)

Table 1. Avon Wheatbelt 1 key indicator species for each vegetation class, showing observed or inferred (?) fire response, observed or inferred (?) Juvenile Period (minimum age to first flowering), and Minimum Fire Interval (3x JP). ([1] Fire response is shown as: RS – resprouter, with type of shoots after fire ep – epicormic shoots, bs – basal shoots; OS – obligate seeder, seed stored mostly on plant and released after fire; RS/OS – able to respond by either method; SS – soil stored seed)

Vegetation Class	Key Indicator Species	Fire Response[1]	Juvenile Period (years)	Minimum Fire Interval (years)
Woodland	**Trees**			
	Allocasuarina huegeliana	OS	5	15
	Eucalyptus capillosa	OS/RS?	?	
	Eucalyptus corrugata	RS?	?	
	Eucalyptus erythronema	RS?	?	
	Eucalyptus longicornis	OS/RS?	?	
	Eucalyptus loxophleba	OS/RS bs	4	12
	Eucalyptus salmonophloia	OS/RS ep	5	15
	Eucalyptus salubris	OS/RS ep?	?	
	Eucalyptus sheathiana	RS?	?	
	Eucalyptus transcontinentalis	OS/RS?	?	
	Eucalyptus wandoo	OS/RS ep	4	12
	Shrubs			
	Acacia acuminata	SS/RS bs	5	15
	Acacia lasiocalyx	SS?	?	
	Acacia microbotrya	SS	4	12
	Allocasuarina campestris	OS	5	15
	Callitris columellaris	OS?	?	
	Eremophila clarkei	?	2	6
	Eremophila drummondii	?	?	
	Eremophila scoparia	?	?	
	Gastrolobium crassifolium	SS	?	
	Grevillea acuaria	?	?	
	Hakea preissii	OS?	?	
	Lepidosperma drummondii	RS bs	6	18
	Leptospermum erubescens	RS bs	4	12
	Melaleuca acuminata	OS	4	12
	Melaleuca lateriflora	OS/RS?	?	
	Melaleuca laxiflora	OS/RS?	6	18
	Melaleuca pungens	OS/RS?	6	18
	Melaleuca uncinata	RS bs	6	18

	Santalum acuminatum *Santalum spicatum*	SS SS	8 ?	24
Low Woodland	**Trees** *Acacia acuminata* *Acacia lasiocalyx* *Allocasuarina huegeliana* *Eucalyptus loxophleba*	 SS/RS bs SS? OS OS/RS bs	 5 ? 5 4	 15 15 12
	Shrubs *Acacia microbotrya* *Allocasuarina campestris* *Dryandra sessilis* *Hakea decurva* *Leptospermum erubescens*	 SS OS OS OS? RS bs	 4 5 2 ? 4	 12 15 6 12
Mallee Shrubland	**Mallee** *Eucalyptus eremophila* *Eucalyptus loxophleba* *Eucalyptus redunca* *Eucalyptus sheathiana* *Eucalyptus transcontinentalis*	 RS bs OS/RS bs RS bs RS? OS/RS?	 ? 4 ? ? ?	 12
	Shrubs *Allocasuarina acutivalvis* *Banksia media* *Callitris columellaris* *Calothamnus gilesii* *Hakea multilineata* *Melaleuca hamulosa* *Melaleuca uncinata*	 OS? OS OS? OS/RS? OS RS bs RS bs	 ? 4 ? 4 5 ? 6	 12 12 15 18
Shrubland	**Mallee** *Eucalyptus burracoppinensis* *Eucalyptus gardneri* *Eucalyptus gracilis* *Eucalyptus leptopoda* *Eucalyptus loxophleba* *Eucalyptus sheathiana* *Eucalyptus wandoo*	 RS? RS? RS? RS? OS/RS bs RS? OS/RS ep	 ? 5 ? ? 4 ? 4	 15 12 12
	Shrubs *Acacia acuminata* *Acacia beauverdiana*	 SS/RS bs ?	 5 ?	 15

	Acacia neurophylla	?	?	
	Actinostrobus arenarius	?	?	
	Allocasuarina acutivalvis	?	?	
	Allocasuarina campestris	OS	5	15
	Allocasuarina huegeliana	OS	5	15
	Banksia audax	OS?/RS bs	3	9
	Callitris preissii	?	?	
	Calothamnus quadrifidus	OS?	3	9
	Dryandra cirsioides	SS	6	18
	Grevillea didymobotrya	?	6	18
	Hakea circumalata	?	?	
	Hakea falcata	OS	6	18
	Hakea gilbertii	?	?	
	Hakea multilineata	OS	5	15
	Hakea subsulcata	?	6	18
	Isopogon teretifolius	?	?	
	Leptopspermum erubescens	RS bs	4	12
	Melaleuca cordata	?	6	18
	Melaleuca lateriflora	?	?	
	Melaleuca uncinata	RS bs	6	18
	Xylomelum angustifolium	?	?	
Shrub Heath	**Mallee**			
	Eucalyptus burracoppinensis	RS?	?	
	Eucalyptus leptopoda	RS?	?	
	Shrubs			
	Adenanthos argyreus	?	?	
	Actinostrobus arenarius	?	?	
	Allocasuarina acutivalvis	?	?	
	Allocasuarina campestris	OS	5	15
	Allocasuarina huegeliana	OS	5	15
	Allocasuarina humilis	RS bs	3	9
	Banksia attenuata	RS ep	4	12
	Banksia audax	OS?/RS bs	3	9
	Banksia menziesii	RS ep	2	6
	Banksia prionotes	OS	3	9
	Beaufortia micrantha	OS	6	18
	Calothamnus lateralis	OS?	2	6
	Calothamnus quadrifidus	OS?	3	9
	Dryandra cirsioides	SS	6	18
	Eremaea pauciflora	SS	4	12

