A Review of Current Knowledge and Literature to Assist in Determining Ecologically Sustainable Fire Regimes for the Southeast Queensland Region

Cuong Tran and Clyde Wild

August 2000

Griffith University
and
The Fire and Biodiversity Consortium

This project was funded by the following agencies, shires and councils

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Also including: ***Toowoomba Shire Council*, *Tweed Shire Council* and *Beaudesert Shire Council***

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Executive Summary

Fire exerts a dominating influence on the Australian landscape. Southeast Queensland is recognised as containing one of Australia's highest levels of biodiversity, with a high degree of localised endemism. This region contains a wide variety of habitat types occurring on different geological formations. This project was aimed at determining ecologically sound fire regimes that would maintain the high level of biodiversity within this region. This involved a comprehensive review of the currently available literature. The analysis of the literature showed that much research has been conducted, especially in other areas of Australia, and the majority of this research is still ongoing. Further, the review also showed that some specific areas within the southeast Queensland region (such as Cooloola National Park, Beerwah etc.) have been extensively studied for fire effects on native flora and fauna.

However, the majority of the other areas and habitats within this region have been poorly examined, and only recently have research projects been initiated to examine the effects of different fire regimes. Nevertheless, a considerable amount of anecdotal evidence and personal observations on associated fire regimes exists, but requires documentation, before the important information is lost.

The literature was also examined to determine if useful indicator species (floral and/or faunal) could be identified to accurately evaluate the efficacy of an implemented fire regime for a specific broad vegetation type. The results of the literature review indicated that there are many potential floral and faunal species to use as indicators of fire effects upon biodiversity. However, this too requires much more research.

Ecologically Sensitive Fire Regimes – refer supporting documentation below table

For the prioritised vegetation types desirable fire regimes identified in this work include:

PRIORITY	BROAD COMMUNITY TYPE	FIRE FREQUENCY RECOMMENDATIONS	
		MAIN	OTHER
High	*Wet Sclerophyll Closed Forest*	5, 20–50[a] years	More than 200 years
High	*Dry Eucalypt Open Forest with Grassy understorey*	(minimum of) 4–5 years, 8 years	More than 10 years
High	*Dry Eucalypt Open Forest with Shrubby understorey*	7–12 years	
High	*Melaleuca Forest*	More than 15 years	
High	*Coastal Woodland–Open Forest*	(minimum of) 8–10 [b] years	More than 10 years, incl. high intensity fire
Medium	Subtropical Rainforest	Total exclusion, wildfire events may occur at more than 200 year intervals	
Medium	Dry Rainforest	Total exclusion	
Medium	Mountain Heath	More than 15 years	
Medium	Estuarine Complexes	More than 10 years	No fire
Medium	Wet Lowland Heath	(minimum of) 8–10[c] years, but unsure	
Medium	Dry Lowland Heath	(minimum of) 8–10[c] years, but unsure	
Low	Naturally Bare Areas	None stated. Most likely fire is unimportant	
Low	Cleared Areas	None stated. Most likely fire is unimportant	

[a] Burn at rainforest/wet sclerophyll edge, usually with follow up fire within two years of first fire (with 20–50 year initial interval)

[b] Information retrieved from areas outside SEQ. This would be the absolute minimum periods, not suggested fire frequencies.

[c] Work completed in SEQ. However, focussed on one rare species with no longer-term studies. Again, these are suggested minimum periods.

NB: For all vegetation types, there is widespread agreement that increasing the *irregularity* (ie the variability) of fire frequencies will positively influence the biodiversity within these areas.

The fire regimes outlines above should only be used as a guide. It should be noted that these fire intervals were drawn from a variety (of quality) of literature, and should not be treated as 'hard-and-fast' regimes. Each region within southeast Queensland will differ in its response (sometimes markedly) to an imposed fire regime, which only shows the need to initiate research programs to thoroughly investigate such impacts. The review also highlighted the need to consider maximising the variability in the inter-fire intervals. This was stated as being crucial for the survival of many species, mainly rare or threatened species. As a result of this review, increasing the variability of fire frequencies is highly recommended. This can be achieved by incorporating wildfire events into the fire frequencies, allowing fires to be burn 'unbounded', and when prescription burning is performed, the fires are allowed to create their own mosaics. These points need to be accounted for when deciding which fire frequencies should be implemented. Furthermore, this review also showed that there has been only a very small amount of work on determining the *maximum* inter-fire intervals. In the light of this, these desirable fire regimes listed above are suggested minimum fire intervals. There are no maximum intervals of the fire frequencies which can be reliably established for any of the forest types listed above. Unless work is initiated to determine these maximum intervals, then any fire regime listing any strict upper and lower limits would be only speculation, and should be treated as such.

It is clear that more work is required throughout the region to determine appropriate fire regimes to maintain biodiversity. In particular, research is urgently needed in sclerophyll open forest (eucalyptus communities) with both grassy and shrubby understoreys, as this is the dominant vegetation community within the region, comprises the largest land area in southeast Queensland, contains a considerable amount of biotic diversity and is in closest proximity to occupied land. The wet sclerophyll/rainforest boundary is another area where much contention over the appropriate fire regimes exists. Clearly, with diminishing amounts of this very specific and unique vegetation type, work is urgently needed here. Furthermore, work is also needed in coastal and mountain heathland, as these areas have high levels of endemism, and are under considerable pressure from urbanisation (for coastal heathland) and ecologically damaging fire effects.

Potential Indicator Species of Fire Regimes

Some *potential* indicator species for specific vegetation types that were identified include:

PRIORITY	BROAD COMMUNITY TYPE	POSSIBLE INDICATOR SPECIES
High	*Wet Sclerophyll Closed Forest*	Possibly Eastern Bristlebird (*Dasyornis brachyterus*) especially areas bordering Rainforest elements. Invertebrates
High	*Dry Eucalypt Open Forest with Grassy understorey*	Invertebrates (Rapid Biodiversity Assessment)
High	*Dry Eucalypt Open Forest with Shrubby understorey*	Brush-Tailed Rock Wallaby (*Petrogale pencillata*), Black-Breasted Button Quail (*Turnix melanogaster*), amongst many, also Invertebrates
High	*Melaleuca Forest*	None stated. Herptiles distinct possibility
High	*Coastal Woodland–Open Forest*	Possibly *Allocasuarina littoralis.* Invertebrates
Medium	Subtropical Rainforest	Invertebrates and possibly epiphytes
Medium	Dry Rainforest	*Lantana camara* (negative indicator of biodiversity). Invertebrates other distinct possibility
Medium	Mountain Heath	Numerous: birds and rodents distinct possibility
Medium	Estuarine Complexes	None identified
Medium	Wet Lowland Heath	Ground Parrot, Invertebrates
Medium	Dry Lowland Heath	Ground Parrot, Invertebrates
Low	Naturally Bare Areas	None Stated
Low	Cleared Areas	None Stated

The indicator species above were mainly selected from reviewing the relevant articles and deciding upon those species that would best represent each individual vegetation type. Recommendations of other likely indicator species from other researchers were also noted during the decision-making process. It is important to recognise that possible indicator species may not necessarily have to originate from 'noteworthy species' such as those under threat from the impacts of fire. Examination of the available literature shows that common species, such as invertebrates, avifauna and herpetofauna may be excellent indicators of ecosystem health. Furthermore, it was shown that climatic conditions, topography and other easily measurable factors could be used to describe the 'proneness' to fire for certain regions, to identify fire seasons and provide an indication of fire frequency ranges.

The available literature was ranked for 'Reliability' and 'Relevance' to the southeast Queensland region with a scale of 1–5 (lowest–highest) according to the source of the article, scientifically-sound methodology, accurate statistical analysis and then included in an updateable database. The majority of currently available literature shows that most research relevant to this region registered lower on the reliability-rank of the reference. This indicates that there has been little scientifically rigorous examination of fire and its effects in southeast Queensland. There are some existing fire management plans, especially for Conservation Areas, National Parks and property estates, which have included the ecological implications of altering or enforcing certain fire frequencies in these areas. Although there are some continuing long-term projects investigating fire regimes and its impacts on the landscape, most of this work remains unpublished. Furthermore the literature review showed that the majority of the work completed on fire relates to the recovery processes post-fire, but relatively few studies on the effects of several previous fires on our present vegetation communities.

Following the review, the gaps identified in the review can be summarised as:

- Fire History – knowledge and extent of previous fires is essential for examination of current patterns in vegetation structure and determination of effects on overall biodiversity
- Fire Season – overall, a poorly studied subject throughout Australia, especially in southeast Queensland.
- Multiple fire events in one area are also poorly examined. Most of the available work has concentrated on a single fire and neglect to consider the effects of many fires, particularly on species diversity.
- More specifically, *Melaleuca* Forest and Coastal Woodlands were identified as community types that contained the least amount of information on the effects of fire on the biodiversity within these areas.
- The boundaries between Rainforest and Wet-Sclerophyll forest contain a rich plant and animal diversity. These areas are also subject to the widest variations in suggested fire regimes and desirable fire frequencies. Research in this area is urgently required.
- While a considerable amount of information exists on *Eucalyptus* Forest with Grassy and Shrubby understoreys, there remains much conjecture on the effects on imposed fire regimes on overall biodiversity. The effect of multiple fire events on biodiversity in these areas is required.
- To some extent, the use of invertebrates as a general indicator of ecosystem 'health' is well established, particularly for fire events. Continuation of research into the use of this indicator group is encouraged.
- There are some existing methods on the evaluation and assessment of fire regimes on a regional and community vegetation-type level. Some of these methods can be implemented using retrospective or inferential analysis. This may provide some important

clues to previous fire regimes and their effects on biodiversity. These methods require further work to determine the level of reliability for the southeast Queensland area.

Finally, even though there is a limited amount of information on fire and its effects within the southeast Queensland context, there is a great amount of interest from government agencies and local shires and councils. This interest will hopefully extend to lead to the initiation and maintenance of experiments to answer some fundamental questions in relation to fire and ecologically sound fire regimes for southeast Queensland.

Biological diversity (**biodiversity**) can be defined as:

> *"The variety of all life forms – the different plants, animals and micro-organisms, the genes they contain and the ecosystems of which they form a part. It is not a fixed entity, but constantly changing; it is increased by genetic change and by evolutionary processes and reduced by extinction and habitat degradation. The concept emphasises the inter-relatedness of the biological world. It covers the terrestrial, aquatic and marine environments."*

(Biological Diversity Advisory Committee 1992)

This is an apt statement regarding the state of fire research in Australia:

> *"Considering the frequency with which fires occur in eucalypt forests and woodlands and their dramatic impact on the landscape, there is remarkable little information about the effects of fire on fauna or the long-term consequence of burning of forest ecosystems."*

(Recher *et al*. 1985, as quoted in Gill *et al*. 1999)

Aims of Research

- To determine the ecologically sustainable fire regimes for the south-east Queensland region
- To examine existing literature on fire management in the south-east Queensland region
- To outline future research guidelines for fire management and conservation of biodiversity

Description of Southeast Queensland Region

The southeast Queensland region covers a broad area of between 6.15–6.6 million ha (Sattler and Williams 1999, Queensland CRA/RFA Steering Committee 1998b) and is considered to be among Australia's richest areas of flora and fauna (Sattler and Williams 1999, Gold Coast Nature Conservation Strategy 1998). Within the southeast Queensland region, 44.2% is forested (native = 2.54 m ha and plantation =0.18 m ha), with a climate that is typically by warm to very warm, wet summers and mild to cool dry winters (Queensland CRA/RFA Steering Committee 1998b).

Furthermore, the area contains numerous localised centres of endemism occurring within numerous habitat types. Young and Cotterall (1993, as cited in Sattler and Williams 1999) originally defined thirteen distinct 'environmental provinces' for the south-east Queensland region, but upon review of geological information, ten provinces are now recognised within the bioregion. These region are summarised below:

Table 1: Provinces in SEQ Bioregion (reproduced from Sattler and Williams 1999)

PROVINCE	INFORMATION/DETAILS
Province 1 – Scenic Rim	Covers McPherson and Main Ranges. Major vegetation types are: complex notophyll rainforest & tall open forests. Province closely aligned to parts of northern New South Wales
Province 2 – Moreton Basin	Area consisting of low, hilly relief and broad alluvial valleys. Some areas considered dry (<750 mm rainfall/year). Major vegetation types are: eucalypt woodlands and open forest, *Acacia harpophylla* open forest and semi-evergreen vine thicket also present.
Province 3 – Southeast Hills and Ranges	Area within Beenleigh and North and South D'Aguilar Blocks, has moist climate and is mostly hilly to mountainous. Main vegetation types include: eucalypt open forest, eucalypt tall open forests, complex notophyll rainforest and araucarian notophyll rainforest.
Province 4 – Southern Coastal Lowlands	Consist of Nambour basin, but covers southern offshore islands. Major vegetation types include: Heathlands, *Banksia* woodlands, *Melaleuca quinquenervia* forests, mangrove forests, sedgelands, *Eucalyptus racemosa* and *E. pilularis* open forests and tall open forests.
Province 5 – Brisbane–Barambah Volcanics	Covers area of upper Brisbane Valley and parts of Barambah Creek catchment. Rolling hills and broad stream valleys are characteristic of this province. Rainfall is relatively low (800–1000 mm/year) and has extensive ironbark eucalypt woodland and araucarian microphyll rainforests. Has been extensively cleared
Province 6 – South Burnett	Contains the Bunya Mountains and is relatively elevated. Highest elevations of this province are linked to the Scenic Rim province in terms of flora and fauna. Major vegetation types includes: araucarian microphyll rainforest and eucalypt woodland and open forests.
Province 7 –Gympie Block	Covers the Sunshine Coast hinterland to north of Bundaberg, which contains low, hilly landscapes. The area is moist in the south (rainfall >1500 mm/year) but drier in the north (rainfall 900 mm/year), and the fertile soils contain broad patches of araucarian notophyll and microphyll rainforest and mixed eucalypt forests.
Province 8 – Burnett–Curtis Coastal Lowlands	Includes the Maryborough Basin. Drier than the Great Sandy and Southern Coastal Lowlands provinces to the south and

	contains a marked tropical biota component. Heathlands, *Melaleuca quinquenervia* open forests and eucalypt woodlands and open forests are the major vegetation types.
Province 9 – Great Sandy	Includes Fraser Island and area known as Cooloola, and contains the sandstone hills and riverine plains of the Noosa River catchment. Vegetation in this area includes: notophyll rainforest, *Lophostemon confertus–Syncarpia hillii* tall open forest, mixed eucalypt open forests, *Banksia* woodlands and *Melaleuca quinquenervia* woodlands.
Province 10 – Burnett–Curtis Hills and Ranges	More difficult to differentiate in geological terms. Includes the granite hills and ranges in the east and low rolling hills in the west, of which includes the Kroombit Tops plateau, a moist topographic isolate linked to Blacktown Tableland and the ranges of southern Queensland. Main vegetation types include: *Eucalyptus crebra* and *E. citridora* woodlands, eucalypt mixed open forests and Araucarian microphyll rainforests.

As the description and regional classification indicates, this is a very large area of diverse geological and morphological landscapes. This also translates into fire effects and fire regimes.

Due to the considerable size of the study area and diversity of biota occurring within the region, time constraints limit the detailed examination of fire regimes and its effects on all indices of biodiversity. Therefore, it was decided that determining the ecologically sustainable fire regimes for specific vegetation communities would provide the most suitable outcomes. This was mainly due to the knowledge that fire exerts the highest influence on the floral communities, which then directly affects the impact of fire on fauna and other processes such as hydrology. Table 2 (below) lists the broad vegetation communities in order of preference. The Research Working Group (who initiated the project) of the Fire and Biodiversity Consortium decided which vegetation types were deemed to be of highest significance in the project's infancy according to the following criteria:

- these areas occupy the largest area for all the shires and communities in southeast Queensland,
- are recognised as important vegetation communities (for biodiversity values) within the region,
- represented areas where least information was presumed to be known, and most importantly
- were areas where from inappropriate fire regimes poses the largest threat

Gill and Williams (1996) state that there are correlations between plant community structure and the abundance and richness of birds and small mammals, so that repeated burning which reduces the diversity of vegetation will also reduce the numbers and diversity of vertebrates.

According to the Queensland CRA/RFA Steering Committee (1998a), the state contains a rich native flora of 8655 known species. Further, within the local southeast Queensland bioregion alone there are 143 endangered and vulnerable plants species listed on the schedules according to the *Queensland Nature Conservation Act 1992* (Queensland CRA/RFA Steering Committee 1998b). Of the region's vertebrate fauna, nearly 36% have some distribution limit in the southeast, 3% are endemic and over 10% is endangered (Queensland CRA/RFA Steering Committee (1998b). Furthermore, Queensland CRA/RFA Steering Committee (1998b), stated that of all the species listed as endangered, vulnerable or rare in the *Queensland Nature Conservation (Wildlife) Regulation 1994*, 35% are reported in southeast Queensland, and forest-using species account for 60% of the total terrestrial vertebrates. These numbers highlight the rich diversity that is contained within southeast Queensland. It also serves to demonstrate that inappropriate fire regimes, amongst other threats such as weeds etc., can have a potentially dramatic impact on the environment, which may cause localised extinctions of fire-sensitive flora and fauna.

The lists of prioritised vegetation communities are as follows:

Table 2: Prioritised Vegetation Types for Research Project

NUMBER	PRIORITY	BROAD COMMUNITY TYPE	INCLUDES (WHERE APPLICABLE)
1	**High**	*Wet Sclerophyll Closed Forest*	
2	**High**	*Dry Eucalypt Open Forest with Grassy understorey*	
3	**High**	*Dry Eucalypt Open Forest with Shrubby understorey*	
4	**High**	*Melaleuca Forest*	
5	**High**	*Coastal Woodland–Open Forest*	
6	Medium	Subtropical Rainforest	
7	Medium	Dry Rainforest	
8	Medium	Mountain Heath	
9	Medium	Estuarine Complexes	*Casuarina glauca* Open Forest Mangrove Forests Saltmarsh
10	Medium	Wet Lowland Heath	Sedgelands
11	Medium	Dry Lowland Heath	
12	Low	Naturally Bare Areas	Rock faces Rocky Outcrops
13	Low	Cleared Areas	Regenerating Areas Open Grasslands

After further discussions with the Working Group it was also decided that the heathland vegetation community type had high biological importance, enough for comment and review. The other vegetation types are also very important to maintain and conserve, but time constraints did not permit an extensive examination of these vegetation types.

Table 3 (below) shows further evidence of the high species diversity that southeast Queensland contains. It also demonstrates that this diversity stretches across a wide variety of ecosystems and biophysically distinct regions. This is quite unique, especially for such a highly populated region like southeast Queensland. However, similar to most of the Australian continent, the effects of fires are not inhibited by these boundaries and will affect the processes within them to some extent. While the protection of life and property will always be the first priority, some consideration of the harmful effects of incorrectly imposed fire regimes on the region's biodiversity must also be noted and, if possible, mitigated.

Table 3: Distribution of centres of high taxa richness across provinces within southeast Queensland (adapted from QDEH 1998b)

Province	National Parks, State Forests, Timber Reserves contained in reference cells	Richness Value	% Forest Cover (range)	Notes
1. Scenic Rim (west)	Mt Barney	227/286	65–100	
Scenic Rim (east)	Lamington National Park, Springbrook National Park	259/270	87–98	
2. Moreton Basin	–	–	–	Dwyer's Scrub Conservation Park occurs in a cell, which has richness values just below the threshold set.
3. Southeast Hills and Ranges	Daisy Hill State Forest Venman Bushland National Park Conondale NP SF 788 FTY 1682, SF 272 FTY 1680	461 273	31	Cover is low in areas close to the metropolitan area, higher in the north–west within Brisbane Forest Park
	Toohey Forest	274	7	
	Brisbane Forest Park (SF 309 FTY1307 (Enogerra), SF 1355 FTY1526 (Dundas)	240/ 301	77–95	
4. Southern Coastal Lowlands	North Stradbroke Island (Blue Lake National Park) Noosa National Park, Mt Coolum National Park, SF 1239 FTY, 1255 (Kenilworth), SF 689 FTY 979 (Maroochy)	298/ 407/ 431/ 534/591 /	<46%	Low forest cover is characteristic for cells across the province
5. Brisbane–Barrambah Volcanics	Yarraman–Imbil districts	249/255	63–92	
	Mt Walsh NP SF1344 FTY 1534 (Boompa)	321/349	62–90	
6. South Burnett	–	–	–	–
7. Gympie Block	SF 639 FTY 502 (Wrattens) SF 298 FTY 1230	326 278		
8. Burnett–Curtis Coastal Lowlands	Deepwater National Park	298/314	70/71	
	Burrum Coast National Park	274	64	
9. Great Sandy	Cooloola National Park Great Sandy National Park	280/566	91/100	
10. Burnett–Curtis Hills and Ranges	Eurimbula National Park (Rodd's Peninsula)	324	59	
	Kroombit Tops	266/296/ 377/536	~97	

Introduction to the Format of this Report

This report consists of a critical review of an extensive amount of work completed within southeast Queensland and throughout Australia on fire and its associated effects on the landscape. It is envisaged that conclusions drawn from these studies will provide some idea on how to approach the difficult and extremely complex relationships between fire and biodiversity in southeast Queensland. The suggestions on specific fire regimes are included by the authors after examination of existing literature, but until tested using scientifically accurate methodologies with replication of sites, then they should be treated with caution.

At the initial stages of this project, indicator species (of in/appropriate) fire regimes were discussed and deemed to merit some form of comment and review. This report lists some potential indicator species to evaluate within each community vegetation type for the effects of imposed fire regimes, but these also merit rigorous scientific study on their effectiveness. Readers of this report will discover that the use of such species can be fraught with difficulties, but also may offer many advantages. There will certainly be some trade-offs between each methodology, and it is the onus of the authority to decide whether the protection of biodiversity within their jurisdiction warrants this type of resource-intensive study.

The report initially contains an introductory section on the generalised effects of fire on the landscape, followed by a section on the purposes and predicted outcomes of the Fire and Biodiversity Consortium (FABC) related to this review. The roles and purposes of the FABC is discussed in more detail below.

Following these sections, the report analyses the use of the term 'fire regime' and explores the definitions in an ecological perspective, in addition to the examination of fire regime from a management point of view. A detailed section follows examining the fire histories of the region and how the history of settlement (and their uses of fire) firstly by the Aboriginals and then from the Europeans, has altered the environment. An analysis of the prescribed burning management practices is the focus of the next section, where some well-documented examples of inappropriately implemented fire regimes have compromised the chances of survival of many native species.

After the introductory sections, there is a comprehensive examination of vegetation-types that were classed as 'High' Priority with particular focus on the effects on flora and fauna on their potential use as an indicator species. Gaps in the current knowledge are then identified and explained, and where possible, some ideas of projects to initiate which may be helpful in answering these questions. After the five broad community types are examined, a more detailed section on the use of indicator species is included where the advantages and disadvantages of this form of monitoring is thoroughly examined. Finally, there is a section on recommendations for the local southeast Queensland region, and concluding comments. Wherever possible, goals and outcomes from the review will be included to highlight the areas where agencies in the local region may be able to apply the recommendations.

Numerous articles were examined and reviewed for this report. Not all the information that were gathered from these articles are included in the report, even though they are included in the accompanying cd-rom database. The Reliability and Relevance rankings assigned to each article were based on the following generalised criteria:

- according to the source of the article (for example referred journals, non-referred journals, conference proceedings, workshops, or anecdotal evidence)
- scientifically-sound methodology,
- accurate use statistical analysis
- consideration of ecology in recommendations of fire-regimes,
- if the research was conducted in southeast Queensland, and
- if the origin of the work was conducted in other areas of Australia, how applicable is this work to southeast Queensland?

The ‘Reliability’ and ‘Relevance’ indicators are included with the accompanying cd-rom containing all the references used, collated and read during the project. These indicators are not used in this report.

THE FIRE AND BIODIVERSITY CONSORTIUM

The Fire and Biodiversity Consortium comprises of representatives of the local universities, state government agencies responsible for fire management in southeast Queensland, and environmental officers and planners from the local government shires, councils, and authorities and community groups. It is envisaged that by the conclusion of the two year project, a best-practice fire manual, standardised fire reporting system and demonstration sites would be set up and implemented. This reports fulfils the first objectives; identifying the areas where we have some knowledge or information on fire and biodiversity and to outline the gaps where work is required. The remainder of the project will focus on instigating research programs and/or monitoring sites in an effort to answer some of the questions outlined in this report.

Introduction

DATABASES

As part of the overall 'Fire and Biodiversity' project, a database of fire projects and monitoring sites within southeast Queensland will be established and periodically updated. It may be possible to use some existing databases as templates, and update (or alter) these with information relevant to southeast Queensland. Gill and Bradstock (1992) used an innovative initiative to collate plant responses to fire and created an Australia-wide reporting system for this type of information. At the inception of the fire-response database in 1992, there were approximately 1500 plant species collected, and at the end of 1999, over 3470 plant species have been included, with Queensland providing over 1000 records, accounting for just under 30% of the total records. This 30% is not exclusively plants endemic or restricted to Queensland, as the distribution of some plants may be continent-wide, so this number may be slightly exaggerated. Even though the information contained within the database is by no means a complete reference guide for a specific plant species, it is a worthy start for cataloguing this type of information for plant responses in southeast Queensland. Perusal of the most recent database shows that the majority of the work completed solely in Queensland (ie, not including information from other states or territories) shows most the information to be quite limited in extent and distribution. In particular, the information in the database relates to studies completed by researchers in Cooloola National Park (and surrounding area) and Beerwah (and surrounding area), the latter being the only long-term monitoring plot in southeast Queensland. These specific areas of local research are discussed in later sections.

Currently, the 'Monitoring' section of the Fire and Biodiversity Consortium (FABC) is working on a database to record and document this type of research, and will be discussed in more detail from reports produced by the FABC Coordinator. This database will include any study completed (or ongoing) that covers any aspect of fire ecology in southeast Queensland. Information will not be restricted to flora, but will also encompass faunal aspects of fire ecology. The database has been specifically designed to allow future potential researchers to determine the level and extent of previous work and to decide which aspects to pursue in further detail.

Fire Management and Biodiversity

MODELLING

Richards *et al*. (1999) attempted to model the optimal fire regime for a conservation park in South Australia for the sole purpose of maintaining biodiversity. This type of modelling requires knowledge of fire histories, fire intensities and management strategies employed by each park, (in this case, Ngarkat Conservation Park). Furthermore, the modelling was designed specifically to maintain the populations of four rare and threatened birds that are found within Ngarkat CP. This required specific information on each bird species' responses to fires, both wildfire and prescribed burns, recovery mechanisms, and habitat requirement. For birds (and many other vertebrate fauna, invertebrates and plants) in southeast Queensland, this information is not yet currently available. Using the mathematical model, the researchers found that by incorporating the (human) costs (predominantly economical) the 'best' strategy seemed to be to adopt "sub-optimal" strategies which did not involve active suppression of all wildfires (Richards *et al*. 1999). According to the authors, using this strategy the biodiversity would be accounted for, if the park was in several stages of ecological succession. To manage the Park using the optimal (maximum input) strategies – was considered far too costly, and if management strategies were aimed for the 'short-term' (within 5 or so years) then the 'cheaper' (economically) strategies were considered to be adequate (Richards *et al.* 1999). Notwithstanding, for this management-for-biodiversity strategy to be used in the local region will require many years of research to determine empirical information of fire response and habitat needs before a model of this type can be used. It is, however, an attainable goal.

FIRE REGIMES–WHAT ARE THEY?

According to Lang (1999) fire regimes can be classified according to frequency, intensity, season and fire extent (and the influence of these effects on flora and fauna). Asquith and Leishman (1999) defined *fire regime* as "the set of natural or induced fires that have occurred within a defined area over a given period, and takes into account the frequency of fires, intensities of individual fires, seasons of their occurrences, patchiness of their occurrences over the area, and time elapsed since last fire". Though used to describe what has occurred in the past, the term is also often used in prescribing a management goal to be achieved over a given period in the future. Survival of *fauna* is determined by these fire regime variables in the following order of importance:

Fire intensity → fire extent → fire frequency → season

The survival of fauna is usually depends on the effects of fire on flora, and the fire regime variables that affect flora range in the following order of importance:

Fire frequency → season → fire intensity (Adapted from Lang 1999)

The effects on both flora and fauna are strongly interrelated, with fauna highly dependent on surrounding flora for food and shelter (Lang 1999). Given the dependence of fauna on the flora and the pre-eminence of fire frequency in structuring flora, then of all the variables, fire frequency assumes overriding importance. Fire frequencies vary dramatically throughout the continent, as figure one illustrates.

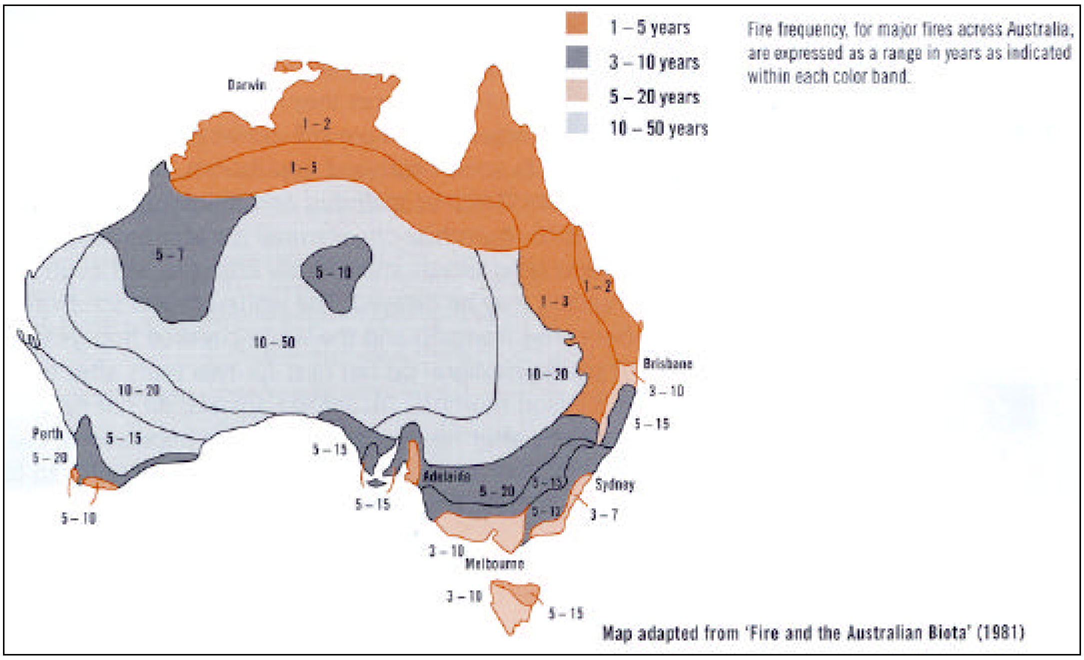

Figure 1: Variation in fire frequencies in Australia (reproduced from Lang 1999)

As the generalised Figure 1 shows, the southeast Queensland and northern New South Wales region experiences fire frequencies in the range 1 to 10 years. The fire frequency shown in Figure 1 is a record of the *major* regional fires and probably does not imply such a high frequency of burning of smaller portions of land containing a diversity of vegetation types. Despite the generalisation of the fire frequencies, it does illustrate the mosaic effect of fire across the continent (Lang 1999). The figure does provides a reliable indicator of the flammability of the local region and the need to correctly use fire as a land management tool. It also further emphasises the pre-eminent role of fire in shaping landscape processes.

FIRE AND THE AUSTRALIAN BIOTA

Fire and its effects on the Australian biota have been well-documented (e.g. Gill *et al.* 1981). However, in the majority of these reports the effect of only one fire on the landscape is examined, and as Williams and Gill (1995) state, consideration of the fire regime is necessary to better understand responses of species and assemblages of species. Further, Williams and Gill (1995) mention that the effects of repeated prescribed burning on forest ecosystems is still not well understood. It is well known that many floral species depend upon fire or its associated effects (such as heat and smoke) for regeneration (Whelan 1995). The 'drying' of the Australian continent after the Tertiary Period and the use of fire by pre-settlement Australians ensured that the flora and fauna were specifically adapted to fires, whether natural or human-induced (Lang 1999).

FIRE REGIMES AND BIODIVERSITY

Bradstock *et al*. (1995) showed (Table 4) that inappropriately implemented fire regimes may lead to decline in plant populations, especially declines in plant species density and diversity in coastal heaths (and associated shrublands and woodland), in a patch of 1 ha (or less) through these mechanisms (below):

Table 4: Plant declines resulting from inappropriate fire regimes in coastal heath, and associated shrublands and woodland (Bradstock *et al*. 1995)

A decline in populations of plant species can be expected when: • there are more than two consecutive fires less than 6–8 years apart (fire sensitive shrubs decline); • intervals between fires exceed 30 years (herbs and shrubs with short-lived individuals and seed-banks decline); • three or more consecutive fires occur at intervals of 15–30 years (sub-dominant herbs and shrubs decline); and • more than two consecutive fires occur which consume less than 8–10 tonnes ha-1 of surface fuel (species with heat-stimulated seed-banks in the soil decline)

Furthermore, the authors stipulate the need for flexible fire management (refer to figure below), to continually assess the fire regimes in an area/landscape and evaluate the biotic responses based on appropriate criteria (similar to those in Table 3). As Bradstock *et al*. (1995) state, if fire regimes and associated strategies are to be regarded as hypotheses based upon the best information available at the time, evaluation of these regimes (indicated as feedback loops in the figure) become a test for these hypotheses. Then, finally these outcomes result in the persistence *or* extinction of the particular species.

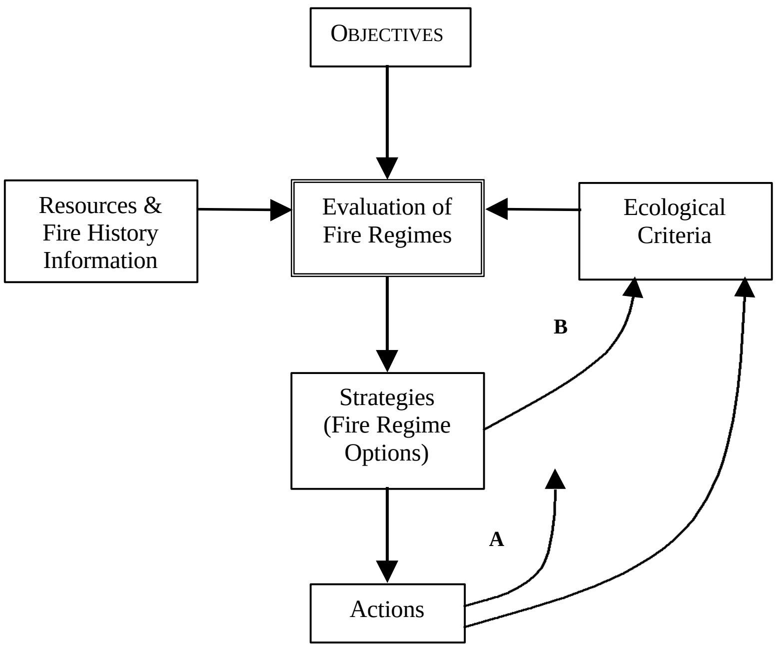

Figure 2: An outline of the way in which fire management (for conservation) may be planned in a flexible manner. The 'A' and 'B' pathways involve independent steps in assessment of the outcome, of planned actions. Path 'A' assesses how fire management actions influence fire regimes. Path 'B' assess how fire regimes affect biota and whether these effects are in accordance with predictions based on existing knowledge. (Modified from Bradstock *et al*. 1995)

As Bradstock *et al*. (1995) correctly mentions in reference to Figure 2, the obvious pitfall with this type of strategy is that feedback loops must encompass fire regimes (arrow A), and biotic responses (arrow B), with a temptation to do the former and not the latter. Furthermore knowledge of the relationships between fire regime 'patchiness' and the persistence of species is limited and warrants urgent attention (Gill and Bradstock 1995, stated in Bradstock *et al.* 1995). To conserve biodiversity and limit extinction of local species, one of the recommended strategies would be to maximise the variance in fire intervals (Gill and Bradstock 1995). The notion of returning the remaining bushland and surrounding forest to the 'normal' fire regime is no longer achievable (House 1995). This is due (in part) to widespread human occupation of the landscape, fragmentation of previously large tracts of forests and the permanent alteration of previous fire regimes through specific land management practices such as prescribed burning. As Gill and Bradstock (1995) mention, maximising variance is the opposite extreme (even though this may have been what has happened in the past). For the conservation of species, maximising variability would be achieved through time, not spatially (Gill and Bradstock 1995). The notion of treating only one block (of forest/reserve) is also not endorsed by Gill and Bradstock (1995). They cite the example, that burning one block may eliminate the entire food source of *Casuarina* for the Glossy Black Cockatoo residing in that area (say, a coastal woodland forest). Similar to other researchers, Gill and Bradstock (1995) recommend "patch" or mosaic burning, to create (within the reserve) different aged stands of forests to suit a variety of (rare/threatened) species. However, as Gill and Bradstock (1995) state "…if [a] portion of the reserve is to be burnt, which portion of the which species distribution, community, ecosystem or reserve area should be chosen? This question is unanswerable at the moment, but a general principle for plants … is that the reserve be managed for the most sensitive species and that not all the reserve be burned at any one time."

In northern Australia, a comprehensive experiment involving extensive sampling of 108 plots for fuel loads and many other variables in addition to satellite imagery of previous fires were used to determine appropriate fire regimes (Russell-Smith *et al*. 1998). The study revealed that for the heath-type vegetation in Kakadu National Park, a fire regime of five or more years is required (especially on harsher, rockier sites), though some sites burnt on a three-year rotation were tolerated without the loss of floristic diversity (Russell-Smith *et al*. 1998). However, in the sandstone-derived landforms of Kakadu NP, 40% of the vegetation burnt at frequencies of one–three years could not be sustained without substantial loss of obligate seeders, which comprised the majority (54%) of the sampled (heathy) flora (Russell-Smith *et al*. 1998). Obligate seeders could be defined as those plants which are normally killed by fire but release seed (prior to, or during the fire) as its process of regeneration. The authors concluded that the continuance of this contemporary fire regime would result in 'catastrophic' impact on the fire-sensitive flora in this part of northern Australia. This is the type of procedure that is required for our region. Accurate interpretation and examination of local fire histories has been documented from discussions with prominent researchers in southeast Queensland as being the highest research need. The work completed by Russell-Smith *et al*. (1998) may provide some useful methodologies for southeast Queensland. It may provide some answer to one of the larger gaps in the knowledge identified during the course of the study – fire history. This will be discussed further in another section.

Another poorly examined feature of fire regime is seasonality. The fire season throughout Australia varies enormously, dependent upon the local topographical and climatic factors. McLouglin (1998) compared the historical use of fire in the Sydney region with the current prescribed burning regimes. As McLoughlin (1998, page 393) states:

"Despite the significant body of research on the interaction between fires and Australian ecosystems, ecologists have barely started to understand the complex relationships between fire, ecosystems and ecosystem components over space and time, in part due to the range of variables involved. While knowledge is accumulating, changes in vegetation (and consequently habitat and dependent fauna) will take place in response to the prevailing fire regime. Fire management plans must be developed and hazard levels in forests and bushland adjacent to urban areas and rural properties controlled."

Further, McLoughlin (1998) reviewed previous studies in an effort to gather more information on season and the use of prescribed burning: the primary conclusions were:

- much of the work on fire and plant ecology has concentrated on the frequency component of fire regimes, limiting the amount of work on seasonality,
- autumn fires may damage resprouters, while not conferring benefits to species which reproduce from seed only. Resprouters are those plants which may be killed by fire, but the have below-ground parts which survive and are able to regenerate, post-fire. Indeed, some studies have shown that the amount and timing of germination relates to the timing of post-fire rainfall, rather than season of fire or season of rainfall, with great variation occurring between sites and among years,
- vegetation composition may be altered from burning in different seasons. For some areas, spring fires favour shrub regeneration, while autumn fires seem to favour herbaceous species,
- repeated burning experiments in spring and autumn are also compounded and affected by the intensity and (extent of) patchiness of the fire and post-fire climatic conditions. Two specific studies (one near Sydney, the other near Ballarat), for two successive fires prescribed burns in autumn and spring, there was little difference in ecological effects between the two seasons of burning, though some species (and groups) were more affected (both positively and negatively) by the season of burn. However, the second fire much reduced the species in the autumn plots (in Sydney) and recovery was much slower for both seasons at the Victorian site. This is more likely a result of fire frequencies and not the effect of fire season.
- food resources are critical for fauna to survive post–fire. Further, low intensity burning (by inducing a simplification of the understorey) may have significant long-term impacts on fauna. Since animals show a complex range of responses to fire, life history characteristics will vary dramatically. For example, it is still unknown if a fire is more detrimental in autumn where males (rodents) may not yet have bred, or in spring before males are weaned,
- some evidence of the effect of fire season may be elucidated from the effects of ntensity, as a result of fuel, season and weather conditions. Some species require a certain amount of heating before seed coats are opened, and low intensity burns may not provide adequate heating. Furthermore, intensity and season of burning may be a critical factor in the impact of fire on fauna in moister areas, especially post-fire. Autumn burning would mean gully vegetation is more likely to burn (in Victoria) – reducing refuge areas and affect habitat diversity. This is unlikely to be the case for Queensland, where Spring is more likely to be the time where gullies are more inclined to burn,
- the effect of season of fire on invertebrates varies, with some studies finding that the impact of low intensity burning on soil invertebrates was greater in spring than in autumn for a dry sclerophyll forest in Victoria. Other studies have found that winter burning had

greater impacts than spring (in the ACT), with no difference found between autumn/spring burning for another dry sclerophyll forest in Victoria. The impacts and utility of invertebrates with respect to fire has been subject of a considerable amount of work and will be discussed in the Indicator Species section.

Refer to McLoughlin (1998) for further information on these studies.

Historically, the majority of fires lit (by the Aborigines) in the Sydney region were largely confined to the spring–early summer fire season (August–January). However, over 60% of prescribed burning implemented by authorities such as the NSW National Parks and Wildlife Service have been conducted in autumn–winter (April–July) (McLoughlin 1998). For southeast Queensland, most of the burning still occurs in the autumn/winter period due largely to 'controllability' of the fire, availability of personnel, and weather conditions (Hall and Gourley, *pers. comms*.). However, there has been acknowledgment from many local authorities such as the Department of Primary Industries, and Department of Natural Resources of the desirability to implement more 'summer-burning' practices mainly for ecological purposes (Gourley *pers. comms*). Even in places where some considerable work has been conducted, there are still some very basic ecological questions that have not been answered and much work is still to be done.

Christensen and Abbott (1989) compiled a comprehensive review of literature on the effects of fire on soils, nutrient cycling, microorganisms, vascular flora, soil and litter invertebrates, other invertebrates, reptiles, amphibians, birds, mammals and exotic plant species in jarrah (*Eucalyptus marginata*) and karri (*Eucalyptus diversicolor*) forests of Western Australia. A review of pre–European fire regimes suggested that a mosaic-burning pattern of frequent fires occurred under relatively mild conditions. Further, the overall response of the flora and fauna is:

1. reduction in density and sometimes number of species immediately after fire,
2. recovery in density and number of species after fire, usually from propagules within burnt areas, and
3. transient changes in relative density of species after fire.

(Christensen and Abbott (1989)

Importantly it is stated that the rate of recovery of fauna is mostly dependent on the vascular flora, where each floral species has a well–defined response to fire. Furthermore as Christensen and Abbott (1989) state "*there is very little information on the long-term effects of repeated fires on the flora and fauna*". The authors conclude by presenting a set of proposals to guide fire management in forest and adjacent woodland areas, where there are specific areas set aside for conservation purposes.

The other states in Australia have a decided advantage over Queensland, in that, they have initiated and examined many long-term experimental plots and have adequate knowledge of the fire histories of a particular region. Lunt (1997) examined the differences in a grassy-woodland with differing fire regimes (from rarely burnt to frequently burnt, possibly annually). With such varying fire regimes, there were distinct differences in the vegetation composition between sites, and as Lunt (1997) states, the imposition of divergent management regimes in different sites has led to an ecological segregation of native species according to their tolerance to prevailing management. The results showed that the rarely burnt forests containing significantly more species than frequently burnt sites, and four of the five frequently burnt sites were treeless (Lunt 1997). Interestingly, the species composition of the frequently burnt plots were not simply a subset of species of the rarely burnt plots (which might be expected), but supported many native species uncommon or absent from the rarely burnt plots (Lunt 1997). Notwithstanding, the frequently burnt sites still make an important contribution to the regional diversity, and show the need for a diversity of ecosystems to conserve species diversity at regional levels (Lunt 1997).

FIRE AND BIODIVERSITY IN SOUTHEAST QUEENSLAND

Drake (1998) considered the dilemma of limited resources and large gaps in knowledge for the local region. Even though, for some areas (such as Beerwah and Cooloola National Park) where a good amount of work has been conducted, it has taken some time before any management decisions have been made. Drake (1998) pointed out that subsequent work by McFarland (1988a and b – see Fire and Avifauna section, page 67) ten years previously, only recently there been any work commenced again in the area. In relation to decision-makers (local shires, councils and authorities etc.) the expected immediacy of results is unable to be met, and requires many years before an adequate amount of data is available to assist in these decisions. Coupled with the lack of fire-research in southeast Queensland, it will require several years of dedicated research and establishment of appropriate monitoring sites (of varying levels) before any ecologically reliable decisions are made. House (1995) reiterated this by stating, "…(there is a strong) need for long-term studies that monitor change over a *number* of fire intervals." Furthermore, as House (1995) states, the only site where long-term fire ecology experiments currently exist is the Beerwah Experimental Plots. The Dept. Primary Industries (Forest Research) coordinates and controls the research conducted here, but the plots are now controlled by the Dept. of Natural Resources. Recent personal communication with Dr Alan House, elucidated the following information about the sites at Beerwah, some of which is summarised below:

BEERWAH LONG TERM EXPERIMENTAL PLOT

- **Site History**: The only real long-term plot in south east Queensland, replicates in other areas relatively close to Beerwah
- **Fire Regime**: 3 sites: 3–yr burnt, 5–yr burnt, and unburnt (since 1972)
- **Methodology**: variables measured include cover and abundance, and since 1972 grass/sedges have not been separated into distinct categories
- **Results**: Regeneration of native plants have been recorded, along with cover and abundance measures. More variables can be measured, but lack available resources to do so.

More specifically, these additional sites used in conjunction to Beerwah have the following characteristics:

At the **Baupal Site** – a monitoring site since 1952.