	Grevillea excelsior	SS?	?	
	Hakea multilineata	OS	5	15
	Leptopspermum erubescens	RS bs	4	12
	Nuytsia floribunda	RS ep	2	6
	Santalum acuminatum	SS?	8	24
	Verticordia monadelpha	?	?	
	Verticordia nitens	SS	?	
	Verticordia plumosa	?	?	
	Xylomelum angustifolium	?	?	
Succulent Steppe	**Trees**			
	Eucalyptus kondininensis	OS/RS bs	4	12
	Eucalyptus loxophleba	OS/RS bs	4	12
	Casuarina obesa	?	5	15
	Shrubs			
	Acacia colletioides	SS?	?	
	Acacia ligulata	SS?	?	
	Melaleuca hamulosa	RS bs	?	
	Melaleuca thyoides	RS bs	?	
	Melaleuca uncinata	RS bs	6	18
Mosaic	**Trees**			
	Eucalyptus gracilis	?	?	
	Eucalyptus loxophleba	OS/RS bs	4	12
	Eucalyptus kondininensis	OS/RS bs	4	12
	Eucalyptus occidentalis	RS ep	4	12
	Eucalyptus salmonophloia	OS/RS ep	5	15
	Eucalyptus salubris	OS/RS?	?	
	Eucalyptus sheathiana	RS?	?	
	Shrubs			
	Acacia multispicata	SS	5	15
	Melaleuca hamulosa	?	?	
	Melaleuca uncinata	RS bs	6	18
Rock Outcrops	**Trees**			
	Eucalyptus caesia	RS bs	5	15
	Eucalyptus crucis	?	?	
	Eucalyptus loxophleba	OS/RS bs	4	12
	Shrubs			
	Acacia acuminata	SS/RS bs	5	15
	Acacia lasiocalyx	SS?	?	
	Allocasuarina campestris	OS/SS	5	15
	Allocasuarina huegeliana	OS/SS	5	15
	Calothamnus quadrifidus	RS bs	3	9

	Calytrix angulata	?	?	
	Gastrolobium spinosum	SS	3	9
	Grevillea paniculata	?	4	12
	Hakea petiolaris	OS?	4	12
	Hakea preissii	OS?	?	
	Kunzea pulchella	OS?	3	9
	Leptospermum erubescens	RS bs	4	12
	Melaleuca fulgens	?	4	12
	Melaleuca lateriflora	?	?	
	Melaleuca radula	RS bs	3	9
	Melaleuca uncinata	RS bs?	6	18
	Santalum acuminatum	SS?	8	24
	Thryptomene australis	RS?	?	
	Verticordia preissii	?	?	

AVON WHEATBELT 2 (AW2)

Table 2. Avon Wheatbelt 2 key indicator species for each vegetation class, showing observed or inferred (?) fire response, observed or inferred (?) Juvenile Period (minimum age to first flowering), and Minimum Fire Interval (3x JP). ([1] Fire response is shown as: RS – resprouter, with type of shoots after fire ep – epicormic shoots, bs – basal shoots; OS – obligate seeder, seed stored mostly on plant and released after fire; RS/OS – able to respond by either method; SS – soil stored seed)

Vegetation Class	Key Indicator Species	Fire Response[1]	Juvenile Period (years)	Minimum Fire Interval (years)
Woodland	**Trees**			
	Allocasuarina huegeliana	OS	5	15
	Allocasuarina obesa	?	5	15
	Corymbia calophylla	RS ep	4	12
	Eucalyptus accedens	RS bs	4	12
	Eucalyptus astringens	OS	12	36
	Eucalyptus longicornis	OS/RS?	?	
	Eucalyptus loxophleba	OS/RS bs	4	12
	Eucalyptus rudis	RS ep	4	12
	Eucalyptus salmonophloia	OS/RS ep	5	15
	Eucalyptus salubris	OS/RS ep?	?	
	Eucalyptus wandoo	OS/RS ep	4	12
	Shrubs			
	Acacia acuminata	SS/RS bs	5	15
	Acacia microbotrya	SS	4	12
	Acacia pulchella	SS	5	15
	Astroloma pallidum	RS bs	2	6
	Boronia capitata	SS?	?	
	Calothamnus planifolius	RS bs	5	15
	Dryandra armata	RS bs/SS	4	12
	Dryandra sessilis	OS	2	6
	Grevillea huegelii	?	?	
	Hakea prostrate	RS bs	3	9
	Hovea chorizemifolia	RS bs/SS?	2	6
	Melaleuca lateriflora	OS/RS?	?	
	Melaleuca pauperiflora	?	?	
	Santalum acuminatum	SS	8	24
	Santalum murrayanum	SS?	?	
	Templetonia sulcata	SS?	?	
Low Woodland	**Trees**			
	Acacia acuminata	SS/RS bs	5	15