- **Fire Regime**: There are three sites: annually burnt, 2–3 yrs burnt, and unburnt (since 1952). Site size of unburnt plot is 200-300 ha
- **Site Characteristics**: Climatic conditions: ≈ 1000 mm annual rainfall, with Spotted Gum/Coastal forest and Dry Scrubland
- **Results**: In the unburnt site (unburnt since 1946–1952), *Eucalyptus* recruitment is unchanged from burnt areas, with the ground layer consistent with that of a weedy flora, including lantana and passion-vine
- **Impacts of Regime**: Most interesting result is the unburnt site *invertebrate indicators* show a community characteristic of a *disturbed* environment, due to the *exclusion* of fire

Wallum Site (Scientific Area#24) – this is close to Scientific Area#1

- **Fire Regime**: This site has been unburnt since the 1920s
- **Site Characteristics**: Fuel loads of 18/20 T/Ha
- **Other notable features**: Has wet-sclerophyll vegetation that are obligate seeders, eg. *Eucalyptus grandis*, *E. pilularis*. Also has *E. conglomerata* (Swamp Stringybark) within study site, plus many other rare species. The swamp stringybark responses to fire are unknown

- **Conclusions**: The uniqueness of this site (no fire for 80+ years) coupled with the rare *Eucalyptus conglomerata* should warrant further study in this area

As the information above shows, from this (relatively) small area within southeast Queensland, a decent amount of work has been completed in vegetation types listed as high priority. Further discussions with Alan House showed that even though they are entrusted with the maintenance of the site and conduct the burning at prescribed intervals, there is not enough available personnel to perform any high-level analysis of the fire effects other than regeneration, cover and abundance. Here exists an avenue of research in southeast Queensland that remains under-utilised. The summary of results above are just a small indication of what could be achieved if more resources were directed to allow more specific assessments of these fire regimes. The lack of any scientifically rigorous work in southeast Queensland was recognised many years ago, which is discussed below.

Sandercoe (1990) highlighted the need for more work in the southeast Queensland region, by stating that for the Greater Brisbane region, over 40% of the parks are burnt frequently by wildfires, and that most of these areas are also regularly prescribed burnt. The ecological effects of this regime were probably not evident at the time. Sandercoe (1990) also reported (for Cooloola National Park) that there was a great lack of information on the extent and location of wildfire and prescribed burns. This remains true even now, years after this statement was written. The lack of accurate fire histories for many areas within this region is one of the major gaps in current knowledge.

FIRE–HEATING EFFECTS ON SOIL AND GERMINATION

After fires of moderate to high intensity engulf an area, the plant litter is completely cleared leaving bare soil (Johnson 1991). This exposes the soil to increased risks of erosion, especially if sufficient amounts of rainfall occur after the fire. Therefore, apart from reducing the acidity of the soil (Johnson 1991), fire can accelerate soil erosion and increase runoff. Prosser and Williams (1998) showed that following the wildfires in New South Wales in 1994, runoff increased, enhancing sediment transport, as a result of the reduced ground cover, but also revealed that this runoff was very localised and that large amounts of rainfall are required to generate substantial runoff.

The duff layer (also known as humus) is the layer of decomposing organic matter situated below the fresh litter layer and above the mineral soil, and plays an essential role in forest ecosystems in protecting the mineral soils from erosion, retaining moisture, and releasing nutrients (Valette *et al*. 1994). Valette *et al.* (1994) note that during forest fires, if the duff layer is not burnt it can protect the soil from high temperature exposure. Fire management practices are being used globally to make 'fuel-breaks' that lessen the ignitable fuel load in an area. But if prescribed burning is to be used effectively as a management tool to lower fuel amounts, to reduce the risks of wildfires, or to improve seedling regeneration, the (insulating) role of duff needs to be known. Once the effects of fire on the duff layer are known, this will minimise excessive consumption of organic matter during prescription burning (Valette *et al*. 1994). The results showed that with low intensity fires (25–35 kWm^{-1}) the presence of a duff layer was sufficient to lead to a reduction in the temperature at the soil surface of at least 330°C. For more intense fires (>50 kWm^{-1}) that produced longer-lasting surface heating, duff thickness and moisture content played an important role in significantly reducing the temperature rise at the soil surface, but the insulating effect was not as pronounced as with low intensity fires (Valette *et al.* 1994). Heating of plant tissues to about 60°C leads to plant mortality, and even with the low intensity fires experienced in this experiment, it showed that the duff layer did not insulate the plants (or shoots) adequately (Valette *et al*. 1994). Prescribed burning methods (of intensities of 25-50 kW/m, as in this experiment) can cause mortality of roots and soil biota (Valette *et al*. 1994). Concomitantly, Valette *et al*. (1994) reported that the temperatures recorded in the duff could produce water repellency, increased erosion and some delay in regeneration. Therefore in low intensity fires, the presence of a duff layer will reduce the temperatures, but can cause a delay in regeneration of the treated areas, and other detrimental effects.

Depending on fuel loads, fires can destroy the duff layer. Valette *et al*. (1994) showed that fire intensities of 40-50 kW/m (regularly used in prescribed burning) destroys the duff layer. Bradstock and Auld (1995) explained that with the use of prescribed burning, soil temperatures may be insufficient (ie low) which may result in the seeds of some plants not germinating. In leguminous species, heat breaks innate seed dormancy by mechanical alteration of the seed-coat, thereby allowing germination (Bradstock and Auld 1995). Auld and O'Connell (1991) found that for all the 35 species of Leguminosae (Fabaceae) from the Sydney region, there needed to be an exposure to temperatures above 80°C for seed germination. Prescribed burning does not provide this temperature exposure and will therefore not provide the mechanism to break seed dormancy. Sadler (1993) explained that some species of Eucalypts requires a high intensity fire at some stage of its life cycle to survive. High intensity fires open the canopy, eliminate competing species for resources and liberate nutrients for recycling, and the presence of large-seed banks, lignotubers and other regenerative components allow eucalypts to flourish (Morrison and Cary 1994, Mutch 1970). Griffin and Friedel (1984) demonstrated that fire is essential to initiate mass germination in *Acacia aneura*, *Eremophila gilesii*, and *Cassia* spp. even though they may be killed during the fire. Since prescribed burning techniques incorporate low intensity 'controlled' fire regimes, many indigenous species of plants will not regenerate after the fire, permanently altering the landscape. The relationship between soil temperature and fire intensity is depicted in Figure 3.

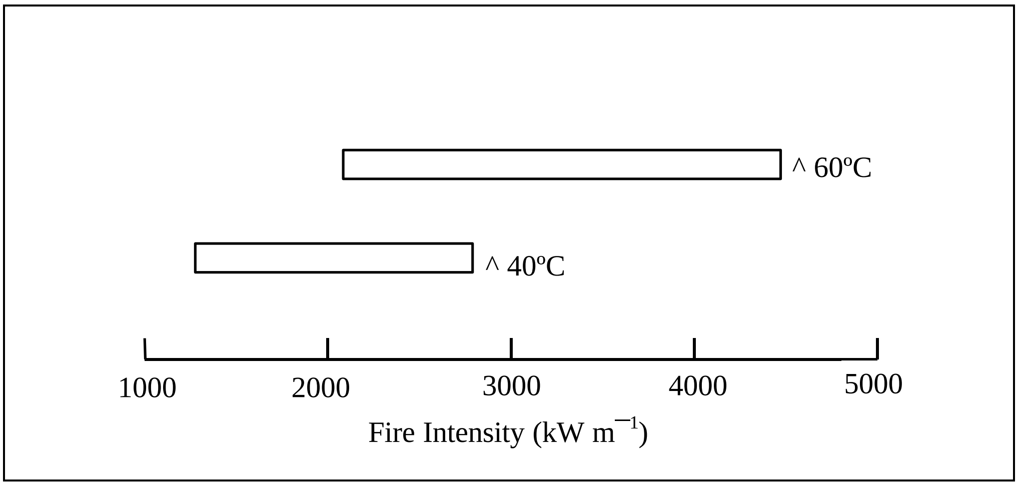

Figure 3: Ranges of fire intensities needed to achieve maximum temperature changes (^) of 40°C and 60°C in the upper 3 cm of the soil. Adapted from Bradstock and Auld (1995)

Therefore low intensity fires do not produce the required temperature changes that will germinate leguminous shrubs, and Bradstock and Auld (1995) emphasise that fire managers should allow for fuel accumulation that will facilitate propagation of leguminous shrubs. Leguminous plants are vital components of a plant community, since they are primarily responsible for nitrogen-fixation in the soil. This is demonstrated in Figure 4, where there must be fire intensities of 2000 to 4500 kW/m to achieve a change in temperatures of 60°C which would not be achieved with present prescribed burning methods. Adams *et al. (*1994) demonstrated that the effects of fire on heathland soil in Wilsons Promontory (Victoria) were confined to just 2 cm below the surface. Also in repeatedly burnt heathlands (one fire every 5 to 6 years) they found that phosphatase activity and concentrations of carbon, nitrogen, and potentially mineralisable N were less in soils when compared to unburnt heathlands (Adams *et al*. 1994). There is more discussion on prescribed burning and its effects on the Australian biota on page 27.

Fire Regimes and Variability

Morrison and Renwick (2000) investigated the effects of fire intensity on the regeneration of co-occurring species of plants in the Sydney region, in an effort to explain the possible differences of the effects of variability of fire events on plant response. It was stated that most previous studies had focussed on only one species, and it was inappropriate to suggest community-wide management strategies based upon fire response of one species. Furthermore, there is little known about variability among individuals within a species in terms of response to fire characteristics such as intensity, frequency and season (Morrison and Renwick 2000). As they state, one of the main sources of variability in fire intensity (along with fuel loads, fuel type, topography, and climatic conditions) is whether it is a prescribed fire or a wildfire. The results of the study showed that plants with high fire tolerances, such as the possession of insulating bark and epicormic or lignotuberous buds had increased survival rates over other plants and that stem diameters were correlated with plant survival after fire (Morrison and Renwick 2000). As the authors state "…if new individuals of these small tree species are not recruited to the populations, but the adult individuals continue to survive prescribed fires, then the populations will eventually become senescent." It was discussed that the present prescribed fire regime has a 'significant impact on relative abundances of these [plant] species'. Furthermore, the prescription of an 'invariant' fire regime will favour a particular subset of species within the community … inevitably leading to the diminution of the local biodiversity (Morrison and Renwick 2000).

Work completed by Morrison *et al.* (1996) emphasised the importance for variations in fire intervals. They showed that inter-fire intervals of decreasing length were associated with decrease in the "evenness of the fire-sensitive species, particularly large Proteaceous shrubs which often dominate the community biomass in dry sclerophyll shrublands". Furthermore, "increasing variability of the length of the inter-fire intervals is associated with an increase in the species richness of both fire-sensitive and fire-toilinet species". The authors stated that these results implied that it may be variation of the fire intervals that is primarily responsible for maintaining the presence of a wide variety of plant species in a particular community (Morrison *et al*. 1996), ie maintaining biodiversity. For southeast Queensland, Watson (1999) examined the effects of different fire frequencies and time-since-fire in a dry-sclerophyll region of Girraween National Park. The results of the study showed some variations in response to other research completed in other areas of Australia namely in the Sydney region. Interestingly, Watson (1999) found that there was no substantial variation in the (floral) biodiversity of an area with a 28–year and 20–year inter-fire interval respectively, suggesting that inter-fire intervals above 15–20 years may not be detrimental, but possibly necessary to maintenance of overall biodiversity. However, this would vary quite dramatically according to each area of interest. The important point to note from Watson (1999) is that much like other work such as Woinarski (1999) — that long intervals may be necessary for some species to survive and that variability is critically important to conserve biodiversity.

It is recognised that most of the work completed on fire regimes and variability originates from New South Wales and Western Australia, and that the conclusions from those research studies are only slightly relevant to southeast Queensland. However, the general principle of increase or maximising the variability of fire intervals to maintain a reasonable (ie-controllable) fire-risk and also to maintain species diversity remains very relevant. In recent discussions with land managers and representatives from the various local authorities and state agencies also revealed that prescribing a variation in fire frequencies is considered important to maintain some form of ecological mosaic for the conservation of the diversity. While these variations in fire intervals were not formally specified, other mitigating influences such as prevailing climatic conditions and availability of personnel (to conduct prescribed burns) has resulted in some degree of variation as an *ad-hoc* outcome.

Fire History

Fire has probably been a part of natural ecosystems since the origin of flora on the surface of the earth (Wright and Bailey 1982). Lightning strikes were probably the main cause of fire before human activity contributed an additional major source of ignition. Attiwill (1994) reports that other sources of fire are spontaneous combustion and volcanic activity. The potential for fire severity depends on the fuel source, the quantity and continuity of fuel, the fuel moisture, and the existing weather (Wright and Bailey 1982).

Wright and Bailey (1982) explain that dry lightning storms in continuous fuels with gentle rolling topography are most likely to cause large fires if winds and temperatures are high and relative humidity is low. The fire disasters in Victoria in 1983, and New South Wales in 1994, can attest to this. To lessen the impacts of fire, different methods have been applied, including (i) total fire suppression, (ii) prescribed burning to reduce fuel loads, and (iii) the establishment of firebreaks to stop fire spreading to adjoining countryside and/or properties. Attiwill (1994a) reported Mediterranean-type vegetation, in Europe, has had a long history of fire dependence. Fire at frequencies of 10-25 years or 20-50 years are fundamental to the maintenance of composition and structure of these shrublands and heathlands, and these ecosystems regenerate quickly after fire (Attiwill 1994a). Moreover, this regular fire frequency is necessary for the heathlands to regenerate and eliminate encroachment of more floristically open sedgeland flora (Gill and Groves 1981). Doubts about the desirability of fire in European Mediterranean vegetation have arisen from the destructive combination of overgrazing and land abuse with frequent burning and wildland fires, and Attiwill (1994a) regards this combination of factors as a wholly condemnable, human-made element.

Specific (and accurate) knowledge of exact fire histories for a certain region has been shown to be an effective measure of the ecological validity of fire regimes. This conclusion has resulted from many discussions with prominent fire ecologists from here and other regions of Australia. It seems that the long term fire histories for much of southeast Queensland are very poor and/or exaggerated. However, for some areas of southeast Queensland, particularly in National Parks, and especially those on the Scenic Rim there is some well-documented and mapped fire histories. Nevertheless, documented experience in southeast Queensland represents a major gap in current knowledge. Replicating the work completed by Russell-Smith *et al.* (1998) in Northern Australia here in southeast Queensland would allows us to take a considerable step in answering the question of knowing the fire histories for our region. Kitchin and Reid (1999) outlined a possible method of retrieving information about the fire history of an area using satellite imagery and local information on past fires. They studied a large portion of Guys Fawkes River National Park and incorporated satellite photos and fire-incidence reports dating back to the early 1960s. The results were highly accurate in estimating the fire histories in the mapped sites. However, it was also discussed that steep and rugged terrain was much more difficult to pinpoint, as the satellites seemed to 'miss' these areas in the analysis (Kitchin and Reid 1999). This methodology could be easily applied to many areas within southeast Queensland, particularly for National Parks, where there is a good system of fire reporting. Many other areas could also be mapped for their fire histories using this methodology. The importance of an accurate idea of the fire history is outlined in more detail in the next section. The adaptation of the Australian landscape to use fire is unique and specific, and only shows that fires are an unavoidable and necessary impact on the environment.

EVOLUTIONARY HISTORY OF FIRE IN AUSTRALIA

Fire has exerted a dominant influence on the Australian landscape, whether artificially or naturally induced. In Australia, there is a marked and significant increase in charcoal in lake deposits dating from 120,000 years ago (Attiwill 1994a). Fire has played a role in shaping the landscape, probably since before the first permanent human settlement by the Aborigines. Any noticeable effects of the Aborigines date from approximately 40–50,000 years ago. The Australian sclerophyll flora is dominated by *Eucalyptus*, which is highly fire prone, due mainly to the high oil content in the leaves

(Rudolph 1993). The Aborigines, by the use of fire over many millennia, have permanently altered the biota of Australia (Pyne 1991). Even after European settlement, only the forces of drought have had a larger influence than fire on settlement patterns and agricultural practices. Drought increases the prospects of large-scale bushfires, since the reduced rainfall significantly increases the amount of ignitable fuel present, and fires will inevitably occur, providing the ingredients for major wildfires in Eastern Australia.

Australia's forests developed their modern vegetation character over the period 60 to 20 million years ago (mya) (through the Tertiary and Quaternary periods), and for most of that time southern and central Australia was covered by rainforest (Clark 1981). About 20 mya, patches of grassland appeared in central Australia, suggesting that the climate was becoming drier (especially in the inland), and fires probably occurred more frequently (Clark 1981). Grasses appeared in the understorey only 4 mya (Clark 1981) which probably increased the fire frequencies. The progressive drying of the continent associated with the change in vegetation structure, (and to a lesser extent) the use of periodic fires by the first indigenous settlers and then from Europeans had permanently altered the Australian environment. Some examples of fire-adapted vegetation include the familiar gum trees, stringybarks, and boxes (*Eucalyptus* spp.), bottlebrushes (*Callistemon* spp.), paperbarks (*Maleleuca* spp.), tea-trees (*Leptospermum* spp.) are just some examples from the Myrtaceae family. Other fire-adapted genera include *Banksia* spp. from the Proteaceae family and wattles (*Acacia* spp.) from the Mimosaceae family (Clark 1981). The Myrtaceous family have plants with fleshy-fruited genera which are concentrated in South and Central America, whereas the dry-fruited genera are concentrated in Australasia (Pyne 1991). Within Australasia, the family Myrtaceae features ninety-five genera, ninety three of which are endemic, and in Australia, there are sixty-nine genera, of which forty-five are endemic, some of which are listed above (Pyne 1991). As the discussions show, there is a close association between fire and eucalypts, which is explored further in the next section.

FIRE AND *EUCALYPTUS* IN AUSTRALIA

Pyne (1991) stated that even though it is uncertain when the first eucalypt emerged from the rainforest flora, what is 'incontestable' is the degree that the genus *Eucalyptus* is endemic to Australia, and how it came to dominate the forest and woodlands environments of Australia. Pyne (1991) explains that this has come about from the alliance of eucalypts with both fire and with humans (Aborigines). Eucalypts have developed extensive, deep roots, capable of feeding widely, and in dry Australian climates, this is a definite advantage (Pyne 1991). In addition to this, eucalypts have evolved chemical and biological aids to improve access to nutrient reservoirs, particularly phosphorus, e.g. their alliance with soil microbes and mycorrhizae that evidently improves phosphorus intake and ensures eucalypts can grow where other trees starve (Pyne 1991). As well as an effective nutrient gathering processes, eucalypts have also developed storage mechanisms (such as lignotubers) which provide essential nutrients in very harsh times. Pyne (1991) explains that when the environment becomes chronically arid, eucalypts surrender to grasses, scleromorphic shrubs like saltbush (*Atriplex* spp.), and *Acacia* spp.

The 'partnership' between eucalypts and fire is a dominant landscape process in Australia The spread of *Eucalyptus* traces the spread of fire (Pyne 1991). Pollen and charcoal records are parallel throughout the late Pleistocene and Holocene periods (Pyne 1991). The Australian bush owes its peculiarity to *Eucalyptus* and no other continental forest or woodland is so dominated by a single genus (Pyne 1991). Catling (1994) summarised the (general) effects of a high intensity fire on Australia's sclerophyll forests, which is shown in Table 5.

Table 5: Recovery of sclerophyll forest after high intensity fire

Time after fire	The Year Post-Fire	Years 2-4	Years 5-15	Years >15
Canopy Cover	Canopy removed or scorched	Canopy open	Canopy closing	Canopy near maximum
Understorey Cover	Shrubs removed	Good shrub cover but not height	Shrubs taller	Shrubs at maximum but decreasing
Ground Cover	Litter and ground vegetation removed	Ground vegetation thick	Ground vegetation dense but thinning	Litter increasing; ground cover decreasing

These recovery processes will vary according to location, and it should be noted that in some areas, such as the wet-sclerophyll forests of the Greater Brisbane area, the recovery processes would be very different to Table 5. However, Table 5 does shows some generalised effects in the amount and type of cover following fire, and will vary to some extent depending upon the area, even though the successional paths are quite similar. What can also be concluded is that in most areas which have 15+ years of no fire, will contain a considerable fire risk ... and possibly create an environment detrimental to species diversity. The table also shows that in the event of fire (of medium–high) intensity, there is a remarkable and fast recovery of sclerophyll woodlands. This, of course, would vary according to the location of the site. The historical records certainly seem to suggest that the Aboriginals knew much about the fire ecology aspects of the landscape. Whilst the main purpose of burning (on quite regular frequencies) was for hunting purposes (Pyne 1991), it was their long history of knowledge which prevented the Aboriginals from over-burning the environment, causing decline is diversity, and ultimately, their food source. The effects of the Aboriginals and their use of fire on the Australian landscape is discussed further in the next section.

FIRE AND THE ABORIGINES

Even though there is some indirect evidence which suggests human occupation of Australia around 120–150,000 years ago, it was not until 40–50,000 years ago that humans had noticeable impact on the Australian landscape. Aborigines immigrated from mainland Asia to Australia via land bridges, and possibly using canoes, at a glacial minimum in sea level, and by 20,000 years ago they had colonised the perimeter of Australia, with only the most arid core, small offshore islands, and the higher mountains of the southeast, not fully settled (Pyne 1991). About 14,000 years ago as sea levels rose, Australia became an isolated continent, when the seas separated Australia from Papua New Guinea (Pyne 1991). The author reports that with the arrival of the Aborigines, there was an unprecedented wave of burning that reinforced, if not catalysed, the internal revolution within the scleroforest that assured the dominance of *Eucalyptus*.

The reasons for Aboriginal use of fire (Pyne 1991) can be summarised as:

(i) The use of fire permitted enhanced hunting techniques, since fires would clear certain areas, and expose or force animals within that area into open land, where they could be easily hunted down. Fire was the most effective weapon used in hunting.

(ii) Since Aborigines were nomadic, and periodically moved from one area to another, the use of fire assisted in clearing areas of bushlands otherwise impassable. Plus it provided a trail that could be used annually. As Pyne (1991) stated it was far simpler for the natives to keep an existing fire going than to start a new one. By perpetuating the fire as one tribe moved, the flora and fauna needed to adapt to this new increased fire regime, where the periods between fire were much shortened. If the tribe kept fire moving, then the food supply for them would also continue.

(iii) Campsites were regularly burned so that if a fire would approach, then the inhabitants would be safe because the fuel would have already been consumed.

(iv) By putting a fire through an area, food was readily available, but this also encouraged new plant growth (mainly grasses and other annuals) which would attract more grazing animals (such as kangaroos and wallabies) to the area, which could then be hunted.

(v) Fires were lit as a signal to neighbouring tribes and clans of their territory.

(vi) Plants such as Bracken (*Pteridium*) an aggressive fire weed, provided a palatable root, so the Aboriginals would 'till' the land with their fire-sticks, and collect the roots of such plants, including cycads, and yams.

As this short summary shows, the Aboriginals extensively used fire as an advanced hunting tool. Coupled with their nomadic lifestyle and the progressive drying of the continent, this inadvertently created or sufficiently altered the landscape to the one that we have today … a fire-adapted and dependent environment. The overall effect of the Aboriginals use of fire in shifting the environment to what we have today is probably not as influential as the various hypotheses that Flannery (1995) suggests, but there is no argument that they did have an influential effect. The arrival of Europeans in the late 1800s certainly marked a change in the thinking about fire and how it affected the environment. The next section discussed this subject in greater detail.