	Acacia lasiocalyx *Allocasuarina huegeliana* *Eucalyptus loxophleba*	SS? OS OS/RS bs	? 5 4	 15 12
	Shrubs *Acacia microbotrya* *Allocasuarina campestris* *Dryandra sessilis* *Hakea decurva* *Leptospermum erubescens*	 SS OS OS OS? RS bs	 4 5 2 ? 4	 12 15 6 12
Shrubland	**Mallee** *Eucalyptus eremophila* *Eucalyptus gardneri* *Eucalyptus loxophleba* *Eucalyptus wandoo*	 RS bs RS? OS/RS bs OS/RS ep	 ? 5 4 4	 15 12 12
	Shrubs *Acacia acuminata* *Allocasuarina acutivalvis* *Allocasuarina campestris* *Allocasuarina huegeliana* *Calothamnus quadrifidus* *Dryandra nobilis* *Melaleuca uncinata*	 SS/RS bs ? OS OS OS? SS RS bs	 5 ? 5 5 3 5 6	 15 15 15 9 15 18
Shrub Heath	**Mallee** *Eucalyptus todtiana*	 RS ep	 4	 12
	Shrubs *Acacia lasiocarpa* *Actinostrobus arenarius* *Allocasuarina acutivalvis* *Andersonia simplex* *Banksia attenuata* *Banksia menziesii* *Banksia prionotes* *Calothamnus quadrifidus* *Calothamnus sanguineus* *Eremaea pauciflora* *Leptopspermum erubescens* *Leucopogon serratifolium* *Petrophile brevifolia* *Xylomelum angustifolium*	 SS ? ? SS? RS ep RS ep OS OS? RS bs SS RS bs RS bs ? ?	 3 ? ? 4 4 2 3 3 3 4 4 2 ? ?	 9 12 12 6 9 9 9 12 12 6
Succulent	**Trees**			

Steppe	*Casuarina obesa*	?	5	15
	Eucalyptus kondininensis	OS/RS bs	4	12
	Eucalyptus loxophleba	OS/RS bs	4	12
	Shrubs			
	Acacia acutata	SS?	?	
	Acacia colletioides	SS?	?	
	Acacia ligulata	SS?	?	
	Melaleuca eleuterostachya	?	?	
	Melaleuca hamulosa	RS bs	?	
	Melaleuca thyoides	RS bs	?	
	Melaleuca uncinata	RS bs	6	18
Mosaic	**Trees**			
	Allocasuarina huegeliana	OS	5	15
	Eucalyptus drummondii	RS bs	4	12
	Eucalyptus occidentalis	RS ep	4	12
	Eucalyptus salmonophloia	OS/RS ep	5	15
	Eucalyptus salubris	OS/RS?	?	
	Shrubs			
	Allocasuarina campestris	OS	5	15
	Dryandra hewardiana	SS?	?	
	Leptospermum erubescens	RS bs	4	12
	Melaleuca uncinata	RS bs	6	18
Rock Outcrops	**Trees**			
	Eucalyptus caesia	RS bs	5	15
	Eucalyptus loxophleba	OS/RS bs	4	12
	Shrubs			
	Acacia acuminata	SS/RS bs	5	15
	Acacia lasiocalyx	SS?	?	
	Allocasuarina campestris	OS/SS	5	15
	Allocasuarina huegeliana	OS/SS	5	15
	Calothamnus quadrifidus	RS bs	3	9
	Calytrix angulata	?	?	
	Dodonaea viscosa	SS	4	12
	Gastrolobium spinosum	SS	3	9
	Grevillea paniculata	?	4	12
	Hakea petiolaris	OS?	4	12
	Hakea preissii	OS?	?	
	Kunzea pulchella	OS?	3	9
	Leptospermum erubescens	RS bs	4	12
	Melaleuca fulgens	?	4	12
	Melaleuca radula	RS bs	3	9
	Santalum acuminatum	SS?	8	24

	Thryptomene australis	RS?	?	
	Verticordia preissii	?	?	

MALLEE 2 (MAL2)

Table 3. Mallee 2 key indicator species for each vegetation class, showing observed or inferred (?) fire response, observed or inferred (?) Juvenile Period (minimum age to first flowering), and Minimum Fire Interval (3x JP). ([1] Fire response is shown as: RS – resprouter, with type of shoots after fire ep – epicormic shoots, bs – basal shoots; OS – obligate seeder, seed stored mostly on plant and released after fire; RS/OS – able to respond by either method; SS – soil stored seed)

Vegetation Class	Key Indicator Species	Fire Response[1]	Juvenile Period (years)	Minimum Fire Interval (years)
Woodland	**Trees**			
	Eucalyptus flocktoniae	RS bs	?	
	Eucalyptus gracilis	?	?	
	Eucalyptus longicornis	OS/RS?	?	
	Eucaluptus occidentalis	RS ep	4	12
	Eucalyptus platypus	RS bs	4	12
	Eucalyptus salmonophloia	OS/RS ep	5	15
	Eucalyptus salubris	OS/RS ep?	?	
	Eucalyptus spathulata	RS bs	?	
	Eucalyptus transcontinentalis	OS/RS?	?	
	Eucalyptus wandoo	OS/RS ep	4	12
	Shrubs			
	Acacia erinacea	SS?	?	
	Boronia capitata	?	?	
	Gastrolobium celsianum	SS	4	12
	Grevillea huegelii	?	?	
	Melaleuca lateriflora	OS/RS?	?	
	Melaleuca pauperiflora	?	?	
	Santalum acuminatum	SS	8	24
	Templetonia sulcata	SS?	?	
Low Woodland	**Trees**			
	Allocasuarina huegeliana	OS	5	15
	Eucalyptus loxophleba	OS/RS bs	4	12
Mallee Shrubland	**Mallee**			
	Eucalyptus eremophila	RS bs	?	
	Eucalyptus cylindriflora	RS bs	4	12
	Eucalyptus decipiens	RS bs	1	3
	Eucalyptus pleurocarpa	RS bs	3	9
	Eucalyptus redunca	RS bs	?	

	Eucalyptus uncinata *Nuytsia floribunda*	RS bs RS ep	? 2	 6
	Shrubs *Adenanthos cuneatus* *Banksia media* *Hakea laurina* *Hakea multilineata* *Melaleuca hamulosa* *Melaleuca uncinata*	 RS bs OS OS OS RS bs RS bs	 2 4 3 5 ? 6	 6 12 9 15 18
Shrubland	**Mallee** *Eucalyptus flocktoniae* *Eucalyptus redunca* *Eucalyptus uncinata*	 RS bs RS bs RS bs	 ? ? ?	
	Shrubs *Allocasuarina acutivalvis* *Allocasuarina campestris* *Hakea laurina* *Hakea pandanicarpa*	 ? OS OS OS	 ? 5 3 7	 15 9 21
Shrub Heath	**Mallee** *Eucalyptus albida* *Eucalyptus redunca*	 RS? RS bs	 ? ?	
	Shrubs *Acacia acanthoclada* *Acacia fragilis* *Acacia lasiocalyx* *Acacia multispicata* *Adenanthos argyreus* *Adenanthos flavidiflorus* *Allocasuarina campestris* *Banksia gardneri* *Banksia sphaerocarpa* *Grevillea excelsior*	 SS? SS? SS SS ? RS? OS RS bs RS bs S?	 ? ? ? 5 ? ? 5 3 4 ?	 15 15 9 12
Succulent Steppe	**Trees** *Eucalyptus kondininensis*	 OS/RS bs	 4	 12
	Shrubs *Acacia colletioides* *Acacia ligulata* *Melaleuca eleuterostachya* *Melaleuca hamulosa* *Melaleuca thyoides* *Melaleuca uncinata*	 SS? SS? ? RS bs RS bs RS bs	 ? ? ? ? ? 6	 18

Mosaic	**Trees**			
	Allocasuarina huegeliana	OS	5	15
	Eucalyptus annulata	RS?	?	
	Eucalyptus longicornis	OS/RS?	?	
	Eucalyptus loxophleba	OS/RS bs	4	12
	Eucalyptus occidentalis	RS ep	4	12
	Eucalyptus oleosa	RS bs	?	
	Eucalyptus salmonophloia	OS/RS ep	5	15
	Eucalyptus salubris	OS/RS?	?	
	Shrubs			
	Acacia acuminata	SS/RS bs	5	15
	Allocasuarina acutivalvis	?	?	
	Allocasuarina campestris	OS	5	15
	Dryandra cirsioides	OS	6	18
	Leptospermum erubescens	RS bs	4	12
	Melaleuca pauperiflora	?	?	
	Melaleuca uncinata	RS bs	6	18
Rock Outcrops	**Trees**			
	Eucalyptus caesia	RS bs	5	15
	Eucalyptus loxophleba	OS/RS bs	4	12
	Shrubs			
	Acacia acuminata	SS/RS bs	5	15
	Acacia lasiocalyx	SS?	?	
	Allocasuarina campestris	OS/SS	5	15
	Allocasuarina huegeliana	OS/SS	5	15
	Calothamnus quadrifidus	RS bs	3	9
	Daviesia benthamii	SS?	?	
	Dodonaea viscosa	SS	4	12
	Hakea petiolaris	OS?	4	12
	Hakea preissii	OS?	?	
	Kunzea pulchella	OS?	3	9
	Leptospermum erubescens	RS bs	4	12
	Melaleuca fulgens	?	4	12
	Melaleuca lateriflora	?	?	
	Philotheca gardneri	?	?	
	Santalum acuminatum	SS?	8	24
	Santalum spicatum			
	Thryptomene australis	RS?	?	
	Verticordia preissii	?	?	

COOLGARDIE (COO2)

Table 4. Avon Wheatbelt 4 key indicator species for each vegetation class, showing observed or inferred (?) fire response, observed or inferred (?) Juvenile Period (minimum age to first flowering), and Minimum Fire Interval (3x JP). ([1] Fire response is shown as: RS – resprouter, with type of shoots after fire ep – epicormic shoots, bs – basal shoots; OS – obligate seeder, seed stored mostly on plant and released after fire; RS/OS – able to respond by either method; SS – soil stored seed)

Vegetation Class	Key Indicator Species	Fire Response[1]	Juvenile Period (years)	Minimum Fire Interval (years)
Woodland	**Trees**			
	Allocasuarina huegeliana	OS	5	15
	Eucalyptus capillosa	OS/RS?	?	
	Eucalyptus corrugata	RS?	?	
	Eucalyptus erythronema	RS?	?	
	Eucalyptus flocktoniae	RS bs	?	
	Eucalyptus gracilis	?	?	
	Eucalyptus kondininensis	RS bs	?	
	Eucalyptus longicornis	OS/RS?	?	
	Eucalyptus loxophleba	OS/RS bs	4	12
	Eucalyptus oleosa	RS bs	?	
	Eucalyptus salmonophloia	OS/RS ep	5	15
	Eucalyptus salubris	OS/RS ep?	?	
	Eucalyptus transcontinentalis	OS/RS?	?	
	Eucalyptus wandoo	OS/RS ep	4	12
	Shrubs			
	Acacia acuminata	SS/RS bs	5	15
	Acacia erinacea	SS?	?	
	Acacia ramulosa	SS?	?	
	Eremophila interstans	?	?	
	Eremophila scoparia	?	?	
	Exocarpus aphyllus	?	?	
	Grevillea huegelii	?	?	
	Melaleuca lateriflora	OS/RS?	?	
	Melaleuca pauperiflora	?	?	
	Santalum acuminatum	SS	8	24
	Templetonia sulcata	SS?	?	
Low Woodland	**Trees**			
	Eucalyptus loxophleba	OS/RS bs	4	12
	Callitris columellaris	OS?	?	