FIRE AND EUROPEAN SETTLEMENT

After the first settlement of Europeans in the late 18th Century, fire has always been thought of as a danger. This was due mainly from the effects of fire experienced back in England. Total fire suppression was the recommended action to stop fire from damaging valuable lands (Pyne 1991). This proved totally inadequate since the flora had already evolved to a fire-prone and dependent environment that was emphasised with the actions of the Aborigines. In contrast to the attitudes of the indigenous people, who used fire as an effective tool, Europeans (especially when settlement intensified) thought of fire as a threat to the land, land improvements and homes (Lynch 1981). Fire suppression became a preoccupation and considerable efforts were made to fight both natural and artificial fires (Lynch 1981). European settlement of Australia subsequently decreased the aboriginal population drastically, and with the extension of agricultural settlement, fire incidence decreased for a significant period (McArthur 1970). It was soon realised that prescribed burning was required to stimulate plant growth in the arid interior for the increase in domesticated animals, particularly sheep (Griffin 1981).

Despite this, the advocation of fire suppression strategies lead to quite damaging fires in Victoria in 1851 (Black Thursday) where ten settlers and hundreds of thousands of sheep perished (Pyne 1991). The first serious fire-storm that affected the European settlers occurred in 1939 on the 13th of January (Black Friday) (Pyne 1991). The weather conditions provided the basis for such a maelstrom of fire: 35+ºC, prolonged drought, dry winds, and relative humidity below 10% and to compound this, the fire suppression strategy (enforced mainly by the foresters to keep fire damage in their plantations to a minimum) made for large fuel loads (Pyne 1991). Subsequently, a 'fire triangle' stretching from South Australia to Tasmania to the Australian Capital Territory developed, with the worst conflagrations occurring in the Victorian mountains (Pyne 1991). After the fire, sixty-one lives had been lost, and countless numbers of horses, sheep, cattle, buildings, and invaluable forest plantations. There have been many other destructive fires in Australia, but the disaster in 1939 emphasised the need to revise how to deal with fire. Consequently, the use of prescribed burning is increasingly becoming commonplace, however, they cannot be used to achieve a predictable outcome due to the low level of understanding of the effects of fire (Hengst and Dawson 1994) especially fire of high frequency and low fire intensity. There are many other effects of prescribed burning and these are discussed in more detail below.

Prescribed Burning

THE ART OF HAZARD REDUCTION

Brown and Davies (1973) classified the purpose of hazard reduction as:

1. Removal of all ignitable fuels in limited areas of 'special' risk.
2. Removal of all fuel in a strip close to or around the source of risk in order to confine any fire that may be ignited to a small isolated area. Cleanup of fuels or exposure of soil to create firebreaks is a familiar example.
3. Removal of fuel in a strip where the purpose is to exclude fire from a high-value or high-hazard area. This includes firebreaks around forest plantations.
4. Removal of fuels to reinforce natural breaks and to create new ones by which an area can be broken up into blocks to facilitate control of wildfires. This provides a system of accesses and firebreaks to fight the fires.
5. Use of prescribed burning, when coarse and intermediate fuels are moist, to safely remove flash fuels from considerable areas. This will reduce the energy output and the rate of fire spread so control is much easier and less damage is done. This is regularly practiced in Australia.
6. Breaking the vertical continuity of fuels and the horizontal continuity of tree crowns with measures such as pruning and thinning and removal of undergrowth. This includes the removal of dead snags or trees which would create firebrands if ignited.

The summary above shows and as name suggests, the sole purpose of this strategy is purely to reduce the hazard to one that is considered manageable. As Pyne (1991) stated:

> "Current fire-management practices in the fire-prone vegetation of south-eastern Australia are based mainly on the concept of hazard reduction by the use of periodic low-intensity fires to maintain the amount of flammable fuel within specified (low) limits (Pyne 1991)" (quoted in Morrison and Renwick 2000).

These prescribed fires are thus intended to reduce the hazards associated with subsequent non-prescribed fires (wildfires) in the managed area (Gill *et al.* 1981). These fire-management practices are not based on concepts such as maximising or even maintaining species biodiversity, in spite of the fact that these practices can be expected to have a significant effect on the continued long-term survival of plants in the managed area (Whelan and Muston 1991; Williams *et al.* 1994; Morrison *et al.* 1996). Moreover, if a single invariant fire regime is prescribed for an area, then it may favour a subset of the species rather than being appropriate for the needs of all of the species in the community as a whole (Morrison *et al.* 1996).

James (1999) investigated some simple methodologies to assess the effectiveness of prescribed burning practices. While many studies on this research question focus on how long the fuel levels remain reduced, some authors have recommended studies based upon the evaluation of the actual results of burns in terms of management objectives (James 1999). The simple methodology used by James (1999) involved using a pro-forma including information on assessing the fuel levels, cover, and different variables of fire behaviour. Furthermore, fire size was assessed by walking the perimeter of burns and ocular observations were made of the amount of fuel removed. This was designed to enable people with no formal training to perform post-burn assessments. Results indicated that 30% of burns were negative for fuel removal (ie burnt more than anticipated), 40% were sub-optimal, and 30% were rated as effective burns (James 1999). Further, canopy scorch was assessed as not extensive, with only 79% of burns only reaching up to 15% of the canopy. However, the use of visual assessment is very subjective and differs among observers for efficacy of burns, especially with the pro-formas designed for people with no formal training. In addition, there are other compounding factors, which make this type of assessment problematic. It was discussed that the denser the vegetation, the poorer the results (increase in vegetation complexity), and also that different types of woodland gave different results (James 1999). Despite some failings, meaningful trends could be seen in the results, and provided an adequate assessment of the effectiveness of prescribed burns for fuel management purposes (James 1999).

Rose *et al*. (1999) highlighted the notion that spatial variability is required when planning prescribed burning, especially for the conservation of biodiversity. This is very similar to the Bradstock *et al*. (1995) research paper, however, this is depicted with a practical perspective. There is the notion that returning the landscape to its ‘natural’ fire regime (no prescribed burning and no active suppression of wildfires) is quite attractive, but as House (1995) states, probably not achievable in today’s environment. However like much of southeast Queensland, there are few areas in New South Wales, which have the (large) size, remoteness and appropriate *natural* fire barriers needed to support such regimes (Rose *et al*. 1999) where no controls are implemented. However, from my (Cuong Tran) own experience in Tasmania, there is a wide recognition that for some World Heritage Areas, there is a notion of ‘unbounded’ burning, that is, active burning for maintaining a mosaic pattern, where no active fire suppression is enforced (Marsden–Smedley *et al*. 1998). The climatic conditions in Tasmania allow the leaving of large-area burns where strict boundaries are not adhered too, and also allows the fires to create its own amount of variability and patchiness. There is possibility of this type of unbounded burning in Queensland, but this is probably restricted to some remote areas and probably not applicable to southeast Queensland, largely due to the potential threats to life and property.

Rose *et al.* (1999) pointed out the relevant issue of the ecological edge effects on artificial fire-lines and breaks, which experiences a wide range of detrimental effects from prescribed fire management. Their main concern is the use of roads, which are basically artificial edges in the centre of some vegetation community, a spatial difference exists on either side of that road where each block receives a markedly different fire regime. This is better illustrated in Figure 4.

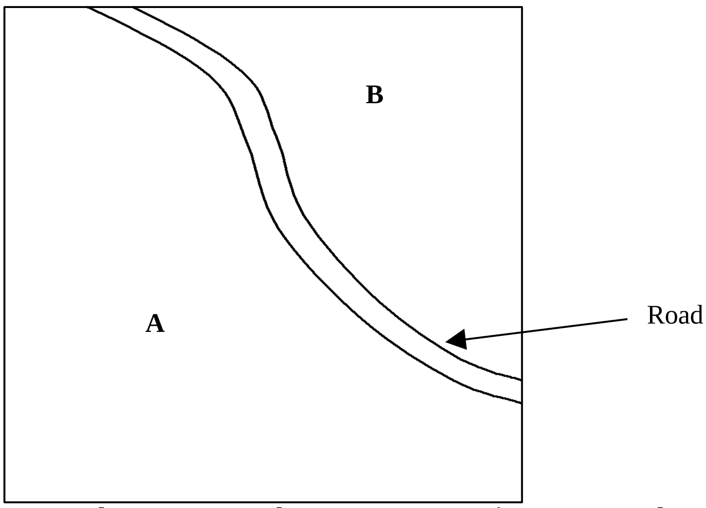

Figure 4: Blocks “A” and “B” are the same vegetation type. The road marks some geographical/topographical boundary. Each block will receive a markedly different fire regime, eg Block A is the control, ie unburnt, and Block B is burnt annually. Rose *et al.* (1999) points out that the edges along the road to both blocks will be very different, and worth further investigation.

According to Rose *et al*. (1999) there are two principles for determining spatial patterns of fire intervals within planning: 1. One plan derived from fuel loads, and applied near settlements, and 2.

The other aimed at conserving biodiversity, generally applied away from community assets. Furthermore, Rose *et al*. (1999) identifies four spatial levels based upon a criteria of fire behaviour potential, fire regime threshold guidelines for broad community types (very broad), location and vulnerability of assets and fire sensitive ecological features, and naturally occurring fire mitigation factors and fire suppression options. Once these criteria were applied, the four spatial levels identified were: 1. Regional; 2. Management Zone; 3. Vegetation Community; and 4. Burn Patch. These four separate levels require further exploration (below). It must be emphasised that Rose *et al*. (1999) take the perspective that life and property protection essentially becomes the primary objective of fire management, and then the ecological values are then examined. While this may clash with the holistic view of the conservation of biodiversity, it is nonetheless a 'real-world' perspective on an extremely complex issue. Furthermore, it would seem that to initiate *any* concept of conserving biodiversity within a region, the protection of any human life and other asset must firstly be assured. Recent conversations with the land managers and planners for the local councils, shires, and government departments certainly seems to abide by this view, of 'life and property first, ecology second'. With this in mind, the following perspective will be worthwhile to consider for southeast Queensland. These strategies are recommended for southeast Queensland:

1. **Strategic Mosaic at Regional Level**: Fire management is best when it is planned across land tenure boundaries, where a location such as conservation reserves (becomes the 'core') should consider the management of nearby lands and within the region. Beyond this core area the region may include: all local communities affiliated with the core; a 'fire catchment' (that would contain the largest sized fire to enter/leave the core); political boundaries; or biophysical regions. This regional perspective allows identification of the relative locations of ecological and community assets, historical fire paths, ignition sources as some examples, to allow appropriate options for planning in the regional context. Further, a national (and international) perspective on the significance of the core area may also be relevant. A regional analysis would then allow prescribed burning to be applied to the most effective location, so the most 'intense fuel reduction' occurs adjacent to community assets. Furthermore, a regional level mosaic becomes apparent in the spatial patterns of current (and proposed) fire regimes. If fire histories and planning shows an area to have a high fire frequency then it may be appropriate for a conservation reserve to provide longer fire intervals (for some large proportion of the vegetation communities). Identifying the existing and proposed mosaic at this regional level clarifies specific objectives of fragmented or remnant qualities of a reserve; which may include the maintenance of habitat corridors, continuity of food supplies and appropriate pattern of post-fire stages.

2. **Strategic Mosaic at Management Zone Level**: More detailed planning usually occurs at the sub-regional level (eg conservation reserve) involving more specific management zones and strategies. The aim of this sub-regional planning, particularly for conservation reserves, should be to maintain fire regimes for all zones within some 'biodiversity threshold guidelines'. This is the first stumbling block for southeast Queensland, as for some areas this is just not known. As readers may note, this contrasts with the simpler approach where 'fuel-related' thresholds are pursued first regardless of the thresholds for biodiversity. For this to be implemented, a range of fire intervals should be applied across the landscape with as Rose *et al*. (1999) explains, each of these intervals a subset of recognised biodiversity threshold guidelines for each vegetation community. As discussed above, the most intense fuel management for property protection occurs adjoining some community asset (eg building). At this sub-regional level, protection actions are focussed adjoining these assets, and management in the wider area complements those actions in an ecologically sustainable manner. Interestingly, it is stated that the differences in burning regimes for conserving biodiversity and property protection (through wildfire mitigation)

is reduced once the focus is on spatial patterns of burning (within those threshold guidelines) rather than on the temporal patterns (fire frequencies) alone. With this in mind, it could be comprehended that an application of this sort would occur like:

- Shorter intervals for zones near vulnerable property or fire sensitive biota;
- Mid-range intervals for zones with 'mid-range fire potential; a predominance of species or habitats requiring this range; and
- Longer intervals for zones located distant from vulnerable property, sites containing fire sensitive biota or predominance of biota needing longer intervals, and on less fire prone sites.

While this may not be the 'natural' mosaic, this spatial mosaic would maximise natural control zones, fuel and moisture gradient and minimise human impact.

3. **Strategic Mosaic at a Vegetation Community Level**: As vegetation communities are recognised to have different flammability potentials and require different regimes to maintain biodiversity, such that a strategic mosaic (based on vegetation level) is both warranted and feasible. Using the biodiversity threshold once more, the plan is to identify the portions of a reserve in each successional stage and the various inter-fire intervals and areas where biodiversity thresholds have been exceeded.
4. **Strategic Mosaic at the 'Burn Patch' Level**: This is aimed directly at minimising the impact of fire management on threatened species. Where specific management objectives are planned for threatened species, a 'burn patch' size (discrete areas within the perimeter) can be provided. This would vary according to the species under threat, but it was stated that burn patch size not exceed four hectares (4 Ha) to minimise the effects of changing fire behaviour associated with changes in wind speed and direction.

(Information adapted from Rose *et al.* 1999)

Combining all four strategic spatial levels, a complex and landscape wide pattern of fire regimes is possible to be maintained. For southeast Queensland, to achieve a similar result for areas of high conservation value in addition to areas of high life/property value would be an ideal conclusion to this study. As previously mentioned, the most significant amounts of published literature on the effects of fire on the landscape have been conducted at Cooloola National Park and the long-term plots managed by Department of Primary Industries. An example of work relating to the effects of prescribed burning on forest ecosystems is discussed below.

Guinto *et al*. (1999) assessed the long–term effects of repeated prescribed burning on diameter growth of trees in mixed species dry and wet sclerophyll forest sites in southeast Queensland. The results show that variable growth response of species to fire, but for most species frequent burning had no deleterious effect on tree growth (Guinto *et al*. 1999). The sites used in the study were the Baupal and Peachester plots mentioned previously. Annual burning (since 1952) at the dry sclerophyll site had no effect on the growth rates of *Eucalyptus drepanophylla* or *E. acmenoides*, with *E. tereticornis* actually responding favourably to annual burning. Other interesting results showed that the smaller *Corymbia variegata* trees appeared to respond positively to annual burning, whereas larger trees responded negatively, with this effect lessening over time. Periodic burning (every 2±3 years since 1973) had no (significant) effect on the growth rates of any of the tree species examined (Guinto *et al*. 1999). At the wet sclerophyll site (biennial burnt since 1972) burning has enhanced the diameter growth of *Lophostemon confertus* but depressed that of *Syncarpia glomulifera*. As Guinto *et al*. (1999) state further, the basal area growth of most eucalypts at this site was unaffected by burning, however, both *S. glomulifera* and *L. confertus* was adversely affected by burning due to tree mortality. According to the results, the authors decided that this was related to the diameter dependency (smaller trees mean less chances of survival) and fire related (more frequent fires reduces survivability). This result and the greater mortality of smaller trees with frequent burning suggest that if these trends continue future stand growth and hence productivity of these species could be jeopardised because of the reduction of the regenerative capacity of the forest (Guinto *et al*. 1999). Unfortunately, this experiment lacks an additional site which examines the effects of some mid-range (eg 7–10 years) fire interval. If this were available for study, then there would certainly be a clearer picture of the effects of repeated fires on one vegetation community within southeast Queensland.

Because of increasing concern over the constancy of intervals between prescribed fires within a vegetation type, various sources of evidence that can be used to determine variation appropriate to the conservation of biodiversity while minimising the chances of economically destructive fires were examined by Gill and McCarthy (1998). As Gill and McCarthy (1998) state, the "primary juvenile periods of plants (especially of '*serotinous* seeders') and non–breeding periods of birds (especially poorly dispersed species) suggest extreme lower limits for fire intervals whereas longevity of plant species which usually only reproduce after fire, set the extreme upper limits … (and that) the modelling of the behaviour of selected plant and animal species may be used to set 'optimal' mean intervals". Furthermore, historical fire–interval data might seem a useful way to determine the variation about the mean fire–interval but "*data are scarce and interpretations are controversial*" (Gill and McCarthy 1998). Lastly, any practical solutions to the question 'what range of fire intervals should be used at any one site?' may be achieved using highly simplified skewed distributions, constructed on the basis of land–management objectives. This statement rings true for this and many other regions of Australia.

Hazard reduction has become the main focus in preventing fires. The use of prescribed burning is actively encouraged, by many authorities and town planners. In the United States, Wright and Bailey (1982) have stated that prescribed burning is the best method of preventing fires. Prescribed burning is now a widely accepted practice, and more emphasis has been placed on such use, after the fires in Yellowstone National Park (Wyoming) in 1991 (Turner *et al.* 1995).

However, this form of fire management has some distinct disadvantages. Periodically burning off 'high-risk' vegetation such as *Eucalyptus* every 5–10 years does not provide enough time for the trees to flower and produce enough seed-bank reserve, so in one fire season with an inappropriate fire frequency the natural inhabitants of the area may be lost. This is especially important in Australia, since trees such as eucalypts require 20–30 years or more before the fire can perpetuate the trees life cycle, by stimulating regeneration or seed release. Eucalypts recover through budding from lignotubers (in young and old trees) or release from seed from capsules after the fire has killed the branch or parent tree (Gill *et al*. 1981). In this case, the tree may need to be quite mature to possess arrested fruits. As Sadler (1993) reports, if a fire goes through a stand before trees start to produce seeds, say in the first twenty years, the species will have no mechanism of regeneration and may be eliminated from the area. Even though the author was referring to the effects of fire on the Victorian Mountain Ash (*Eucalyptus regnans*) which does not occur in Queensland, the main point to understand is that for some areas, there is a need for longer inter-fire intervals. In addition to longer fire intervals, increasing the variability in these fire intervals will be a good first step for managing the landscape for life and property *and* biodiversity.

Catling (1994) reports that use of prescribed burning (usually low intensity fires) can have detrimental effects (long term) on abundance and species diversity of ground-dwelling mammals and birds. This finding contradicts the perceived notion that high intensity bushfires, which cause high mortality to fauna in the short term, that may threaten the long-term survival of fauna (Catling 1994). Moreover, the use of low intensity fires by many forest managers (which are perceived to have low impacts on fauna) can dramatically alter the habitat. Widespread and major changes to the forests, such as a change in fire regime, could threaten the fauna (Catling 1994). Habitats that regenerate after high intensity bushfires can be lost by the frequent use of low intensity prescribed fires, and Catling (1994) poses the question, “what effect does frequent, low intensity fires over the last 60 to 70 years in the 1 million hectares of Australia’s forests that are burnt annually, have?” The reported effects of low intensity fires are:

- reduced shrub cover and total biomass of shrubby and herbaceous species;
- exposure of soil;
- invasion of alien species; and
- stimulation of grass seed production; Catling (1994)

Intense fires (more than 3500 kW/m^2) usually defoliate trees, destroy understorey shrubs and totally remove the forest floor cover. Such fires result in rapid proliferation of shrub and coppice forest, seed-germination, vegetative regeneration and perpetuations of nitrogen-fixing plants such as native legumes (Catling 1994). Catling (1994) also concluded that a drought before a low intensity fire would hasten the decline in understory cover and of native fauna.

Bradstock and Auld (1995) discussed that low intensity, prescribed fires (use for fuel reduction) may be detrimental to the conservation of flora because the heat derived is insufficient to stimulate the germination of buried, dormant seeds. Catling (1994) confirmed this finding and further added that hard-seeded plants belonging to families such as Myrtaceae, Casuarinaceae, Proteaceae, and Leguminosae do not germinate in low intensity fires. This demonstrates that prescribed burning, while reducing the fuel in a habitat, actually can be very detrimental to the survival of habitat, since the change in fire regime can alter the natural vegetation that occupied that habitat. The problem also lies with the fuel reduction purposes of prescribed burning. Low intensity fire are not the problem for land managers, moreover, it is the very notion that fuel reduction is all that can be achieved using these fire intensities. As previously stated, a fire regime incorporates many aspects including fire intensity and managers must realise that the notion of using high intensity fires is as important as varying fire intervals and varying fire seasons.

Therefore, the use of prescribed burning, while widespread, has some major failings, which make it unsuitable for fire management (of course any other method of fuel reduction will have similar effects), especially in Australia. With the specific requirements of various Australian species, frequent low intensity fires may permanently alter the landscape. An alternative fire management practice/plan is required, which includes high intensity fires.

The next section of the report examines the use of fire to manage for different areas of southeast Queensland and critiques the impact on biodiversity and what measures may be implemented to mitigate the loss of species. Furthermore, the report also highlights any gaps in our current knowledge. For some sections of the report, the identified gaps are recommended as future research projects which may answer some of the pressing questions of fire regimes for southeast Queensland.

Review of Fire Management Plans of Southeast Queensland

A number of fire management plans (FMP) have been collected and reviewed (summarised in Table 6, below). The majority of the FMP's were from National Parks (e.g., Scenic Rim National Parks by Novello and Klohs 1998) and Conservation Reserves, though there were samples of FMP's from local shires and council (e.g., Helidon Hills by Gardner 1998). The table below lists the currently reviewed FMP's – with possible indicator species for each area.