Mallee Shrubland	**Mallee** *Eucalyptus cylindriflora* *Eucalyptus eremophila* *Eucalyptus loxophleba* *Eucalyptus oleosa*	 RS bs RS bs OS/RS bs RS bs	 4 ? 4 ?	 12 12
	Shrubs *Acacia acuminata* *Acacia prainii* *Banksia media* *Callitris columellaris* *Hakea multilineata* *Melaleuca uncinata*	 SS/RS bs SS? OS OS? OS RS bs	 5 ? 4 ? 5 6	 15 12 15 18
Shrubland	**Mallee** *Brachychiton gregorii* *Callitris columellaris* *Eucalyptus oleosa*	 ? OS? RS bs	 8 ? ?	 24
	Shrubs *Acacia acuminata* *Acacia aneura* *Acacia quadrimarginea* *Acacia ramulosa* *Acacia stereophylla* *Allocasuarina acutivalvis* *Allocasuarina campestris* *Allocasuarina corniculata* *Dodonaea viscosa* *Euryomyrtus maidenii* *Grevillea paradoxa* *Hakea falcata* *Melaleuca cordata* *Melaleuca uncinata* *Senna artemisioides* *Thryptomene australis* *Thryptomene kochii* *Triodia rigidissima* *Triodia scariosa*	 SS/RS bs SS/RS bs SS? SS? SS? ? OS ? SS ? ? OS ? RS bs SS? RS? ? SS? SS?	 5 4 ? ? ? ? 5 4 4 ? ? 6 6 6 4 ? ? ? ?	 15 12 15 12 12 18 18 18 12
Shrub Heath	**Mallee** *Eucalyptus burracoppinensis* *Eucalyptus incrassata*	 RS? RS?	 ? ?	

	Shrubs *Acacia acanthoclada* *Acacia beauverdiana* *Acacia multispicata* *Adenanthos argyreus* *Baeckea grandibracteata* *Banksia audax* *Banksia sphaerocarpa* *Grevillea excelsior* *Triodia rigidissima*	 SS? SS? SS ? ? RS bs RS bs SS? SS?	 ? ? 5 ? ? 3 4 ? ?	 15 9 12
Succulent Steppe	**Trees** *Eucalyptus loxophleba*	 OS/RS bs	 4	 12
	Shrubs *Acacia acuminata* *Acacia tetragonophylla* *Callitris columellaris* *Eremophila oldfieldii* *Grevillea sarissa* *Melaleuca lateriflora*	 SS SS OS? ? ? OS/RS?	 5 6 ? ? ? ?	 15 18
Mosaic	**Trees** *Eucalyptus flocktoniae* *Eucalyptus oleosa*	 RS bs RS bs	 ? ?	
	Shrubs *Eremophila dempsteri* *Melaleuca pauperiflora*	 ? ?	 ? ?	
Rock Outcrops	**Trees** *Allocasuarina huegeliana* *Eucalyptus caesia* *Eucalyptus crucis* *Eucalyptus orbifolia* *Eucalyptus websteriana* *Pittosporum phylliraeoides*	 OS/SS RS bs ? RS? RS? RS bs	 5 5 ? ? ? 7	 15 15 21
	Shrubs *Acacia acuminata* *Acacia jibberdingensis* *Acacia lasiocalyx* *Acacia multispicata* *Allocasuarina acutivalvis* *Allocasuarina campestris*	 SS/RS bs SS? SS? SS ? OS/SS	 5 ? ? 5 ? 5	 15 15 15

	Allocasuarina dielsiana	?	?	
	Alyxia buxifolia	?	?	
	Boronia ternata	?	?	
	Diplolaena micrcephala	?	?	
	Dodonaea inaequifolia	?	3	9
	Dodonaea viscosa	SS	4	12
	Grevillea paniculata	?	4	12
	Hakea preissii	OS?	?	
	Hakea recurva	?	?	
	Kunzea pulchella	OS?	3	9
	Leptospermum roei	?	?	
	Melaleuca fulgens	?	4	12
	Melaleuca uncinata	RS bs?	6	18
	Santalum acuminatum	SS?	8	24
	Santalum spicatum	SS?	?	
	Thryptomene australis	RS?	?	

APPENDIX V

RARE AND PRIORITY FLORA SPECIES IN THE IBRA SUB-REGIONS

The Department of Environment and Conservation's corporate database for rare and priority flora, called the Declared Rare Flora database (DEFL), was intersected with the boundaries of the Avon Basin NRM used in these Fire Management Guidelines, and the individual IBRA sub-regional boundaries for the Avon Wheatbelt 1 (AW1), Avon Wheatbelt 2 (AW2), Mallee 2 (MAL2) and Coolgardie 2 (COO2). These datasets therefore only contain those species within each sub-region that is also contained within the Avon Basin boundary. The conservation codes were current for the DEFL database at 30.9.2006. It is likely that there have been some name revisions and changes in conservation codes since that time. The location of rare and priority flora populations can be determined from the DEC corporate database GIS layer but the identity of each population is generally not divulged to the public. There may be other populations that have been recorded in the WA Herbarium specimen database (Florabase) that have not been captured in the DEFL database.

The fire response data for declared rare species has been compiled by Dr Colin Yates from DEC district and regional records, Brown *et. al.* (1998) and from his personal observations (C. Yates, unpublished data). However, there was insufficient fire response data for priority species in the database compiled by Dr Neil Burrows to be useful. The fire response for many threatened species is unknown, and applications to take threatened flora during prescribed burning operations will only be approved after careful consideration of each individual population. Where possible, every effort should be made to protect populations of declared rare flora from wildfire until their fire response is well known.

Table 1. Total number of declared and priority taxa in each IBRA sub-region, within the Avon Basin NRM boundary. Some taxa may occur in more than one sub-region.

IBRA sub-region	No. of Taxa	No. populations and sub-populations
AW1	97	488
AW2	92	531
MAL2	117	627
COO2	48	170
Total	**354**	**1816**

AVON WHEATBELT 1 (AW1)

Table 2. Avon Wheatbelt 1 rare (R) and priority species (P1 – P4) showing observed (*) or inferred (?) fire response for rare species (C. Yates, unpublished data). (Fire response is shown as: RS – resprouter, with type of shoots after fire ep – epicormic shoots, bs – basal shoots; OS – obligate seeder, seed stored mostly on plant and released after fire; RS/OS – able to respond by either method; SS – soil stored seed; GEO – geophyte; AQ – aquatic plant)

Conservation code	**Species**	**Fire response**
R	*Acacia ataxiphylla* subsp. *magna* *Acacia cochlocarpa* subsp. *velutinosa* *Acacia denticulosa* *Acacia lobulata* *Acacia sciophanes* *Acacia volubilis* *Banksia cuneata* *Boronia adamsiana* *Caladenia drakeoides* *Calectasia pignattiana* *Conostylis wonganensis* *Cyphanthera odgersii* subsp. *occidentalis* *Daviesia cunderdin* *Daviesia euphorbioides* *Eremophila resinosa* *Eremophila virens* *Eremophila viscida* *Eucalyptus brevipes* *Eucalyptus crucis* subsp. *crucis* *Eucalyptus recta* *Eucalyptus synandra* *Frankenia conferta* *Frankenia parvula* *Gastrolobium diablophyllum* *Gastrolobium graniticum* *Grevillea dryandroides* subsp. *dryandroides* *Grevillea dryandroides* subsp. *hirsuta* *Grevillea scapigera* *Guichenotia seorsiflora* *Gyrostemon reticulatus* *Hakea aculeata* *Hemiandra rutilans* *Isopogon robustus*	OS/RS? ? OS? OS* OS? ? OS* GEO ? OS* OS* OS* OS* RS* OS? ? RS? RS? ? RS/OS? ? ? ? ? RS* RS* RS? ? RS? ?

	Jacksonia quairading *Melaleuca sciotostyla* *Microcorys eremophiloides* *Myriophyllum lapidicola* *Philotheca basistyla* *Pityrodia scabra* *Ptilotus fasciculatus* *Rhizanthella gardneri* *Roycea pycnophylloides* *Stylidium merrallii* *Symonanthus bancroftii* *Tetratheca deltoidea* *Verticordia hughanii*	? OS* ? AQ ? ? ? GEO ? ? ? ? OS?
P1	*Acacia caesariata* *Acacia desertorum* var. *nudipes* *Acacia sclerophylla* var. *teretiuscula* *Acacia tetraneura* *Dampiera glabrescens* *Dampiera scaevolina* *Eucalyptus myriadena* subsp. *parviflora* *Eucalyptus subangusta* subsp. *virescens* *Gastrolobium tenue* *Grevillea minutiflora* *Guichenotia glandulosa* *Jacksonia debilis* *Leucopogon teretostylus*	
P2	*Acacia cowaniana* *Acacia lirellata* subsp. *compressa* *Acacia sclerophylla* var. *pilosa* *Dryandra lindleyana* subsp. *agricola* *Dryandra speciosa* subsp. *speciosa* *Eremophila adenotricha* *Eremophila brevifolia* *Eremophila complanata* *Fitzwillia axilliflora* *Grevillea rosieri* *Hakea pendens* *Lepidium genistoides* *Leucopogon amplectens* *Leucopogon* sp. Bungulla (R.D.Royce 3435)	
P3	*Acacia campylophylla* *Boronia penicillata* *Cryptandra dielsii* *Frankenia glomerata* *Leucopogon* sp. Ironcaps (N.Gibson & K.Brown 3070)	

	Melaleuca sclerophylla *Monotoca leucantha* *Stylidium rhipidium* *Verticordia huegelii* var. *tridens*	
P4	*Acacia merrickiae* *Acacia semicircinalis* *Caladenia cristata* *Calothamnus brevifolius* *Centrolepis caespitosa* *Daviesia oxylobium* *Eremophila caerulea* subsp. *merrallii* *Eucalyptus latens* *Gastrolobium callistachys* *Gonocarpus intricatus* *Lechenaultia pulvinaris* *Lepidium pseudotasmanicum* *Myriophyllum petraeum* *Stylidium scabridum* *Verreauxia verreauxii*	

AVON WHEATBELT 2 (AW2)

Table 2. Avon Wheatbelt 2 rare (R) and priority (P1 – P4) species showing observed (*) or inferred (?) fire response for rare species (C. Yates, unpublished data). (Fire response is shown as: RS – resprouter, with type of shoots after fire ep – epicormic shoots, bs – basal shoots; OS – obligate seeder, seed stored mostly on plant and released after fire; RS/OS – able to respond by either method; SS – soil stored seed; GEO - geophyte)