Table 6: Analysis and Review of Fire Management Plans for SEQ Region

Area/Region	Source of Reference	Major Vegetation Community	Suggested Fire Regime	Ecologically Sensitive Species (Possible Indicators)
Treetop Sanctuary Property (nr. Gatton)	Cox (1998)	1. Tall Open Eucalypt Woodland 2. Dry Rainforest	1. None stated. Any fires must be small and controlled 2. As above	• *Boronia* sp.
Dwyers Conservation Park	Hughes (1999)	1. Tall Open Forest 2. Tall–Very Tall Open Forest 3. Low Closed Forest (Vine Scrub) 4. Riparian & Cypress Pine	1. 8–20 yrs (varying intensity) 2. 8–15 yrs (varying intensity) 3. Fire exclusion 4. Fire exclusion	• Black Breasted Button Quail (*Turnix melanogaster*)
Helidon Hills (nr. Gatton)	Gardner (1998)	1. Tall Open Eucalypt 2. Open Eucalypt 3. Open Eucalypt Woodland with mixed understorey	1. >10 yrs 2. 4–5, >10 yrs 3. 4–5, >10 yrs, up to 25 yrs for shrubby understorey	• Glossy Black Cockatoo (*Calyptorhynchus lathami lathami*)
Burrum Coast NP – Kinkuna	DoE, Central Coast (1997)	1. Mixed Open Shrubland 2. Low Open Woodland 3. *Melaleuca* Low Woodland 4. Open Woodland	1. 6+ yrs. Vine–no fire 2. 6–8 yrs 3. 8+ yrs. 4. 4–6+ yrs	Many possible fauna species. Good indicators incl. • Wallum Froglet (*Crinia tinnula*) • Freycinet's Frog (*Litoria freycineti*) • Wonga Pigeon (*Leucosarcia melanoleuca*) • Feathertail Glider (*Acrobates pygmaeus*)
Mount Coot-tha	USQ (1995)	1. Open Eucalypt Woodland	1. 3–8 yrs	
Logan City Bushfire Hazard and Risk Assessment	Kingston *et al.* (1996)	1. Dry Sclerophyll Open Forest/Woodland 2. Wet Sclerophyll Open Forest (with Rf Elements) 3. *Melaleuca* Forest	1. 5–12 yrs, also 5–20 yrs 2. Depends upon Wet Sclerophyll or Rf structure 3. No fire	• *Melaleuca nodosa* • *M. tamarascina* subsp. *irbyana* • Koala (*Phascolarctos cinereus*)
Border Ranges NP, Limpinwood NR	NSWNPWS (*c*. 1996)	1. Subtropical Rainforest 2. Dry Sclerophyll Forest 3. Wet Sclerophyll Forest	1. Fire exclusion 2. <30 yrs, more than 5 yrs 3. 50 yr interval, <200 yrs	• Eastern Bristlebird (*Dasyornis brachypterus*)
Moogerah Peaks NP	DEH (1995)	1. Tall Woodland with Heath understorey	None stated. Mosaic burning recommended	• Brush-Tailed Rock Wallaby (*Petrogale pencillata*)

				• Black Breasted Button Quail (*Turnix melanogaster*)
Magnetic Island NP	Thomas (1997)	1. Coastal Woodland 2. Araucaria & Vine Forest 3. Mixed Eucalypt Woodland	1. Coastal She-Oak– No Fire. Other–1–3, 3–5 and 7 yrs alternating 2. No Fire 3. 3–5, 5–10 yrs	None stated
Springbrook NP	Hall (1997)	1. Rainforest (incl. Vine, Warm Temperate Rf) 2. Open Eucalypt Forest 3. Eucalypt Forest with Mixed Species 4. Mid–Tall Dense Forest (*Eucalyptus campanulata*) 5. *E. oreades* Forest 6. Heath on Rocky Areas 7. Lantana with Scattered Trees	1. No Fire 2. +10 yrs 3. None stated, but fuel layer ensure high frequency 4. +50 yrs 5. +40 yrs 6. 15+ yrs 7. Frequent for weed management	*Numerous significant species*. Many possible indicator species–further research needed.
Gold Coast Bushfire Management Strategy	Gold Coast City Council (1998a)	1. Vine Forest including Riparian Forest 2. Wet Sclerophyll Forest 3. Dry Open Eucalypt Woodlands 4. Paperbark Open Forest 5. Estuarine Complexes 6. Coastal Complexes	1. No fire 2. 15–20 years 3. 7-12 years 4. +15 years 5. No fire 6. 8–10 years	*Numerous significant species*. Many possible indicator species–further research needed.
Scenic Rim National Parks	Novello and Klohs (1998)	1. Cool Temperate Rainforest 2. Warm Temperate Rainforest 3. Subtropical Rainforest 4. Dry Hoop-Pine Rainforest 5. Wet Sclerophyll Open Forest 6. Dry Sclerophyll Open Forest/Woodland *7. E. oreades* 8. Shrubland 9. Open Grassland 10. Rocky Pavements	1. No fire, maintain low adjacent fuel loads 2. No fire, maintain low adjacent fuel loads 3. No fire, maintain low adjacent fuel loads 4. No fire, maintain low adjacent fuel loads 5. 50–200+ yrs 6. Shrubby understorey: 8–10 yrs; Woodland: 5+ yrs 7. 50–100 yrs, minimum 25 yrs 8. 8–10 yrs 9. 3–7 yrs 10. No fire	*Numerous significant species*. Many possible indicator species–further research needed. Recommend–Rapid Biodiversity Assessment for this region. Best candidate: Invertebrates

NB: Some FMPs are in **Draft** form. Furthermore the recommended fire regimes listed were predominantly for ecological benefit and not human/property safety.

As the table shows, there are some distinct differences in recommended fire regimes for vegetation communities, which contain the same types of canopy and understorey vegetation. This is partly due to the different strategies employed in recommending a fire regime; whether it is more strongly influenced by the prime objectives of either:

(i) ensuring maximum safety for local residents, or

(ii) maximising ecological benefits of prescribed fires

Importantly, Table 6 shows that there are some very site specific needs that are incorporated into these fire management plans which may explain the close intervals in some of the fire regimes. As previously stated, when considering the entire southeast Queensland region, maximising the variability of fire regimes is very important, and as these cases have shown, more stringent regimes can be recommended when more information is known about the ecological processes in each area. The table contains a wealth of information and may be applicable to shires/councils close to the area originally under examination in the management plans. It must be noted that the primary objective of most of these management plans is the protection of life and property assets, and not conserving ecological diversity. With this in mind, it is recommended that before accepting the suggested fire regimes in their entirety, there should be some consideration of increasing the minimum fire intervals and the variability of the fire regimes. The use of ecologically sensitive species in each region is also worth pursuing.

The following sections of the report shows a more comprehensive examination of fire within each specific vegetation type, which was prioritised, as ‘High’ on research agendas. The first vegetation community that is examined is Wet-Sclerophyll Forests, followed by Eucalyptus Open Forest, Melaleuca and Coastal Woodland, and finally Heathlands.

Fire in the Wet-Sclerophyll–Rainforest Boundary

Melick (1990) examined the ecology of the boundaries between the rainforest and wet sclerophyllous boundaries within a National Park in Victoria. It is recognised, none more so than by the authors, that there are some wide differences between the stated vegetation types in Victoria and Queensland, but Melick (1990) provides a very good discussion explaining the variations in the two community types. Furthermore, with the lack of rigorous scientific research in these areas in Queensland, particularly, southeast Queensland, this is a firm starting point for further research.

There are some striking similarities though. Rainforests in Gippsland are recognised as being "fire-proof" and only through major disturbance, ie a *major* fire, will there be any penetration of sclerophyll species, sometimes as emergent species (Melick 1990). This has also been recorded in many regions of southeast Queensland (Hall *pers. comm*.). Rainforest communities were generally restricted to gullies and damper regions, near creeks and permanent water bodies, whereas sclerophyll vegetation dominated the dry plateaux and ridge tops (Melick 1990). This geographic distribution of vegetation community type, to some extent also determines the likely paths of any fires in the region, as the fire history within these areas show that the rainforest remains relatively untouched (with a few highly-damaging exceptions). As Melick (1990) states, the relationship between the plant communities and topography appears to be related to the susceptibility to fire at these sites…with the hotter drier ridges and plateaus generally more fire prone than the gullies and gorges. It is also apparent, that in between the dry (sclerophyll) and wet (rainforest) communities exists an unstable sclerophyll-rainforest mix, that is known as wet-sclerophyll vegetation. In the Mitchell River, where this study was conducted, the fire regime permitted the re-establishment of *Pomaderris aspera*, allowing the invasion *of Kunzea ericoides*, and enabling thick stands of *Acacia mearnsii* to colonise as well (Melick 1990). It is apparent that devoid of fire, *P. aspera* would seem likely to disappear from this area. It would seem that in Queensland, changes in edaphic (soil) conditions have been attributed to the decline of rainforest and sclerophyll communities. However, it is also important to recognise that the effects of climate, topography, and disturbance (history) can be far more important in determining the composition of vegetation in these ecologically sensitive areas (Melick 1990).

Turton and Duff (1992) explained that in the absence of fire, rainforest (in many sites) will advance into areas previously dominated by fire tolerant (pyrophytic), open-forest or grassland vegetation, namely *Eucalyptus* species. Conversely, following fire, pyrophytic vegetation may replace rainforest, and through time, with frequent occurrences of disturbance, will permit further penetration into previously exclusive rainforest territory. Ash (1988) also studied the boundaries of northeastern Queensland similar to Turton and Duff (1992) and in addition to the contraction and expansion of the boundaries between the two forest types in response to disturbance, the extent of this disturbance needs to be considered. Mild fires occur usually every few years to decades, but highly intense fires which will kill the tall *Eucalyptus* trees but it also facilitates (ie triggers) their regeneration from seed (Ash 1988). This cycle may occur at intervals of many decades or even *several* (authors' emphasis) centuries (Ash 1988). This long fire-free period will allow rainforest species, some species of which are typically slow growing, to regenerate, and to dominate the canopy as it did before fire. As Ash (1988) explains, the advancement of the rainforest when devoid of fire may be apparent, but over a long (centuries) period of time, these boundaries repeatedly advance and contract such that they end up in the same position. For this vegetation type, at least, an ecologically sustainable fire regime may not be achievable.

Floyd (1976) examined the regeneration of seeds of wet-sclerophyll species from burning. From the results, it is apparent that in addition to the requirement of fire as a disturbance mechanism, the frequency and intensity of the fire is vitally important in determining the species composition that results from regeneration. Within the forests of Coffs Harbour, northern New South Wales, Floyd (1976) found that the wet-sclerophyll species remain dormant and buried in the soil until heated, and that each species required specific heating requirements for regeneration. With such particular requirements for germination, a fire's duration and intensity will affect what species will revegetate in the area (Floyd 1976). The results indicated, that hot fires favoured species such as: *Dodonaea*, *Kennedia*, *Commersonia*, and many *Acacia* species, whereas light fires (fuel reduction burning) stimulated rainforest seral species such as *Callicoma*, *Piptocalyx*, *Helichrysum*, *Zieria* and *Halogaris* (Floyd 1976). Interestingly, Floyd (1976) stated that frequent fuel reduction burning not exceeding 14 years, could result in species of *Callicoma*, *Piptocalyx*, and *Halogaris* replaced with more aggressive pioneers, such as *Phytolacca* sp., and *Acacia bivernata*, reiterating that fire intensity and frequency can cause major changes in composition of understorey vegetation.

Unwin *et al.* (1990) examined the effects of different fire intensities on a rainforest-eucalypt boundary in north Queensland. Even though much of the work revolved around the fire behaviour on the boundary, there were some discussions about fire regime, in which the authors mention the following influences:

- topography,
- soils, especially topographic variations (which affect moisture and fuel distribution),
- climatic patterns and seasonal growing conditions,
- structure and the successional patterns of the forest type,
- type of land use (grazing etc.), and
- recent fire history, intensity, pattern, and frequency.

This presents a few ideas for assessing the effectiveness of fire regimes for many areas. Despite the fact these are quite broad and generalised characteristics that Unwin *et al.* (1990) has outlines, the principles of these influences on fire regimes should not be understated. Burrows *et al*. (1999) used similar categories to define some variables to use to measure fire regimes in Western Australia. The most interesting results of the post-fire recovery in Unwin *et al.* (1990) report showed that:

- intensity was a strong influencing factor on the survival of plants, especially for plants within the rainforest,
- good indication that many species within the rainforest margin (though quite 'sensitive' species) are also quite fire-resilient (results showed over two-thirds of plants killed directly by the fire, recovered within the following season, mostly by basal coppicing), and
- topography has a large role in the shaping the fire pattern, which in turn, determines the nature/extent of the fire along the margin.

This suggests (as others have claimed through anecdotal evidence) that fires must be of high intensity to have any *lasting* impact on rainforest margins. Further, this may include 'follow-up' burning in the following seasons (2–3 years) after the initial 'hot' fire as indicated by some managers in southeast Queensland. Without any scientific study to determine if this 'follow-up' burning is necessary, then this should be treated with some caution. It may seem apparent that this second, lower intensity, fire would kill those plants that were not killed in the first fire. However, more work is definitely needed before this can be recommended for wet sclerophyll forests of southeast Queensland.

Harrington and Sanderson (1994) examined the contraction of wet-sclerophyll forest from invasion by adjacent rainforest in northern Queensland. This contraction was due to the increased length of a fire-free period, which allowed rainforest seedlings to establish and suppress the grass-layer (Harrington and Sanderson 1994). It was discussed that the wet-sclerophyll vegetation type is an endemic and isolated region, which supports poorly known fauna groups such as the Yellow–Bellied Glider (*Petaurus australis reginae*) which is totally dependent upon wet-sclerophyll forest (Harrington and Sanderson 1994). The authors point out that it is highly debatable whether to keep wet-sclerophyll forest and prescribe fires to halt the spread of rainforest into these areas or to permit the encroachment on wet-sclerophyll forest and possibly endanger the existence of dependent species, such as the Yellow–Bellied Glider. This only highlights the need for further research into this community type.

Kellman (1986) investigated the fire sensitivity of *Casuarina torulosa* in wet sclerophyll forest in northern Queensland. Too frequent fires lead to the decline in *C. torulosa* stands, however, it was also shown that infrequent fire also decreased the competitive abilities of the plant which would lead to its decline as well. This was discussed as being a typical characteristic of all wet sclerophyll woodland (Kellman 1986). It was suggested (though no actual fire regime is stated) that fire periods between these two extremes would lead to a population of *C. torulosa* forest (and presumably all wet sclerophyll vegetation) that would be capable of indefinite persistence (Kellman 1986).

Guinto *et al.* (1999) showed that frequent burning in wet-sclerophyll forest greatly reduced the numbers of *Syncarpia glomulifera*, and overall frequent burning (biennial) showed that smaller trees have lowered chances of survival than larger trees and further burning reduces this probability. More significantly, recruitment was adversely affected by burning, and coupled with higher mortality of smaller trees, too frequent fires would jeopardise productivity due to the reduction in the regenerative capacity of the forest. Hannah (1998) compared the population of herpetofauna in two sites – one burnt regularly (2–5 years) and the other unburnt for 50 years. Species richness and abundance (ie biodiversity) was highest in the unburnt area and showed that the distribution of *Lampropholis* sp. and *Carlia* sp. were significantly affected by changes caused by fire. Even though maintaining an area (which is also highly flammable) unburnt for a long period of time is impractical, it was suggested that mosaic burning at irregular intervals would assist in maintaining these populations of reptiles and amphibians (Hannah 1998).

Despite a considerable amount of work in this forest type, there is much speculation on the most appropriate fire regimes, especially for the unique wet-sclerophyll forests in southeast Queensland. As these areas contain some very distinctive and limited (in distribution) fauna and floral species, this area represents a significant gap in our knowledge. However, from the review and discussions with other researchers, a regime between 20–50 years is needed to maintain this vegetation type. Variability and mosaic burn patterns is very crucial for the survival and conservation of species dependent upon this community type. The development of even-aged stands of wet-sclerophyll forest will eventually lead to the decline of this forest type. Wet-sclerophyll forest is an inherently disturbed (or constant state of flux) environment – so to maintain a stand of forest that does not represent this disturbance is detrimental to the species dependent upon it.

Dry Sclerophyll Forest with Grassy Understorey and Dry Eucalypt Open Forest with Shrubby Understorey

Yates and Hobbs (1997) stated that only 3% of the original temperate eucalypt woodland remains following European settlement. Altered fire regimes were listed among many causes for this dramatic decline. The authors highlight the fact that degradation and loss of biodiversity will continue unless management of the existing areas are altered, and the review shows that there are few data (including fire regimes) currently available which describes techniques to reverse the degradation (Yates and Hobbs 1997).

In southeast Queensland, a small (but significant) amount of work has been conducted on some specific vegetation types and organisms. White (1990) studied the effects of experimentally prescribed burns on the small mammal populations within Brisbane Forest Park. Different time-since-fire intervals allowed the researchers, to investigate the post-fire recovery of mammals with particular emphasis on rodent species. Even though total species numbers recorded were quite low (ranging from few individuals to tens), the results indicated that the numbers of all species trapped (eg, *Pseudomys gracilicaudatus*, *P. fuscipes. and Antechinus flavipes*) recovered (and exceeded) pre-burn population numbers. This increase in breeding numbers also correlated with favourable climatic conditions after the last fire. Furthermore, the author states that to maintain the population of these mammals, then burning must be conducted in June/early August, before the breeding season (White 1990). Conducting spring burns would, according to the researchers, have more impact, as this is the breeding season and when mates establish territories. However in recent discussions with Neil Gourley (Dept. Primary Industries–Forestry, *pers. comm*.), he stated the preference for switching to summer burning for ecological purposes. The impact of this on the populations of small mammals is probably unknown and requires work to clarify any potential impact.

Tasker *et al*. (1999) investigated the effects of grazing pressures and fire on small mammals in an eucalypt forest in northern New South Wales. Grazing effects on mammals is another poorly researched topic, especially in relation to fire. This project is a small component of a larger study examining the complete effects of grazing on flora and fauna, with extensive studies on invertebrates and flora still to be completed. No differences in species richness or diversity of small mammals were noted in the grazed and ungrazed sites, even though the component of each species was different (Tasker *et al*. 1999). In the grazed areas, only three species were found: New Holland Mouse, *Pseudomys novaehollandiae*, the Hastings River Mouse, *P. oralis*, and the Common Dunnart, *Sminthopsis murina*, an indication of the early successional stages following a fire, and indicative of the fire regime at the sites studied (Tasker *et al.* 1999). The authors concluded, that preliminary analysis of the vegetation composition of the sites showed more diversity at the grazed sites (consistent with the Intermediate Disturbance Hypothesis), though species composition was quite different between the grazed/ungrazed sites (Tasker *et al*. 1999). No doubt, as more work is completed, a clearer indication of the responses to the combined fire use and grazing pressure will be elucidated. Work of this nature is similarly required for southeast Queensland, particularly the non-coastal areas/shires. Even though the study showed some interesting results where there were some more abundant species in the ungrazed areas, it probably highlights the notion that

Watson (1998) provides a comprehensive review of the adaptations and responses of plants to different fire regimes, in accordance to fire intensity, season and frequency. Broad guidelines were recommended for planners for fire management in sclerophyllous communities, which are particularly to this project. These include:

1. *Management of fire is essential in all Australian sclerophyllous communities* – Fire is a natural and necessary event in the regeneration of many endemic species.

2. *Planning for the Management of Fire should be based on explicit, and ecologically sound objectives* – Community-level consideration is required rather than particular individual populations/species.

3. *Planning for the Management of Fire should be based on a sound knowledge and understanding of the biological resources of the area* – Apart from knowledge of the distribution of the community and its species, regeneration mechanisms used by these species is also required, such as seed bank, longevity, juvenile period.

4. *Fire managers should recognise that not all species present in a community may be visible above ground* – Recognition of dormant (or 'buried') species.

5. *Planners need to recognise that in some areas, human habitation and other land uses which demand fire exclusion are not compatible with conservation of biodiversity* – There is a need to recognise that some areas must be designated solely for hazard reduction and other areas solely for conservation and biodiversity.

6. *In areas managed for conservation, fire regimes should involve a diversity of inter-fire intervals and intensities. This diversity should lie within parameters developed from knowledge of vulnerabilities of different functional species types* – Fire frequencies and intensities require variability to achieve the mosaic patterns which are believed to support the highest biodiversity within an area.

7. *Decisions as to whether to suppress wildfire, and to initiate burns should aim to maximise variability.*

8. *Fire regimes should ideally involve some form of patch burning, so that a range of successional stages is maintained.*

9. *All fire management plans should incorporate a comprehensive monitoring program.*

(Adapted from Watson 1998)

Of course, these recommendations apply equally well to other community types. It must be recognised and stated that any planned fire regime will not be optimal for *all* species and decisions have to be made about which one will be the objective of the fire management strategy. The recommendations by Watson (1998) may be correlated and used in conjunction with those of Rose *et al*. (1999) on page 30–31 of this report. The recommendations by Watson (1998) are only valid after some preliminary work is completed to gather information about the area of interest.

For a fire regime to possibly encompass the needs of the most organisms, the inclusion of variability within the fire regimes is extremely important. Setting less stringent limits to the fire regimes (for example, many have argued that in Eucalyptus community types, 7–25 years) would allow for more variability, larger mosaics and consequently, higher diversity. The dilemma is determining the lower limit to fire regimes, which is probably considered the most crucial, when controllable fire risks (especially in highly flammable Eucalypt forest) are close to the threshold limits. This is discussed in the below section.

Morrison *et al*. (1996) concluded that the minimum interval between fire at which little change in the composition of vegetation was noticeable in and around the Sydney region was 7 years. This contrasts with the three–five year fire interval suggested by Simmons and Adams (1986) who strongly advocated that any greater fire frequency would endanger lives and cause extensive property damage. Even Simmons and Adams (1986) state that this fire frequency "may be undesirable for ecological reasons in many areas". This clearly highlights point-5 of the recommended requirements for fire regime planning above.

Cary and Morrison (1995) investigated the effects of different fire-free intervals in dry sclerophyll forest surrounding the Sydney region, and in general, three effects were identified: (i) shorter fire-intervals (1–3, and 4–6 years) were associated with a reduction in species number; unequal abundance in the community (especially dominant Proteaceae shrubs), (ii) fire interval of 1–6 years were associated with an additional *reversible* reduction in the number of fire sensitive species, and (iii) repetition of 1–5 year fire intervals were associated with an increase in abundance of herbaceous fire-tolerant species. The authors concluded by stating that “any variation in these characteristics is thus associated with changes in abundance of different plant species … [where] the floristic composition can be influenced in three different ways by these variables”. And finally, “effective management of fire for plant species conservation must therefore be based on both length of time-since-fire and length of inter-fire intervals” (Cary and Morrison 1995).

Russell and Roberts (1996) examined the effects of four low-intensity burns in a southern Queensland *Eucalyptus pilularis* forest and found that the burning produced less change in the 14-year period than no burning, though only one transect was researched in the study site. Encroachment of *Lantana camara* was also associated with no burning (Russell and Roberts 1996). Nieuwenhuis (1987) studied the effects of different fire frequencies on sclerophyll vegetation at West Head, New South Wales, and the results showed that there was little difference in abundance and composition of vegetation with fire intervals of between 12 and 20 years. Frequently burnt sites showed dramatically lowered live-projection cover of vegetation, particularly from obligate seeders (Nieuwenhuis 1987). Fox *et al*. (1979) showed that 10 years is required for accumulation of litter in *Eucalyptus regnans* forests in Victoria after a fire that will support small mammal species, especially insectivorous mice such as *Antechinus stuartii* and *Sminthopsis murina*, and the bush rat (*Rattus fuscipes*). In all species litter arthropods are a major component of their diets.

The majority of all work on fire effects on the Australian landscape has concentrated in this forest type. The response and regeneration patterns for both flora and fauna are well documented and established. However, the long-term effects of repeated fires is only starting to be established and the response patterns of plants to imposed regimes are only becoming apparent. Eucalypt forest types represent that greatest danger, to both life and property and also loss of biodiversity from incorrectly used fire regimes. Compounding this dilemma further, it is also simply the largest tracts of forests which occur next to urban areas. It would seem appropriate that in these areas, biodiversity values must be considered second to the protection of life (and property). Similar to the zoning proposal by Rose *et al*. (1999) in New South Wales, this is the most likely (and workable) scenario for southeast Queensland. For areas outside of immediate urban (or rural/urban areas) biodiversity values take priority and a suggested fire regime for forest types with shrubby understoreys would be 7–12 (possibly 7–20 years) and in grassy understoreys, a range of 4–8 years is probably suitable. Once again, maximising the variability in this forest type is quite important. Maintaining the largest mosaic within the forest seems to be most crucial for survival of most species. There a few gaps in the knowledge for Eucalypt forests which require further work.

Melaleuca Forest

Very little is known of the effects of fire on *Melaleuca* forest. As shown in Table 4, the Burrum Coast National Park FMP states that this forest type should be burnt on a 8–year minimum rotation, where the Logan City Bushfire Risk Hazard Plan, suggests no fire for this vegetation type. Bartreau and Skull (1994) studied the effects of previous fire regimes in coastal plain *M. viridiflora* woodlands in North Queensland and showed that fire frequencies between 10 and 20 years resulted in a greater proportion of larger trees (>1.5 m) which would, hypothetically, maintain the diversity and regenerative capacity of this community type. However, this research was stated as being the first to study such responses, and to date, no other researched literature has been located. The lack of relevant research in this floral community is critical, given the complexity and uncertainty of long-term altered fire regimes on a fire-adapted environment. To further compound these wide fire frequencies, Twyford (1995) stated that for the Melaleuca forests on Fraser Island, the possible fire frequencies (pre-European) varied between 15–25 years. However the reference to which the fire frequency is stated to originate from is not included in the reference list. Some fieldwork assessments of frequent burning of Melaleuca forests on Russell Island showed that the 1–3 year inter-fire intervals are deteriorating the community with immature even-aged stands of trees are now present in the forest. It would seem that in the lack of any empirical data on fire in this vegetation, a suggestion of long frequencies (15+ possibly up to 30 years) seems suitable.

The lack of available information for fire in Melaleuca forests represents one of the larger gaps in our current knowledge, not only in southeast Queensland but Australia-wide. Urgent work is required in this forest type. This area should be given top priority for research. One of the first research studies in this area should be to identify the life history characteristics of species such as *Melaleuca irbyana*.

Coastal Woodland–Open Forest

This classification is difficult in that it is a very open vegetation community description which could cover many very different forest types. For the purposes of this study, any coastal woodland is one that contains a dominant *Casuarina* canopy with an open understorey mix of plants. While there are many other vegetation types which are located near the coastline, these are covered in the above sections. *Casuarina* (includes *Allocasuarina* species) forests similar to other vegetation types are very susceptible to frequent and intense fires, occur near many urbanised areas but also contains a wealth of diverse plants and animals. Some regionally significant species, such as the Glossy Black Cockatoo (*Calyptorhynchus lathami halmaturinus*), which require *Allocasuarina littoralis* as a food source, is an example of the rich diversity that may be found in these areas. It would seem that fire is needed to halt the establishment of mono-specific stands of Casuarina forests which limits the overall biodiversity (Melville 1995). The question remains over the frequency of such fires and what intensity to allow the burns, as *Casuarina* forests may have tendencies to burn at quite high intensities, which may not suit the urbanised areas which contain this vegetation type. Once again, there should be a trade-off where life and property issues must take priority over biodiversity in these urbanised areas. Species of *Casuarina* are rigorous seeders (Melville 1995) following fire, so it would seem that the minimum period of a fire, should be the time taken for an active seed-bed to establish. For many cases, a minimum period of 6–8 years is probably suitable, but requires validation for each species. An important aspect of all the fire regimes that are suitable for each vegetation type discussed in this report reiterates the need for some variability for the fire frequencies. By maximising the variability, this will allow for a mixed-aged forest, increase the mosaic patterns, and creates different microclimates within the one forest type. This will only serve to increase the species diversity.

Interestingly, Lunt (1998) showed that in an area of coastal woodland in Victoria, which was unburnt for long periods (> 115 years) eucalypt-dominated woodland were suppressed by *Allocasuarina littoralis* and *Acacia pycnantha* coastal woodland. Moreover, it was discussed that in areas where fires and other disturbances such as grazing were absent for some 20–30 years, many fire-sensitive shrubs (such as *Acacia sophorae*, *Kunzea ambigua*, *K. ericoides*, *Leptospermum continentale*, *L. laevigatum* and *Pittosporum undulatum*) invaded coastal and sub-coastal areas to the detriment of rare plants, ecosystem diversity and small mammal habitat (Lunt 1998). Again, Lunt (1998) states that ‘considerable’ work is required to develop ecologically sensitive fire regimes for these environments. The sentiments are also endorsed by the authors of this study.

Fire in Coastal Heathland

Similar to most Australian vegetation, heathlands are highly dependent upon fire to initiate and control regeneration, and in fact fire presents one of the major opportunities, as these events are one of the few times gaps (open areas) appear in these communities (Cheal 1994, Gill and Groves 1981). Previous research shows that fire frequency and season have a strong influence on the composition of heathlands. Cheal (1994) declares that the use of cool season (prescribed) fires does not mimic hot season (wildfires) and any management plans which pretend otherwise will produce long-lasting (even irreversible) changes in the vegetation. Further, it has been shown that heathlands do not reach an ecological climax (unchanging) state – they continue to change as they age. Gill and Groves (1981 in Specht) state that the maximum fire frequency in heathlands is probably once every two years. This frequency was mainly a response to grasses replacing the shrubby understorey. But as Cheal (1994) aptly emphasises, the current ecological knowledge is inadequate for comprehensive management of heathlands long-term; as little is known of the effects of season of burn, regeneration and absence of fire, soil processes, effects of major disturbances in the ecosystem (e.g. drought), and the regeneration of threatened species of fauna and flora. Bradstock *et al*. (1997) examined the effects of high frequency of fire in heathlands surrounding Sydney. The results showed that high fire frequencies reduced the frequency (density) of many plant species; signifying that the structure and composition of this community will be simplified by high frequency fire, resulting in the conversion of heathland to open herb-sedgeland with lowered floristic diversity (Bradstock *et al*. 1997).

Clarke *et al*. (1996) investigated the post-fire environments of wet and dry coastal heaths. The results of the study showed that wet-heaths, in particular, are restricted in terms of re-establishment after disturbance, and that dry-heath species can establish in previously wet-heath areas. Wet-heath species were not restricted from entering the drier habitats, but it was shown that seedling viability of these species were dramatically reduced, particularly as a result of the drier conditions (Clarke *et al*. 1996). Furthermore, the amount of foraging and seedling predation by vertebrates were unchanged between the two areas, though it was theorised that the viability of seed-banks (temporal storage) is far more important than the patch or spatial storage as mechanisms for maintaining diversity and coexistence (Clarke *et al*. 1996).

Brown and Podger (1982) examined an area of sedge-heathland bordering rainforest in southwestern Tasmania for post-fire establishment and recruitment. Admittedly, the peat soil-type in which these vegetation types were found in Tasmania is very limited in its distribution within southeast Queensland, the discussions about recurrent fire regimes and the effects on community structure and composition is quite relevant. Examination of the sites revealed that in the sedge-heathland, unless a fire occurs soon after (a cool, mild fire) heaths will recover from seeding from adjacent areas and it will recover to its initial composition (Brown and Podger 1982).

Successional systems occurred within the sedge-heathlands in accordance to previous studies, whereby sedgeland-heath communities become scrub and then climax as forest communities. As Brown and Podger (1982) correctly state, these zones (community types) are always in a state of flux and become very dependent upon the effects of successive fires, with the floristic association of the remaining broad core areas also in some sort of dynamic equilibrium. This statement may be applicable to any vegetated area throughout Australia. It was discussed that for this particular type of vegetation structure, a fire-free period of 25–30 years will stabilise the scrub communities, due to the low fuel production rate and from the establishment of moister microclimates between the scrub and sedge-heathland. Longer fire free intervals were discussed to promote the establishment of sclerophyll communities (Brown and Podger 1982).

Many studies that have been conducted show the succession of plants/animals/invertebrates from one single fire. While the comment from Brown and Podger (1982) shows that longer term examination of previously burnt sites is needed to attain a quantitative assessment of the true effects of fire on the landscape, this requires an exhaustive amount of resources and some permanent sites to monitor. This is, of course, not always achievable. This is explored further in the section below, where long-term experiments within southeast Queensland and throughout Australia will be discussed. Posamentier *et al*. (1981) examined the processes of succession in a coastal heathland in Nadgee, New South Wales. The sites were examined every two years since the wildfire, and the study finished at the sixth year following the fire. Much like the Brown and Podger (1982) study, species regenerated rapidly following the fire, with many prefire species regenerating quite vigorously (after three years, species richness doubled), with the change in pattern from that of a temporal to a predominantly spatial one of the largest differences (Posamentier *et al*. 1981). It is quite clear, that in the first year following fire with the open ground and rich amount of nutrients available, effectively allows rapid plant regeneration. Posamentier *et al.* (1982) also stated that within 12 months of the fire, 60% of species recorded pre-fire had returned. The authors also hypothesised that the reasons behind the high extent of regeneration may also have be related to the high intensity fire that the area experienced. This is becoming an increasingly important ideal with many researchers believing that to achieve the natural (whether this is pre-European or pre-human settlement is debatable) fire regimes, areas must be allowed to experience high intensity fires (such as Nadgee). As the study has shown, this has permitted the degree and type of regeneration of species to occur, which maintains and strengthens the species richness within that area.

In southeast Queensland, the most studied area containing any significant amount of heathland is Cooloola National Park, and has been predominantly conducted by McFarland (1990, 1988a, 1988b) and Sandercoe (1989). More specifically, these researchers have extensively studied the effects of fire on the flora and avifauna within Cooloola NP. Sandercoe (1989) also provided an extensive review of the level and amount of research throughout heathlands of Queensland. Sadly, it seems since the end of their respective experiments, little work has since been completed in this crucial (for biodiversity) community type (McFarland *pers. comm*.). This area is recognised as being the regional centre for endemic species and contains the highest biodiversity within the southeast Queensland region (SEQ CRA/RFA 1998). Similar to Brown and Podger (1982), Sandercoe (1989) reports that within both the wet- and dry- heaths of Cooloola, rapid flowering and seedling intensity occurred within two years of the last fire, with vegetation cover reaching a plateau after 5.5–6.5 years after which there were no noticeable changes. The work competed at Cooloola was conducted in conjunction with the establishment of other heathland sites in Beerwah (Scientific Area 1), which has been under the authority of the Department of Primary Industries (Qld Forestry Research Institute, QFRI). For the other designated Scientific Areas there are experimentally induced fire regimes of three and five years (with an area left unburnt as the control) which remain (along with Bauple and Peachester, all under the auspices of the QFRI) the only long-term experimental plots in Queensland (Sandercoe 1989, House *pers. comm*.). Interestingly, Scientific Area #24 has not been burnt since the 1920’s and remains a unique area, with excellent potential for future research work (House *pers. comm*.). For most of the research conducted in Queensland, the interaction between fire frequency and vegetation has been predominantly the main focus of the research, whereas other important information such as fire behaviour, and fire season were overlooked due to logistics and the nature of the habitats in which the experiments were conducted (Sanderoce 1989).

McFarland (1990 and 1988b) investigated the composition of vegetation in six heathland sites in Cooloola, varying in time since fire from 0.5 to 10.5 years. Similar to Brown and Podger (1982), there were two distinct microhabitats of wet and dry heath within the study region, and the results showed similar attributes as well. Within one year of the fire, the composition of species recorded were unchanged between each of the six areas, with other indicators of plant growth and composition showing that approximately 6.5 years after the fire, species reached maximum plant growth, plant density, plant height (McFarland 1988b). With these results, it was recommended that for heathlands in Cooloola NP, a fire regime of 8–10 years would provide adequate time for the plants to generate enough seed, and allow the area to burn with sufficient intensity (though this has not been studied)

(McFarland 1988b). Benwell (1998) investigated the post-fire regeneration in coastal heath in northern New South Wales, and found that obligate seeders and facultative resprouters had overlapping but distinctive distributions especially with respect to seedling recruitment. However, obligate seeders were shown to do better than facultative resprouters in seedling density and survival (Benwell 1998).

The following comments from Sandercoe (1989) are very applicable to the present status of fire research in Queensland: standardisation in the methodology is needed, both within Queensland and Australia-wide (this will allow comparisons, both ecologically and statistically). Furthermore, there is an urgent need for more long-term studies, as this is directly applicable to the management of conservation areas. Additionally, detailed fire histories are required as this will help in the better determination of sites suitable for floral and faunal studies. More specifically, *in lieu* of such studies, mosaics (or maximisation of variability) is needed, with a preference for older aged stands of heathland in Queensland (Sandercoe 1989). It is certainly clear that leaving areas unburnt for an indefinitely long period will lead to the loss of some species, but this is obviously dependent on the viability of the seed-banks of those species. As with many other vegetation communities, fire is required in heathlands at some stage to maintain its composition.

Clearly, for heathlands in southeast Queensland, there is a good amount of research data for estimating the fire regime that will maintain the species richness of that area. As McFarland (1988a) concluded, for the Ground Parrot (*Pezoporus wallicus*) a species classified as ‘Vulnerable’ (Qld Nature Conservation Regulation 1994), the recommended fire regime to sustain this bird, was 8–10 years. However, it was discussed that with the lack of replicated sites, compounded by the limited distribution of the Ground Parrot, the results should be viewed cautiously. Interestingly, it was reported by McFarland (1988a) that in other studies (namely Fox 1978) the overwhelming majority of avian deaths (89%) were caused by fire suppression activities (ie backburning) whereas the wildfire accounted for the other 11%, providing managers with an interesting dilemma when they use fire for ecological purposes, as it may achieve the very opposite. It would seem that the combination of the work on fauna and fire-sensitive flora in Cooloola National Park would mean that a minimum fire-free interval of eight years is needed to maintain these populations.

Fox (1982) examined the successional patterns of mammals within a coastal woodland plot in the Myall Lakes National Park, in New South Wales. Similar to its effects on vegetation, fire may be the initial force which rejuvenates an area and initiates rapid and profuse recolonisation. The *possible* mechanisms by which fires can alter the community structure of mammals and thereby influence species richness include:

1. Frequent fire can create new habitats equally available to all species in the community and where they can immigrate;
2. Fires create sequences of microhabitats which are a function of time-since-fire (plant secondary succession), and these are preferentially preferred by mammals;
3. Frequent fires can create under-utilised habitats to be occupied by fugitive species, which are replaced by later mammal species,
4. Frequent fires may lead to the evolution of “fire-specialists” suited to exploiting the early seral stages;
5. Later species can enter the succession by displacing present species; and
6. Early successional species may be able to alter plant pyric succession (eg by grazing) to delay habitat changes favouring later successional species, prolonging mammalian succession.

Modified from Fox (1982)

As Fox (1982) shows, there is a distinct and regular pattern of recovery and colonisation following fire. It may be possible to determine these recovery patterns of rodent species in southeast Queensland. Once these patterns are known, these may provide some indication of the effectiveness of fire regimes. If an area is assessed and shown to only be representative of a lower seral stage of succession, then longer fire intervals may be needed. This is serving as an indicator species, and will be discussed in more detail in a later section.

Small mammal trapping within the sites showed that a range of responses occurred and that their responses (such as recolonisation) were correlated to the amount of time since fire, for many individual species. Fox (1982) identified four stages to recolonisation: first, dispersal from other areas results in the presence of transient individuals on burned sites. Later, these dispersers establish upon the burnt areas, and become colonisers. The third stage involves the colonisers successfully reproducing and the fourth and final stage is the increase in abundance associated with the recruitment of resident juveniles in addition to the original dispersers (Fox 1982). In accordance with similar studies, four species of *Pseudomys* were identified as fire- or disturbance enhanced species: *P. noveahollandiae*, *P. shortridgei*, *P. albocinereus*, and *P. gracilicaudatus* (Fox 1982). What is clear, as Fox (1982) states, is that the community composition changes drastically as a function of regeneration time, and that there is a need to maintain a mosaic of regeneration ages for management and maintenance of a diverse small mammal community. It would seem this applies also to the vegetation community as well. Catling (1986) studied the recolonisation habits of *Rattus lutreolus* following fire in a coastal heathland reserve in southeastern New South Wales. Interestingly, the results showed that, in this heathland, *R. lutreolus* switched from a late regeneration niche (ie usually some time after the disturbance) to an early niche, possibly due to lack of competition.

Much like the other studies summarised in this section, Catling (1986) stated that the abundance and species diversity of small mammals increased as habitats aged and become more complex (in structure etc.). Furthermore, as has been previously mentioned, the reduction in burning after the arrival of Europeans was proposed to be a link in the cause of many declines of pseodomyine species, and also, that too frequent burning may also lead to the same outcome (Catling 1986). Higgs and Fox (1993) confirmed the findings of the previously mentioned work, showing that following fire in the Myall Lakes Reserve, the (experimental) removal of *R. lutreolus* caused a significant increase in the abundance of *Pseudomys gracilicaudatus*, in both the wet and dry heaths. This also shows that following the initial destructiveness of the fire, it also creates a landscape where strong intra-specific competition can occur between many species, which (over time) allows for an increase in abundance of species. As a result, there is increased species diversity.

As the discussion above shows, there has been a considerable amount of relevant research work in heathlands, even in southeast Queensland. It would seem quite appropriate to suggest a minimum fire free period of 8 years to maintain biodiversity within heathlands. However, work is required to determine the upper limit of this inter-fire period. Maintaining good variability in these fire regimes is also quite important … and even though we find that an 8-year minimum fire-free period is desirable, some areas may also benefit from having fires at earlier stages. Therefore, some this community type, there are some gaps in our current knowledge.

General Fire Effects–Regimes, Fire Models, Fauna and Flora

It is worth discussing and exploring the merits of some work completed by fire researchers in other areas of Australia. But it is also pertinent to show that despite the great amount of information on fire effects, the long-term ecological effects of prescribed burning, fire exclusion and other important aspects of fire regimes are poorly understood. Fire ecologists have only started to initiate and implement research studies to determine the extent of these impacts.

Bradstock *et al.* (1998) developed a fire model to test the robustness of typical shrublands in Australia to different fire regimes. Obligate seeders were shown to be prone to higher rates of extinction (than resprouters) from prescribed burning. This extinction probability was also shown to positively relate to the frequency of unplanned fires (Bradstock *et al.* 1998). Fire models developed for Mountain Ash (*Eucalyptus regnans*) showed that the optimum interval for fires to equal 37–75 years (McCarthy *et al*. 1999). Recently, Gill and McCarthy (1998) demonstrated that to maintain a high level of biodiversity (irrespective of vegetation type) variability is required in fire intervals. Furthermore, and most importantly, the modelling establishes the urgent need for information on naturally-occurring fire-interval distributions, so that a statistical model of such distributions can be developed (Gill and McCarthy 1998). Simply, “it is apparent that there is no simple answer to the manager seeking a method to add greater variety to the intervals between prescribed fires but adoption of a variable system of fire application along with targeted monitoring…” (Gill and McCarthy 1998).

The change from ‘regular’ prescribed burning has only been recent. Gill and Williams (1996), like Kellman (1986), showed that regular frequent prescribed burning reduces biodiversity, however they also state that absence of fire may also reduce species richness. Morrison *et al.* (1996) presented the paradigm that the conflict exists between two management objectives, such that inter-fire periods greater than four years will allow a potentially severe fire hazard to exist, while a fire regime less than eight years will almost certainly result in a loss of biodiversity. Therefore, it was concluded that it would not be possible to achieve both hazard reduction and species conservation for any specified managed area, in this case, the dry–sclerophyll shrublands and woodlands of the Sydney region (Morrison *et al*. 1996).

In this review, the effects of fire on community vegetation types have been the primary focus. The main reason for this is as Hannah and Smith (1995) state: “the effects of repeated prescribed burning on fauna appears to be scant”. Woinarski and Recher (1997) provide a thorough review of the impacts of fire on the Australian avifauna. Even though the authors state that there are few detailed long-term examinations of fire and avifauna, present studies do permit identification of response patterns and the prediction of longer term effects. In general, almost all avifauna prefer less frequent fires, though there is still much contention and debate on fire frequencies (Woinarski and Recher 1997). The Ground Parrot is an example of this: early work suggested 4–5 year fire intervals, work completed in south-east Queensland demonstrated that populations were low less than three years or greater than 18 years post-fire; and intervals of 8–10 years (Queensland) and 10–25 years (Victoria) were recommended (Woinarski and Recher 1997). However, more studies suggested that fire exclusion may be a more appropriate regime for this bird (Woinarski and Recher 1997). Clearly, there are great differences depending upon the site where the study is conducted and Bradstock *et al*. (1998) affirms that site needs to be considered in the analysis of the results, as fire regimes for one site (eg. Sydney) may not be applicable to another (eg. southeast Queensland).

Sutherland and Dickman (1999) provide an extensive review of the recovery and response of rodents to fire. Along with many other studies, the review states that there are many areas of Australia where little or no information is available on fire responses. Furthermore, as Whelan (1995) discusses, there are inherent difficulties in generalising these studies where there is so much intrinsic variability to provide clear management direction (Sutherland and Dickman 1999). However, it is clear that Mark–Recapture studies of rodents, avifauna and herptiles may be useful to identify indicator species to explore of the impacts of prescribed fire regimes on biodiversity. Mark–Recapture methods have been proven to provide accurate estimations of population density, and can also provide estimates of birth and death rates for the population under study (Krebs 1986). Many models exist which use the principle of recapturing animals over a set period of time, and with the use of accurate assumptions about the population, an abundance of information can be revealed from a relatively simple methodology.

Fire Effects in Specific Ecosystems or Individual Species

EPIPHYTIC ORCHIDS

Cook (1991) investigated the effects of four fire regimes (ranging from different seasonal and annual, biennial and unburnt) on the survival and distribution of two epiphytic orchids, *Cymbidium canaliculatum* and *Dendrobium affine* in northern Australia. Similar to other parts of Australia, the effects and consequences of fire on epiphytic orchids have not been extensively studied. With a landscape dominated by some tall grass species (*Sorghum intrans*) fires are experienced through the area on a regular 2–3 year cycle (Cook 1991), a similar regime to that of the Greater Brisbane area until recently (Gourley *pers. comm.*). There were demonstrated differences between the fire responses of both orchids, such as the *Dendrobium* species have the ability to withstand some direct fire damage, allowing it to survive. The existence of pseudobulbs (for drought tolerance) also confers some orchids with the ability to withstand fires, which is a common trait of fire-adapted species (Cook 1991). More significantly, the ability of each species of orchid to survive becomes irrelevant if there are no host species for the orchid. The burnt plots had a 'relative lack' of juvenile *Eucalyptus tectifica* trees, which apart from confirming the devastating effects of fires on small trees, may in the longer term gradually reduce the number of trees and their availability for colonisation by the orchids (Cook 1991).

This may have many implications for the southeast Queensland region. Firstly, it has become very apparent that there must be **no** fires within the 3–5 year period following *any* (authors emphasis) fire (wild– or prescribed). Secondly, the potential use of indicator species (the epiphytic orchid is an excellent example) to determine the efficacy of fire regimes becomes problematical when the host species is probably the better choice. In the lack of appropriate scientific information, using the most obvious species (floral or faunal) within the specified region of study is highly recommended.

PITTOSPORUM UNDULATUM

Southeast Queensland (like much of populated Australia) has a large amount of urban bushland, which needs to be maintained to conserve species diversity, but also to limit the damage of habitat fragmentation, caused by expansion of human settlement. Rose (1997) investigated the invasion of the native *Pittosporum undulatum* into suburban forest edges of northern Sydney. It was discussed that European settlement and the associated change in fire regime and other activities, has allowed native species to invade areas that previously excluded them. Rose (1997) reported some strong edge effects, with older sites containing larger (but fewer) individuals, compared with higher densities of smaller plants further inside the reserves, which suggests an advanced successional stage for the edge area. With the proximity of these reserves near suburban areas, there is a markedly different fire management strategy with many reserves (particularly the smaller-sized ones) suppressing fire due to the life and property danger to local residents. This reduction in 'natural fire frequencies' since settlement by Europeans has been cited as a major factor to the spread of *P. undulatum* (Rose 1997). Conversely, for larger reserves, where active prescribed burning is used, it was discussed that such burning 'rarely extends' to the edge (Rose 1997), favouring a zone of fire-sensitive species on the edge (including *P. undulatum*) over fire-adapted sclerophyllous species.