Conservation code	**Species**	**Fire response**
R	*Acacia aphylla* *Acacia ataxiphylla* subsp. *magna* *Acacia brachypoda* *Acacia denticulosa* *Acacia pharangites* *Acacia pygmaea* *Acacia subflexuosa* subsp. *capillata* *Acacia vassalii* *Acacia volubilis* *Allocasuarina fibrosa* *Banksia cuneata* *Boronia capitata* subsp. *capitata* *Caladenia drakeoides* *Caladenia williamsiae* *Conostylis wonganensis* *Daviesia euphorbioides* *Dryandra ionthocarpa* subsp. *chrysophoenix* *Eremophila nivea* *Eremophila ternifolia* *Eucalyptus recta* *Gastrolobium glaucum* *Gastrolobium hamulosum* *Grevillea christineae* *Grevillea dryandroides* subsp. *hirsuta* *Grevillea scapigera* *Guichenotia seorsiflora* *Hakea aculeata* *Lasiopetalum rotundifolium* *Lechenaultia laricina* *Lysiosepalum abollatum*	OS* OS/RS? OS* OS? OS? OS? OS? OS? ? RS* OS* ? GEO GEO ? OS* OS/RS? OS? OS? ? ? RS? OS? RS? RS* RS? ? RS? OS* RS* ?

	Melaleuca sciotostyla *Microcorys eremophiloides* *Philotheca wonganensis* *Rhagodia acicularis* *Rhizanthella gardneri* *Roycea pycnophylloides* *Stylidium coroniforme* subsp. *coroniforme* *Thelymitra stellata* *Thomasia glabripetala* *Thomasia montana* *Verticordia fimbrilepis* subsp. *fimbrilepis* *Verticordia staminosa* subsp. *staminosa*	? ? ? ? GEO ? ? GEO ? ? OS* OS* OS?
P1	*Acacia sclerophylla* var. *teretiuscula* *Acacia trinalis* *Gastrolobium rotundifolium* *Senecio gilbertii* *Thysanotus sabulosus*	
P2	*Acacia congesta* subsp. *wonganensis* *Acacia drewiana* subsp. *minor* *Acacia gemina* *Acacia tuberculata* *Amperea micrantha* *Andersonia carinata* *Boronia ericifolia* *Calytrix oncophylla* *Dryandra lindleyana* subsp. *agricola* *Dryandra speciosa* subsp. *speciosa* *Eremophila sargentii* *Grevillea biformis* subsp. *cymbiformis* *Grevillea kenneallyi* *Leucopogon amplectens* *Leucopogon* sp.Bungulla (R.D.Royce 3435) *Persoonia hakeiformis*	
P3	*Acacia anarthros* *Acacia campylophylla* *Anigozanthos bicolor* subsp. *exstans* *Boronia penicillata* *Dryandra meganotia* *Eucalyptus macrocarpa* x *pyriformis* *Melaleuca sclerophylla* *Phebalium brachycalyx*	

	Phlegmatospermum drummondii *Tetratheca similes*	
P4	*Acacia cuneifolia* *Acacia semicircinalis* *Anigozanthos humilis* subsp. *chrysanthus* *Asterolasia grandiflora* *Caladenia integra* *Calothamnus brevifolius* *Calothamnus rupestris* *Daviesia oxylobium* *Daviesia spiralis* *Eremaea blackwelliana* *Eremophila veneta* *Eucalyptus exilis* *Eucalyptus latens* *Eucalyptus loxophleba* x *wandoo* *Gastrolobium callistachys* *Gastrolobium densifolium* *Lechenaultia pulvinaris* *Persoonia sulcata* *Wurmbea drummondii*	

MALLEE 2 (MAL2)

Table 2. Mallee 2 rare (R) and priority (P1 – P4) species showing observed (*) or inferred (?) fire response for rare species (C. Yates, unpublished data). (Fire response is shown as: RS – resprouter, with type of shoots after fire ep – epicormic shoots, bs – basal shoots; OS – obligate seeder, seed stored mostly on plant and released after fire; RS/OS – able to respond by either method; SS – soil stored seed; GEO - geophyte)

Conservation code	Species	Fire response
R	*Acacia auratiflora*	OS?
	Acacia depressa	OS?
	Acacia lanuginophylla	OS?
	Acacia leptalea	OS?
	Adenanthos pungens subsp. *pungens*	OS*
	Allocasuarina tortiramula	?
	Anigozanthos bicolor subsp. *minor*	OS*
	Banksia sphaerocarpa var. *dolichostyla*	RS*
	Boronia revoluta	?
	Caladenia drakeoides	GEO
	Caladenia graniticola	GEO
	Caladenia melanema	GEO
	Calectasia pignattiana	OS*
	Conostylis rogeri	?
	Conostylis seorsiflora subsp. *trichophylla*	RS?
	Drakaea isolata	GEO
	Eremophila subteretifolia	OS?
	Eremophila verticillata	OS*
	Goodenia integerrima	?
	Grevillea involucrata	RS*
	Muehlenbeckia horrida subsp. *abdita*	OS?
	Ptilotus fasciculatus	?
	Roycea pycnophylloides	?
	Tetratheca aphylla	?
	Thelymitra psammophila	GEO
	Tribonanthes purpurea	GEO
	Verticordia staminosa var. *cylindracea*	OS?
	Verticordia staminosa var. *erecta*	OS?
P1	*Acacia lanei*	
	Acacia mutabilis subsp. *stipulifera*	
	Acacia sclerophylla var. *teretiuscula*	
	Acacia tetraneura	
	Baeckea crispiflora subsp. Ongerup (A.Scougall &	