The effects of fire on epiphytic orchids and the invasion of Pittosporum into forests of New South Wales may show some indication of inappropriate fire regimes. The notion of using certain faunal and floral species as indicators of the environment in response to some disturbance is not new. Indicator species of fire regimes has been lauded as one of the most important aspects that can answer the numerous questions about where and when to burn. This is the focus of the following discussion.

Fire and Potential Indicator Species

One of the purposes of this study was to identify ecologically sound fire regimes for the broad vegetation types listed as 'high importance' in the projects' infancy. Further, it was believed that in addition to sound fire regimes, indicators species would also be identified to use for potential study to determine the effectiveness of these fire regimes. Below is a section of different potential indicator species. However, it should be noted that personal discussions with prominent land managers and current and previous fire researchers have cast some doubt over the validity of such indicator species.

They question the use of one (or a few) indicator species, and as David McFarland (EPA–Biodversity) said recently:

> "…an indicator of what? If we are investigating the use of indicator species on inappropriate or correct (ecologically) fire regimes, then this becomes extremely difficult and hard to quantify and substantiate. Most of the time, an indicator species just indicates 'itself': what it does in response to some disturbance such as fire etc. Furthermore, I am very wary of the use of indicator species, especially the use of a *single* indicator species to judge the overall health of a community. This can be somewhat compensated through the use of multiple indicator species (choosing a gradient of species across that community) for a certain region, such as vertebrates, invertebrates, tree and ground fauna, or hollow-dependent fauna as a few examples. Lastly … the life-history characteristics of plants and animals needs to be known, *before* choosing any one species as an indicator of the community type in which it occurs."

David McFarland (*pers. comm*. June 2000)

This type of comment was common throughout most conversions with prominent researchers. Therefore, it should be duly noted that readers use the following information with caution and not place sole emphasis on using just one (or even a few) indicator species, when it may not represent the overall "health" of a community.

Burrows *et al.* (1999) investigated the use of 'appropriate' fire regimes to maintain biodiversity within the southwestern jarrah forests of Western Australia. It should be noted that this is almost exactly the same goal or outcome that the Southeast Queensland Fire and Biodiversity Consortium is trying to achieve. It was discussed that the ecological effects of fire are multi-dimensional and complex, and highly unlikely that there will be any complete understandings of fire effects on organisms and ecosystems. Using 'key indicators' of fire regimes is advantageous as it allow the (fast) development of fire management guidelines to achieve positive ecological and social outcomes (Burrows *et al*. 1999). Interestingly, the authors state that (potential) indicators may not necessarily be limited to fauna or flora species, but could include:

Climatic Indicators:

- Season and amount of precipitation,
- Frequency of drought, temperature and humidity regimes, and
- Frequency of lightning ignitions.

Use: In conjunction with fuel properties, these can be used to describe the 'fire-proneness' of the region, to identify the fire season and fire frequency ranges.

Historic Indicators:

- Traditional Aboriginal use of fire.

Use: Depending on level of knowledge and detail, this may assist in indicating the 'natural' (ie pre-European) fire regimes, which may provide boundaries for contemporary fire regimes.

Biological Indicators (Flora):

- Post-fire regeneration strategies,
- Post-fire floristic and structural changes,
- Fire sensitive taxa: typically these species have long juvenile periods, thin bark with crowns close to the fuel bed, and dependent on the canopy-stored seed for regeneration with a limited capacity for dispersal,
- The role of fire in seedling recruitment, particularly from fire-sensitive taxa, and
- Post-fire fuel (phytomass) accumulation rates.

Use: Regeneration strategies from plants may provide an indication of historical fire frequencies. Woody plant communities with a high proportion of species capable of resprouting generally indicates moderate frequencies (2–4 year intervals) of fire instead of a low (4–6 years) or high (2 years) frequency. It should be noted that these listed frequencies are very short and not applicable for southeast Queensland. However, the replication of a similar study for our region is highly recommended. Changes to the structure and floristics post-fire may be used to set fire interval ranges to maintain species richness and structural diversity. Juvenile periods of taxa that are fire sensitive may be used to set the minimum fire interval and may provide a biological basis for quantifying fire frequencies. Furthermore, seedling regeneration response could be used to set season and fire intensities. Post-fire accumulation rates could be used to (1) identify the minimum fire interval based upon the fuel available for burning; (2) the time after fire when phytomass productivity plateaus, and (3) identify the fire intervals to manage fuel levels within desirable (with respect to life and property values) limits.

Biological Indicators (Fauna):

- Fire sensitive taxa – with specialised habitat requirements (food, shelter and breeding), long juvenile periods, poor capacity for dispersal, high site fidelity and poor mobility,
- Post-fire response patterns of fire sensitive fauna, and
- Fire-proneness of refuge/shelter site of fire sensitive taxa

Use: Habitat requirements, juvenile period, fecundity, and site fidelity of fire sensitive taxa can be used to set fire intervals and season. Fecundity, dispersal capacity, site fidelity, mobility and post-fire population response patterns of fire sensitive taxa can be used to set fire size and patchiness. Nature of refuge/shelter can be used to set fire season and intensity.

Adapted and modified from Burrows *et al*. (1999)

Interestingly, by the conclusion of the study, it was revealed (at least for Jarrah forests), the most prominent indicators for determining appropriate fire regimes were:

- *Minimum intervals between fire to sustain biodiversity*: estimated from the juvenile period of the slowest maturing vascular plants (in particular obligate seeders) and from post-fire response patterns of fire sensitive taxa of flora and fauna,
- *Minimum interval between fires for wildfire control*: estimated from fuel accumulation and fire behaviour models, and a wildfire threat analysis,
- *Maximum interval between fires to sustain biodiversity*: estimated from plant species richness with time since fire, seed bank quantity and durability, vegetation/habitat biomass increment and structural changes with time, post-fire response patterns and habitat requirements of key fauna,
- *Season of fire*: estimated from requirements to regenerate habitat, requirements for patchiness of fire and extent of acceptable damage to habitat,
- *Fire intensity*: this will determine the level of acute physical impact of fire on vegetation, on habitat and on individual organisms. Generally, the magnitude of fire impact on organisms and habitat is directly correlated with the intensity and scale of the fire, and
- *Scale and patchiness of fire*: estimated from habitat requirements, life histories, and post-fire response patterns of key fauna. (adapted from Burrows *et al*. 1999)

The six prominent indicators outlined by Burrows *et al*. (1999) have high implications for southeast Queensland. It would be reasonable to implement research studies to examine these indicators and be provided with an accurate assessment of ecological impact of fire regimes. Some aspects, such as fire season, fire intensity and maximum fire intervals will require much longer time frames and many sites before an answer is available, but the other measures can be implemented now. It is recommended that these aspects outlined by Burrows *et al*. (1999) be adopted and instigated immediately for all regions in southeast Queensland.

INVERTEBRATES

Along with numerous other species (from faunal and floral origins) the effects of fire also affect ant species composition and behaviour. Andersen (1988) examined the extent of seed predation by ants following a fire in adjoining heathland/open woodland in Victoria. The results indicated that ant abundance and species richness was increased following the fire, the relative abundance of the species recorded was altered, and there were increased foraging populations of two ant species (Andersen 1988). However, the author states that the increase in ant species may be caused by the increased trapping efficiency, and noted in the few weeks post-fire, total numbers dropped in response to the increase in ground litter impeding the ant's ability to travel (Andersen 1988). The implications of finding the ant species is unknown, as Andersen (1988) says, "*too little is known about the biology of the … species concerned to give detailed explanations of the different responses to fire*." This emphasises the importance of McFarland's message above about the need to be wary of single indicator species after Andersen's conclusion to his work.

The effects of three different fire regimes (unburnt, annually and biennially burnt) on ground foraging ant communities in northern Australia were investigated by Andersen (1991). The different fire regimes had a large influence on the communities of ground-foraging ants, whereby the annually burnt and unburnt communities produced two distinctive communities (Andersen 1991). It was discussed that the changes represented large-scale changes in overall community organisation rather than just changes in species composition. Interestingly, the biennially burnt plots (ant species) resembled an intermediate of the unburnt and annually burnt areas (Andersen 1991). Ants have been purported as indicator species as they can find safe refuge, but are also unable to disperse away from the fire front, indicating they are usually the first species to reinhabit a recently burnt area. Even though fire has direct effects on ants, usually they can find refuge in soil nests; the largest influence of fire on ants is indirect. Fire induced modifications to habitat, food supplies, and interspecific competition are some of the major indirect influences of fire on ant species (Andersen 1991). This is represented in Andersen's (1991) results: in annually burnt areas, a mid-story is (virtually) absent, ground layer is sparse, and there is a considerable amount of bare ground which were ideal conditions for the two dominant ant species recorded in trapping. For the unburnt sites, the well-developed mid-story and ground layer of leaf litter, is dominated by myrmicines, which are species that can tolerate a broader range of physical conditions (Andersen 1991). The other reason why ants may make good indicator species is: that they are closely linked with soils, vegetation and other fauna and make up one of the most important group of animals in the savanna (and most likely in other ecosystems) (Andersen 1991). Therefore, any effects of fire on ants are also indicative of more broad-scale community level responses. It was reported that the effects of fire on ants correlated with the effects on other faunal groups (Majer 1983). Furthermore, ants are highly ubiquitous, abundant and highly active (York 1994). Coupled with their functional importance, sensitivity and responsiveness to changes in system structure makes them useful indicators of environmental status and condition (York 1994). Additionally, ants are relatively easy to sample and sort, and these characteristics make them ideal for ecosystem health indicators (Vanderwoude *et al*. 1997a and b). Invertebrates comprise almost 99% of the Animal Kingdom and as such, are an essential part of the ecosystem (Stanisic 1998).

York (1994) conducted some very interesting research in a central New South Wales region (Myall Lakes) to determine if some long-term fire management practices affected the ant communities and how this reflected on biodiversity (and its conservation). York (1994) showed that in the long-term, species diversity declines after fire, and that fire-intervals (or time-since-fire) exert the strongest influence on the richness of any species. Even though the research showed that frequent fire practices *maintain* (not diminish species diversity), management for biodiversity must consider between-habitat (or *beta*) diversity (York 1994). Not surprisingly, the beta-diversity in the frequently burnt sites was low (York 1994). More importantly, the number of rare species found across the broad range of fire-impacted habitats, showing no clear pattern of distribution, appeared to be linked to forest age (York 1994). Overall, ant species (and possibly overall-biodiversity) may be reduced with the simplification of the forest community. Therefore, there needs to be maintenance of habitat mosaics, reflecting different time since fire intervals (York 1994). Unfortunately, like most (inhabited) areas throughout Australia, prescribing 'natural' fire regimes and 'let-burn' strategies for wildfires in areas of human use becomes impractical and the preference for regular fire will ultimately lead to a reduction in forest complexity and (as has been demonstrated) biodiversity (York 1994).

Neville (1999) examined the effects of both prescribed fire and wildfire on epigaeic invertebrates in wet sclerophyll forests within the Dandenong Ranges of Victoria. There was an extensive pre-burn assessment (1.5 years) of the area, which allowed determination of any seasonal patterns allowing the impact of fire to be examined in detail. The author argues that despite the many studies on fire and ant responses, they lack pre-fire data to use as a baseline for meaningful comparisons. A strong seasonal pattern emerged from the results, with very low activity in both burnt and unburnt plots during winter, but much higher activity in summer, but especially so in the burnt areas (1.5–2 times increased activity) over unburnt controls (Neville 1999). Wildfires showed (surprising) less overall impact than prescribed burns. Though this was a study of one fire (both wildfire and prescribed burn) the effects show that species composition changes are well reflected in the invertebrates samples and the functional groups used to separate each species (Neville 1999). Furthermore, the use of the attributes from functional groups showed the importance of lower level identification, (in this case, to *genus* level) (Neville 1999).

Norris and Convoy (1999) completed an extensive study on the effectiveness of various trapping methodologies and the effects of different fire intensities on invertebrates in numerous sites in New South Wales. As they rightly state "(knowledge) of the overall effects of … fire regimes on ecological processes is required before informed (and holistic) management decisions are possible … (and only) continued research in a variety of ecosystems will enhance knowledge of fire ecology" (Norris and Convoy 1999). After some exhaustive analysis, it was shown that populations of invertebrates from the *Acarina* and *Collembola* (active within organic matter decomposition) declined significantly following 'moderately intense fires' and may be considered fire sensitive taxa (Norris and Convoy 1999), and may affect litter decomposition rates. *Coleopteron* populations were not significantly affected by low intensity burning, and populations from *Diptera* and *Hymenoptera* showed short-term increases in activity after moderate intensity fires (Norris and Convoy 1999). Despite some short-term alterations to the populations of invertebrates, 6–12 months following the initial fire, greater diversity was seen in all sites, with low intensity burning seemingly not having a consistent effect on soil and leaf litter invertebrate diversity (Norris and Convoy 1999).

In comparing which sampling method produced the best outcomes, Norris and Convoy (1999), stated that water and pitfall traps were indicators of invertebrate activity tending to collect more *Diptera* and *Hymenoptera*, whereas the litter sampling technique provided more direct results regarding invertebrate abundance, and more effective for the *Acarina* species. The numbers of Collembolans varied with the three techniques which also depended upon seasonal changes (Norris and Convoy 1999). Van Heurck *et al*. (1999) performed a similar experiment investigating the impacts of fire on invertebrates in the jarrah forests of Western Australia. According to the authors, there have been 'few other studies' attempting to link the impact of fire and its changes in the composition of litter invertebrates through trophic guilds or biomass production to the functioning of these invertebrates or the ecosystem in general (van Heurck *et al*. 1999). Van Heurck *et al*. (1999) used the idea of assigning phases to each fire regime to validate the results statistically. These four phases were: (i) "early acute" phase: less than one year post-fire; (ii) "acute" phase: 1–2 years post-fire; (iii) "chronic" phase: 2–4 years post-fire; and (iv) complete post-fire recovery: up to and including four years post-fire (van Heurck *et al*. 1999). This would also mean, that a five year inter-fire frequency is observed in this very flammable vegetation community. The results indicated that the unburnt sites contained more species (overall) than the burnt areas, though this was not statistically significant. There was also a strong seasonality in the results, with the spring and autumn results varying the most. The taxa recorded were typical of unburnt sites (Aranea, Acarina, Blattodea and Thysanoptera) were also the most significantly affected by fire (both +ve and –ve) (van Heurck *et al*. 1999). Post-fire analysis showed that as mentioned many times previously, the fires were very patchy, leaving some microhabitats of unburnt areas which may have served as refuges from the fire. The overall impacts (for Jarrah forests) are: decomposer and small herbivore taxa are favoured by changes in the 'acute' phase; large herbivores (while showing declines after fires) recover (and sometimes increase) by the onset of the 'chronic' phase. Further, large predators and pollinators also have immediate declines but remain at low numbers by the 'chronic' phase; and large decomposers (such as Blattodea and Pauropoda) increase in species at the 'chronic' stage. As van Heurck *et al*. (1999) state, the richness and abundance of the majority of the taxa mirror fire responses. More specifically, the results indicated that monitoring the impacts on Coleopteron (beetles) may be able to provide a link between beetle diversity and the overall ecosystem (within) the jarrah forest (van Heurck *et al*. 1999). The presence of beetle morphospecies specific to autumn, spring or long unburnt sites suggest that mosaic burning regimes could contain and maintain a complete regional beetle fauna (van Heurck *et al*. 1999), and possible overall biodiversity.

The measurement of the type and abundance of invertebrates as an effective indicator of disturbance has been demonstrated. More importantly, it seems that the species identified following disturbance is similar throughout the Australian continent. This gives researchers in southeast Queensland some likelihood that similar species will be recorded after fire. If similar patterns of colonisation by ants are found, the implications of using ants as potential indicators are enormous. Below is a more indepth discussion on the role of ants as indicators in Queensland, and the types of research completed in this area.

SOUTHEAST QUEENSLAND FOCUS

For the local region, there has been a reasonable amount of scientific research investigating the effects of fire on ants and the use of ants to indicate these changes.

Similar to York's (1994) work, Vanderwoude *et al*. (1997a) examined the composition of ant communities over a 12-month period in a southeast Queensland open forest (Bauple), where the study sites were affected by three fire regimes (burnt annually since 1952, periodically burnt since 1973, and unburnt since 1946). The results showed that site species richness to be 74, 63, and 43, respectively at the annually burnt, periodically burnt and unburnt sites. Vanderwoude *et al*. (1997a) explained that there were discernible differences between each site in terms of biogeographical and functional composition and between each site (burnt and unburnt) in terms of abundance and species richness. The low(er) amount of species recorded at the unburnt site [probably reflected an area devoid of fire] and represented an area similar to a highly disturbed one (House *pers. comms*. June 2000). The study did represent that ants were indicative of ecosystem change and shown to be applicable to southeast Queensland. However as Vanderwoude *et al*. (1997a) stated, the use of invertebrates as bio-indicators is most effective when supported by detailed knowledge of the community dynamics of the local fauna.

More specifically related to this project, Vanderwoude *et al*. (1997b) assessed the effectiveness of ant communities as indicators of fire management in Spotted Gum (*Eucalyptus maculata*) forests of southeast Queensland. This study incorporated the same site analysis and fire regimes described above. Again, the results indicated that increasing fire frequency resulted in species richness increases, reflecting changes in the vegetation composition, comparable to the one previously described above (Andersen 1991). The results of this study closely aligned with other work, and basically stated, the relative abundance of ants is inversely related to the frequency of burning (Vanderwoude *et al*. 1997b). It does not mean that burning frequently means higher diversity (as the results may suggest) but it is the overall species composition which is important. Frequently burnt areas typically show more open areas with relatively minor canopy cover, and the opposite is true for less frequently burnt areas. Ants will occupy highly disturbed sites (such as mines) and once vegetation starts to reappear, the numbers ***and*** composition of ant species changes accordingly. The well-documented functional groups of ants that occupy each different successional change (Andersen 1995) may be quite applicable to monitoring the effectiveness of a fire regime.

Stanisic (1998) identified the possible use of the land snail as an indicator. Ants are relatively uncommon in wetter areas such as rainforest and wet-sclerophyll boundaries, so viable alternatives are needed. Stanisic (1998) stated that land snail diversity in Queensland is particularly high, with as much as 90% of land snails living in the Eastern Rainforest biome. The diversity of land snails is sharply and consistently higher in rainforest than in adjacent eucalypt/woodland habitats for some northern New South Wales and southeast Queensland rainforest (Stanisic 1998). Similarly, there is a rich diversity of land snails in tropical rainforest, subtropical rainforests, and dry rainforest (not well documented in Qld, but in other states), which is due to the close (ecological) association between snails and rainforest (related to moisture-sensitivity), whereas eucalypt forests are depauperate in land snails (Stanisic 1998). Wardell-Johnson and Nicols (1991) has mentioned that some areas in southwest of Western Australia have been surveyed to determine the effects of disturbance (including fire) upon the region, and land snails were used. Furthermore, as Stanisic (1998) stated, limestone outcrops are the second significant habitat for the land snail. The presence of land snails in rainforests and not in eucalypt woodland would make them ideal environmental indicators. Similar to the other advantages of using invertebrates as potential indicators (above), land snails are also easy to distinguish (shell shape, size and colour etc.). Another distinctive advantage of land snails, is that their shells generally persist in litter (where they inhabit) for some time after death, allowing for *post mortem* analysis (Stanisic 1998). Whilst no direct studies have been conducted on land snails and fire (other than by Stanisic) some work has been completed on fire effects on land snail habitats, including vine thickets, where fire has strong influences. This is certainly an area, which warrants further study. As Stanisic (1998) states, “The (lack) of detailed studies *seriously* impedes current efforts to draw attention of land managers to the threatening processes of perennial burning…and there is increasing evidence that past practices have had damaging effects and more environmentally sympathetic approaches are required.”

Therefore it is clear from these previous studies that ants have been shown to be effective and accurate in determining the level of effectiveness of fire regimes, with respect to biodiversity values. There is a well-recognised pattern of species recovery and change is species composition as an area ‘rehabilitates’ from the ‘disturbance’. The recovery pattern seems relatively consistent across the Australian continent and studies in southeast Queensland have shown this. It may therefore be quite plausible to use invertebrates as indicator species of fire regimes for most forest types in southeast Queensland. However, it should be noted that this must be used in addition to other measures (ie other indicator species), and not be the only indicator of ecosystem health.

OTHER POTENTIAL SPECIES

McFarland (1998) mentioned for the southeast Queensland region, the selection of gliders and large owls (*Petaurus australis*, *Petauroides volans*, *Ninox strenua* and *Tyto tenebricosa*) as possible indicator species. “These species are considered because of their ecology type (type of forest used, home range size, trophic level, and hollow-dependency) and response to habitat change (mostly logging) or association with unlogged forest" (McFarland 1998). Furthermore, these species are considered as they are recognised as taxa which are sensitive to change, and disturbance and any management aimed at these species usually results in the conservation of a wide range of sympatric forest species (McFarland 1998). This probably applies also to sensitivity to fire regimes, particularly if large tracts of forests are destroyed in large conflagrations. Even though there is some conflicting evidence for the disturbance sensitivity of two species (*P. australis*, and *N. strenua*) there is no argument that these species are forest dependent and require large areas of forest, and any subsequent protection may confer benefits to other species (McFarland 1998). As McFarland (1998) rightly states, these species become not indicator, but umbrella species. How managing for sound fire regimes fits in here can become challenging. It has been well documented that the responses by some bird species can indicate the efficacy of fire regimes within a certain area. Research combining life-history characteristics and responses to experimentally induced fire regimes will provide enough data to warrant the use of these species as indicator (or even umbrella) species or to determine whether the use of these species is warranted.