	C.Garawanta E35) *Brachyloma nguba* *Dampiera scaevolina* *Darwinia divisa* *Drosera grievei* *Eucalyptus myriadena* subsp. *parviflora* *Eucalyptus subangusta* subsp. *virescens* *Grevillea lullfitzii* *Hibbertia axillibarba* *Hydrocotyle hexaptera* *Hydrocotyle muriculata* *Jacksonia debilis* *Melaleuca agathosmoides* *Mirbelia densiflora* *Pimelea pelinos* *Thysanotus lavanduliflorus* *Thysanotus sabulosus* *Xanthoparmelia nashii* *Xanthoparmelia scabrosina*	
P2	*Acacia cowaniana* *Acacia drewiana* subsp. *minor* *Acacia heterochroa* subsp. *robertii* *Acacia mutabilis* subsp. *incurva* *Acacia sclerophylla* var. *pilosa* *Acacia tuberculata* *Astartea clavifolia* *Boronia ericifolia* *Dampiera orchardii* *Daviesia lineata* *Drosera salina* *Dryandra conferta* var. *parva* *Dryandra epimicta* *Dryandra erythrocephala* var. *inopinata* *Dryandra foliosissima* *Dryandra idiogenes* *Eucalyptus sparsicoma* *Fitzwillia axilliflora* *Gastrolobium effusum* *Gastrolobium rigidum* *Goodenia* sp. Lake King (M.Gustafsson et K.Bremer 132) *Guichenotia asteriskos* *Haegiela tatei* *Lepidobolus spiralis* *Millotia steetziana* *Opercularia rubioides*	

	Persoonia hakeiformis *Pimelea halophila* *Stylidium sejunctum* *Synaphea canaliculata* *Synaphea cervifolia* *Synaphea flexuosa* *Synaphea parviflora* *Synaphea tripartita* *Thysanotus acerosifolius*	
P3	*Acacia undosa* *Boronia penicillata* *Bossiaea divaricata* *Daviesia elongata* subsp. *implexa* *Daviesia tortuosa* *Eucalyptus microschema* *Eucalyptus mimica* subsp. *mimica* *Eucalyptus quaerenda* *Frankenia glomerata* *Leucopogon* sp. Ironcaps (N.Gibson & K.Brown 3070) *Melaleuca sculponeata* *Monotoca leucantha* *Persoonia brevirhachis* *Phebalium brachycalyx* *Pityrodia* sp. Yilgarn (A.P.Brown 2679) *Pultenaea daena* *Stylidium pulviniforme* *Stylidium rhipidium*	
P4	*Bentleya spinescens* *Calamphoreus inflatus* *Calothamnus affinis* *Daviesia purpurascens* *Eremophila serpens* *Eremophila veneta* *Eucalyptus latens* *Gastrolobium densifolium* *Grevillea prostrata* *Microcorys* sp. Forrestania (V.English 2004) *Myriophyllum petraeum* *Rinzia affinis* *Thysanotus glaucus*	

COOLGARDIE 2 (COO2)

Table 2. Coolgardie 2 rare (R) and priority (P1 – P4) species showing observed (*) or inferred (?) fire response for rare species (C. Yates, unpublished data). (Fire response is shown as: RS – resprouter, with type of shoots after fire ep – epicormic shoots, bs – basal shoots; OS – obligate seeder, seed stored mostly on plant and released after fire; RS/OS – able to respond by either method; SS – soil stored seed; GEO – geophyte; AQ – aquatic plant)

Conservation code	Species	Fire response
R	*Acacia denticulosa*	OS?
	Acacia lobulata	OS*
	Banksia sphaerocarpa var. *dolichostyla*	RS*
	Boronia adamsiana	?
	Boronia revoluta	?
	Daviesia microcarpa	OS?
	Eucalyptus brevipes	RS?
	Eucalyptus steedmanii	OS*
	Eucalyptus synandra	RS/OS?
	Frankenia parvula	?
	Gastrolobium graniticum	?
	Leucopogon sp. Helena & Aurora Range (B.J. Lepschi 2077)	?
	Muelleranthus crenulatus	?
	Myriophyllum lapidicola	AQ
	Ricinocarpos brevis	?
	Stylidium merrallii	?
	Tetratheca aphylla	?
	Tetratheca erubescens	?
	Tetratheca harperi	?
	Tetratheca paynterae subsp. *cremnobata*	?
	Tetratheca paynterae subsp. *paynterae*	?
P1	*Acacia desertorum* var. *nudipes*	
	Beyeria sp. Jackson Range (R. Cranfield & P. Spencer 7751)	
	Dampiera scaevolina	
	Eucalyptus myriadena subsp. *parviflora*	
	Gastrolobium tenue	
	Grevillea marriottii	
	Grevillea phillipsiana	
P2	*Dryandra epimicta*	
	Haegiela tatei	

	Hakea pendens *Keraudrenia adenogyna* *Lepidium genistoides* *Lissanthe scabra* *Microcorys lenticularis* *Stylidium sejunctum* *Synaphea tripartita*	
P3	*Daviesia elongata* subsp*. implexa* *Galium migrans* *Pityrodia* sp. Yilgarn (A.P. Brown 2679) *Stylidium pulviniforme*	
P4	*Calamphoreus inflatus* *Eremophila caerulea* subsp. *merrallii* *Eremophila racemosa* *Grevillea prostrata* *Microcorys* sp.Forrestania (V.English 2004) *Myriophyllum petraeum* *Sowerbaea multicaulis*	

FIRE & BIODIVERSITY
GUIDELINES FOR THE AVON BASIN

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