Plant and animal species classed as 'Rare' or 'Threatened' according to the *Queensland Nature Conservation Act 1992* are also potential indicator species, purely for the reason, that they rate highly in terms of conservation. Tables 7 and 8 (plants species followed by faunal species) which is a summarised list compiled of all the rare and threatened species in southeast Queensland for possible consideration:

Table 7: Documented Rare and Threatened Species recognised from the Regional Forestry Agreement Flora Study (Qld CRA/RFA Steering Committee 1998a)

Botanical name	QNCR[1]	ESP[2]	Reports	SMP[3]	Current Study
Acacia attenuata	V	V		D	OL
Acacia baueri subsp . *baueri*	V	V			OL
Acacia grandifolia	V	V		D	OL
Acacia perangusta	V	V		D	OL
Acacia sp. (Binjour)	E#			P	
Alectryon ramiflorus	E	E	4,7	D	
Allocasuarina rigida subsp. *exsul*	V#			P	S
Archidendron lovelliae	V	V	3	P	
Arthraxon hispidus	V	E	8	P	OL
Boronia keysii	V	V	8, 10	P	
Bothriochloa bunyensis	V	V		P	OL
Clematis fawcettii	V	V	8	P	OL
Cossinia australiana	E		4	D	
Cupaniopsis shirleyana	V	V	4	P	
Cycas megacarpa	V	E		D	OL
Daviesia discolor	V	V		P	S
Dodonaea rupicola	V	V		D	S
Eucalyptus conglomerata	E	E	5	P	
Eucalyptus hallii	V	V	12	D	OL
Eucalyptus taurina	V#				OL
Floydia praealta	V	V	4, 11	P	
Fontainea rostrata	V	V	4	D	
Fontainea sp. (Binjour)	E#			D	
Fontainea venosa	V	V	4		
Haloragis exalata subsp . *velutina*	V	V		P	OL
Lasiopetalum sp. (Proston)	E#				OL
Leucopogon recurvisepalus	E			D	S
Macadamia integrifolia	V	V	4	P	
Macadamia jansenii	V	E	4	P	
Macadamia ternifolia	V	V	4	P	
Macadamia tetraphylla	V	V	4, 11	P	
Macrozamia lomandroides	E	E		D	S
Macrozamia parcifolia	V	V		D	S
Macrozamia pauli-guilielmi	E	E		D	S

Marsdenia coronata	V	D		P	OL
Medicosma elliptica	V	V		P	OL
Notelaea lloydii	V	V		P	OL
Oldenlandia sp. (Wietalaba)	E#			P	S
Parsonsia kroombitensis	V#			D	
Parsonsia larcomensis	V#	V		D	
Paspalidium grandispiculatum	V	V		P	OL
Phaius australis	E	V	8, 13	D	
Phaius tancarvilleae	E	E	8, 13	D	
Phebalium obtusifolium	V		6	D	
Plectranthus nitidus	E	E		P	OL
Plectranthus omissus	E	E		P	OL
Plectranthus torrenticola	E	E		D	OL
Prostanthera sp. (Mt Tinbeerwah)	V	V		P	S
Pterostylis chaetophora	V#				
Quassia bidwillii	V	V	4	P	
Rhodamnia sp. (Calliope)	E#				S
Romnalda strobilacea	V	E	4	P	
Sarcochilus fitzgeraldii	E	V		P	
Sarcochilus roseus	V	V	11	P	
Sarcochilus weinthalii	E	V	11	D	
Sophora fraseri	V	V	11	D	OL
Thelypteris confluens	V#				
Thesium australe	V	V	8, 14	P	
Triunia robusta	E	E	4	P	
Xanthostemon oppositifolius	V	V	4, 8	D	
Zieria sp. (Binjour P.I.)	E#				

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Legend to Terms Used in Table 7

# Pending inclusion on the schedule of the Queensland Nature Conservation (Wildlife) Regulation
OL = Species outlined in RFA (1998) report
S = Species surveyed Sept to Nov 1997 and outlined in the RFA (1998) report
D = Draft Species Management Profile completed
P = Draft Species Management Profile in progress
V = Classified as Vulnerable according to *Qld Nature Conservation Act 1992* (1994)
E = Classified as Endangered according to *Qld Nature Conservation Act 1992* (1994)

Table 8: Documented Rare and Threatened Species in SEQ (McFarland 1998)

Scientific Name	Common Name	Scientific Name	Common Name
Ornithoptera richmondia	Richmond Birdwing Butterfly	*Ptiloris paradiseus*	Paradise Riflebird
Argyreus hyperbius inconstans	Australian Fritillary Butterfly	*Sericulus chrysocephalus*	Regent Bowerbird
Acrodipsas illidgei Illidge's	Ant–blue Butterfly	*Poephila cincta cincta*	Black–throated Finch (sth subsp.)
Jalmenus evagoras	Eubulus butterfly	*Ornithorhynchus anatinus*	Platypus (Management Interest)
Nesolycaena albosericea	Satin Blue Butterfly	*Antechinus swainsonii*	Dusky Antechinus
Galaxias olidus	Mountain Galaxias	*Dasyurus hallucatus*	Northern Quoll
Adelotus brevis	Tusked Frog	*Dasyurus maculatus maculatus*	Spotted–tailed Quoll (sth subsp.)
Assa darlingtoni	Australian Marsupial Frog	*Phascogale tapoatafa*	Brush–tailed Phascogale
Crinia tinnula	Wallum Froglet	*Phascolarctos cinereus*	Koala
Hiloria kundagungan	Red–and–yellow Mountain–Frog	*Petaurus australis australis*	Yellow–bellied Glider (sth subsp.)
Philoria loveridgei	Masked Mountain–Frog	*Petaurus norfolcensis*	Squirrel Glider
Lechriodus fletcheri	Black–soled Frog	*Petauroides volans*	Greater Glider
Limnodynastes salmini	Salmon–striped Frog	*Pseudocheirus peregrinus rubidus*	Common Ringtail Possum
Mixophyes fleayi	Fleay's Barred–Frog	*Cercartetus nanus*	Eastern Pygmy–possum
Mixophyes iteratus	Giant Barred–Frog	*Aepyprymnus rufescens*	Rufous Bettong
Rheobatrachus silus	Southern Platypusfrog	*Potorous tridactylus*	Long–nosed Potoroo
Taudactylus diurnus	Southern Dayfrog	*Macropus agilis*	Agile Wallaby
Taudactylus pleione	Kroombit Tinkerfrog	*Macropus dorsalis*	Black–striped Wallaby
Litoria brevipalmata	Green–thighed Frog	*Petrogale herberti*	Herbert’s Rock–wallaby
Litoria cooloolensis	Cooloola Sedgefrog	*Petrogale penicillata*	Brush–tailed Rock–wallaby
Litoria sp. cf cooloolensis	Frog (no common name)	*Thylogale stigmatica*	Red–legged Pademelon
Litoria freycineti	Wallum Rocketfrog	*Wallabia bicolor welsbyi*	Swamp Wallaby
Litoria olongburensis	Wallum Sedgefrog	*Nyctimene robinsoni*	Eastern Tube–nosed Bat
Litoria pearsoniana	Cascade Treefrog	*Pteropus alecto*	Black Flying–fox
Litoria revelata	Whirring Treefrog	*Pteropus poliocephalus*	Grey–headed Flying–fox
Chlamydosaurus kingii	Frilled Lizard	*Scoteanax rueppellii*	Greater Broad–nosed Bat
Phyllurus caudiannulatus	Banded Leaf–tailed Gecko	*Scotorepens sanborni*	Northern Broad–nosed Bat
Phyllurus sp	'Oakview' gecko	*Scotorepens sp.*	(Parnaby) bat
Hemiaspis damelii	Grey Snake	*Vespadelus darlingtoni*	Large Forest Bat
Hoplocephalus bitorquatus	Pale–headed Snake	*Vespadelus regulus*	Southern Forest Bat
Hoplocephalus stephensii	Stephen's Banded Snake	*Vespadelus troughtoni*	Eastern Cave Bat
Pseudechis guttatus	Spotted Black Snake	*Vespadelus vulturnus*	Little Forest Bat
Simoselaps warro	Snake (no common name)	*Pseudomys novaehollandiae*	New Holland Mouse
Ramphotyphlops broomi	Blind snake (no common name)	*Pseudomys oralis*	Hastings River Mouse
Ramphotyphlops silvia	Blind snake (no common name)	*Pseudomys patrius*	Eastern Pebble–mound Mouse
Lophoictinia isura	Square–tailed Kite	*Xeromys myoides*	False Water–rat
Accipiter novaehollandiae	Grey Goshawk	*Pteropus scapulatus*	Little Red Flying–fox
Erythrotriorchis radiatus	Red Goshawk	*Miniopterus schreibersii*	Common Bentwinged Bat
Falco hypoleucos	Grey Falcon	*Myotis moluccarum/macropus*	Large–footed Myotis
Rallus pectoralis	Lewin’s Rail	*Nyctophilus timoriensis*	Greater Long–eared Bat
Turnix melanogaster	Black–breasted Button–quail	*Saltuarius swaini*	Gecko (no common name)
Geophaps scripta scripta	Squatter Pigeon (sth subsp.)	*Delma plebeia*	A legless lizard
Ptilinopus superbus	Superb Fruit–Dove	*Delma torquata*	Collared Delma
Calyptorhynchus lathami	Glossy Black–Cockatoo	*Paradelma orientalis*	Brigalow Scaly–foot
Cyclopsitta diophthalma coxeni	Double–eyed Fig–Parrot (Coxen's)	*Varanus semiremex*	Rusty Monitor
Lathamus discolor	Swift Parrot	*Anomalopus leuckartii*	Skink (no common name)
Psephotus pulcherrimus	Paradise Parrot	*Calyptotis lepidorostrum*	Skink (no common name)
Neophema pulchella	Turquoise Parrot	*Cautula zia*	Skink (no common name)
Ninox strenua	Powerful Owl	*Coeranoscincus reticulatus*	Three–toed Snake–tooth Skink

Table 8 continued …

Scientific Name	Common Name	Scientific Name	Common Name
Tyto tenebricosa	Sooty Owl	*Coggeria naufragus*	Satinay Sand Skink
Tyto novaehollandiae	Masked Owl	*Ctenotus arcanus*	Skink (no common name)
Podargus ocellatus plumiferus	Marbled Frogmouth (Plumed)	*Ctenotus eurydice*	Skink (no common name)
Menura alberti	Albert's Lyrebird	*Egernia rugosa*	Yakka Skink
Atrichornis rufescens	Rufous Scrub–bird	*Eremiascincus richardsonii*	Broad–banded Sand Swimmer
Climacteris erythrops	Red–browed Treecreeper	*Eroticoscincus graciloides*	Elf Skink
Dasyornis brachypterus	Eastern Bristlebird	*Lampropholis colossus*	Skink (no common name)
Pyrrholaemus brunneus	Redthroat	*Lampropholis couperi*	Skink (no common name)
Xanthomyza phrygia	Regent Honeyeater	*Menetia timlowi*	Skink (no common name)
Syconycteris australis	Common Blossom–bat	*Nangura spinosa*	Nangur Skink
Macroderma gigas	Ghost Bat	*Ophioscincus cooloolensis*	Skink (no common name)
Taphozous australis	Coastal Sheathtail–bat	*Ophioscincus ophioscincus*	Skink (no common name)
Taphozous georgianus	Common Sheathtail–bat	*Ophioscincus truncatus*	Skink (no common name)
Mormopterus norfolkensis	Eastern Freetail–bat	*Saiphos equalis*	Skink (no common name)
Hipposideros semoni	Semon's Leafnosed–bat	*Saproscincus challengeri*	Skink (no common name)
Chalinolobus picatus	Little Pied Bat	*Saproscincus galli*	Skink (no common name)
Chalinolobus dwyeri	Large–eared Pied Bat	*Saproscincus rosei*	Skink (no common name)
Falsistrellus tasmaniensis	Eastern False Pipistrelle	*Acanthophis antarcticus*	Common Death Adder
Kerivoula papuensis	Golden–tipped Bat	*Denisonia maculata*	Ornamental Snake
Miniopterus australis	Little Bentwinged Bat	*Furina barnardi*	Yellow–naped Snake
Melithreptus gularis	Black–chinned Honeyeater	*Furina dunmalli*	Dunmall's Snake
Grantiella picta	Painted Honeyeater	*Lichenostomus melanops*	Yellow–tufted Honeyeater
Pachycephala olivacea	Olive Whistler		

As Tables 7 and 8 shows, there are already a high number of plants and animals already identified as being under-threat. Habitat loss, fragmentation, urban encroachment are attributed to this (McFarland 1998), but inappropriate fire regimes were also recognised as an important influencing factor, although no work was actually completed to ascertain this level of impact during the Regional Forestry Agreement process, from where this data was collated (McFarland *pers. comm.*). It is important that before any one (or more of these species) it is vital that information on the life history characteristics of each species is known. For most of the species listed here, this type of information is not known or only some aspects are known. This can be quite labour and resource intensive and it may be more applicable to aim towards assessing fire regimes using some more broad categories, such as those outlined by Burrows *et al*. (1999). Once this type of information is known, the next level of research should concentrate on the species listed above.

From McFarland (1998), for the SEQ CRA – RFA process, it was shown (Table 9) that the faunistic surveys showed that the region showed species diversity comparable to the Wet Tropics and Cape York Peninsula, highlighting the importance of forests within the region. These regions which have high species diversity are largely influenced from the large altitudinal gradients (such as upland and lowland rainforest and sclerophyllous forest types) (McFarland 1998). Furthermore, from all the bioregions that have been examined, southeast Queensland contained the highest percentage of threatened vertebrate taxa, but only a moderate level of endemism.

Table 9: Bioregions in Queensland, Terrestrial Native Forest/Woodland-Dwelling Vertebrates

Bioregion	Number of Species				% Terrestrial Species		Source
	Amphibian	Reptile	Bird	Mammals	Endemic	Threatened	
Cape York Peninsula	44	139	238	86	17.4	7.3	McFarland (1993)
Wet Tropics	50	122	223	98	16.0	13.6	Williams *et al.* (1993)
Brigalow Belt (North & South)	29	145	220	65	3.1	8.9	Smyth (1997)
Mulga Lands	20	69	141	37	2.6	5.6	Wilson and Egan (1996)
Channel Country	17	99	133	27	6.9	3.6	McFarland (1992)
Southeast Queensland	49	148	242	87	5.9	14.3	McFarland (1998)

NB: Rarity defined as taxa listed as Endangered, Vulnerable or Rare in the Nature Conservation Act (1992), with the Brigalow Belt Reptile Category contain Non-Forest Species (modified from McFarland 1998)

Appendices 1 and 2 (pages 88 and 92 respectively) contain an extensively compiled list of plants identified throughout the region which are worthy of consideration as potential indicator species. In particular, Appendix 1 is very relevant to many local authorities (Appendix 2 used information from National Parks), as it contains a number of 'indicative species' outlined in the Regional Forestry Agreement process for some key areas in terms of biodiversity and conservation within southeast Queensland. Importantly Appendix 1 also contains a description of the forest type in which these species were identified – so these can be applied directly to each area under examination.

Fire Effects on Fauna

GENERAL EFFECTS

Christensen and Kimber (1975) provide a good overview of the effects of prescribed burning on fauna, especially small mammals, macropods and birds in wet and dry sclerophyll forests of southwest Australian. Interestingly, regeneration after fire in the wet-sclerophyll forests was predominantly through soil-stored seed (similar to southeast Queensland, Hall *pers. comm.*) and from rootstocks in the drier forest. The results showed that the (temporary) removal of vegetation following the fire resulted in the disappearance of small mammals, but re-colonisation quickly took place once this vegetation structure has recovered (2+ years post-fire) (Christensen and Kimber 1975), whereas this was less evident for larger macropods. Furthermore, birds showed little change post-fire, and actually increased in numbers following the fire (again in the 2+ years after the fire) but settled to 'normal' population levels thereafter.

As Christensen (1998) points out, the survival of mammals and birds from fire is largely a function of the fire intensity. It has been clearly documented, that mortality for highly mobile species, such as wallabies, kangaroos, and even large birds can be substantial in high-intensity wildfires, but much reduced for fires of lesser intensity (Christensen 1998). Furthermore, recovery rates can also be very wide-ranging, from immediately post-fire, but sometimes up to 20–40 years, and is dependent upon many factors including:

- For birds, seed production from food plants (eg Ground Parrot, *Pezoporus wallicus)*,
- Nectar production of plants,
- Amount and percentage of vegetation cover,
- Mammals may require different combinations of vegetation cover and food (eg Rufus hare wallaby, *Lagorchestes hirsutus*), and
- Some mammals may require particular 'fire-dependent' food (eg, the association between hypogean fungi and the woylie, *Bettongia peniciallata*).

(Modified from Christensen 1998)

There is no question that the responses of animals to fire is extremely complex and varies with (and between) species, as well as with fire characteristics (such as intensity, patchiness etc.) (Christensen 1998). The optimum fire regime is still unknown for most species (classed as 'vulnerable' or 'endangered' according to CWR: Critical Weight Range system) – and it is clear that no 'blanket' prescription can suit all species (Christensen 1998). Furthermore, as Christensen (1998) states there is no longer fires that resemble a 'natural' fire, especially with the existing patchwork of forested areas and different land-uses, means that fires started naturally (ie lightning) no longer behave naturally. He criticises the (notions of the) cessation of any burning to bring back the natural fire regime as this won't occur, but also does not concur with the current *status quo* of the current fire management procedures. Christensen's (1998) recommendations sound very familiar:

- Location of populations of each species (in the CWR, ie endangered/vulnerable) as well as potential habitats will allow immediate management decisions to be made on the best available information, which may involve local changes to current burning (and grazing) regimes,

- At the same time, *studies designed to fill the knowledge gaps* (authors emphasis) should be set in progress, with two primary level of details including:

    - Further field work to identify populations of each endangered/vulnerable species (including suitable habitats), and
    - Experimental-based studies on the effects of burning and grazing on these selected species.

(Modified from Christensen 1998)

Friend (1993) completed a comprehensive review of the effects of fire on small vertebrates in the mallee woodlands of Western Australia, in which, it was discussed that the impacts of fire have not been adequately explained in relation to life-history patterns and habitat needs of animal species. Once this type of information is elucidated, these patterns and relationships can be used to predict faunal responses to fire, and this can facilitate the development of appropriate strategies to conserve/maintain these species (Friend 1993). "This paper addresses some of the above *gaps in our knowledge* by reviewing the impact of fire on small vertebrates and examining species' response patterns in relation to their shelter and dietary requirements, and their reproductive patterns" (Friend 1993). To mention that this type of information is needed in southeast Queensland is an understatement. Below is a summary of Friend's (1993) review, of the following impacts on small mammals, reptiles and amphibians in three categories (acute impacts, seral responses, and shelter, diet and reproductive patterns). Please consult the article for the full description of impacts.

SMALL MAMMALS:

Acute Impacts: Marked decline (Antechinus spp.), but many survived due to patchiness of fires (even highly intense fires), and in gullies etc. Two months following fire, native and introduced predators easily preyed upon the surviving animals.

Seral Responses: For mallee woodland, there seems to be a reasonably consistent successional response. Immediately after the fire, the exotic *Mus domesticus* and *Pseudomys* spp. colonise the area. *Pseudomys* spp. also tend to characterise the early post-fire successional stages, though some species (particularly rare species such as *P. shortridgei*, and *P. occidentalis*) require vegetation that has not been burnt for 40+ years or more. At the mid-successional stage (no actual timeframe is included) there is an increase in abundance of dasyurids (primarily *Sminthopsis* spp.) following decline in numbers of *Mus* and *Pseudomys* spp. respectively. As the dasyurids fall in number, the larger *Rattus* spp. dominate the area (though this is a less consistent trend as it has been demonstrated that some rare species of *Antechinus* return to areas 2–4 years after fire).

Shelter, Diet and Reproductive Patterns: Generally, this is a consistent trend. For species occupying the early successional stages, they all construct burrows (of varying complexity), with distinct preferences for species-rich areas such as heath. Diets are quite generalised, and they exhibit high reproductive potential. These characteristics are typical of opportunistic breeders. Species in the mid-successional stages such as *Sminthopsis* spp. occupy hollow logs and other 'flammable' microhabitats for shelter, with more specialised carnivorous diets and polyoestrous (many breeding cycles) reproductive patterns. This can facilitate the production of two litters during the spring/summer breeding season. Later successional stage species are far less flexible, with many specific characteristics. This would indicate the need for long unburnt patches of forest. Habitats needed include: hollow logs in damp areas, with thick cover of ground litter, and a more specialised diet than the small rodents. Furthermore, they had a rigid monoestrous breeding pattern (in later winter/early spring) where males (in *Antechinus* spp.) die after mating. Therefore stability and resource predicability for these species needs to be relatively high.

REPTILES:

Acute Impacts: Compared to small mammals, reptiles are more resilient to the short-term impacts of fire. This resiliency relates to their reliance for ectothermy (external source of body heat), strong seasonal patterns, and preference for more open, relatively 'non-flammable' habitats, and use of burrows for shelter. In the short-term, reptiles are influenced by predation rates, and also by fire intensity – which directly influences patchiness, scorch heights, and food/shelter loss. Fire season may also have an effect, with spring fires likely to be more destructive (in the breeding season), though not much data exists to confirm this.

Seral Responses: reptilian species abundance seems to be linked to the change (following fire) in the structure, composition and density of the vegetation and leaf litter. In the studies mentioned as examples, the evidence shows that, in general, species abundant before the fire and immediately post-fire decline markedly over the next few years and remained 'patchily common' in longer unburnt areas. It would seem apparent, that only a small amount of reptile communities show distinct post-fire seral responses.

Shelter, Diet, and Foraging Patterns: This correlates highly with the life-history requirements of each individual species, even though it is stated, that "no consistent patterns are evident in these species' dietary preferences while all exhibit seasonal reproductive patterns". Early successional species, used burrows, foraged in open areas, though some species were quite abundant in recently burnt areas. Typically, mid-successional stage species require low shrubs and hummock-type grasses for shelter, and open clearing between these clumps for grazing. Burrows also remain a main source of shelter. However, it was discussed that diet, activity and reproductive patterns showed no consistent relationships with the successional stage. Finally, later successional staged species identified showed (similar to the mammals) more specific habitat requirements (substantial leaf-litter, typified by long unburnt areas), and dietary requirements.

AMPHIBIANS

Note: It was commented that not many studies have been conducted on amphibians. Impacts on frog populations were unlikely, unless in indirect ways, since they exhibit strong seasonal activity and breeding patterns, highlighting their dependence on high moisture levels, utilisation of deep burrows and 'non-flammable' habitats (creeks, pools etc.) for shelter/breeding sites.

Acute Impacts: Similar to reptiles, intensity and season of burn are important factors. Some frogs have been shown to bury themselves to escape the fire (some utilise termite mounds as protection and food source), but those species without burying capabilities may suffer direct impact.

Seral Responses: With limited data, it would seem that there is no relationship between the number of frog species recorded and time since fire.

Shelter, Diet, and Foraging Patterns: Burrowing skills of many species allow short– and long-term resilience to fire. For most species, activity and breeding occur in the cooler and wetter months, and their distribution is more related to moisture rather than vegetation composition and/or the pyric succession of their habitat. Furthermore, fire is unlikely to have an impact on the diet, as most species have a wide range of invertebrate prey. Though, there were two examples of frog species, which may be affected (*Myobatrachus gouldii*, and *Litoria* spp.). These species had a specialised diet (termites) and specialised habitat requirement (occupying 'flammable' material for shelter) respectively (Bamford 1992). With the lack of information, means that we can only draw on a few conclusions.

(Information reviewed and modified from Friend 1993)

As the information shows, Friend (1993) has completed an extensive amount of work in the recovery of certain *groups* of organisms post-fire. This is the next step from the disadvantageous use of singular species in the assessment of fire regimes. Once some life-history characteristics of species is known and quite a good amount information does exist in southeast Queensland, especially for herpetofauna, more complex examinations of the habitats, shelter requirements etc. in relationship to the imposed fire regime for each animal group can be performed. What is important here is that there is a regular and measurable pattern of re-establishment following fire for small mammals and amphibians. This can be examined in more detail for regions in southeast Queensland where this type of information is lacking.

Lunney (1987) investigated the effects of various disturbances (fire, logging etc.) on possums and gliders in a forest on the south coast of New South Wales. Even though the work was primarily directed at the effects of logging, it was shown that these animals would preferentially inhabit the unburnt areas, such as those located in gullies, which provided them with refuge. The dominant vegetation (rainforest) suggested an area devoid of fire for some considerable amount of time.

Hannah *et al*. (1997) investigated the effects of prescribed burning (in dry sclerophyll forests) on reptiles and amphibians. This is one of the few research papers located within the southeast Queensland area. The area of study is located in the only continuing long-term monitoring plots in the region (House and Gourley, *pers. comm*.), at Baupal. "There has been some concern that frequent burning is leading to reduced diversity of vertebrates within these forests … and in southeast Queensland, results from other studies on the effects of fire on reptiles and amphibians in lowland coastal forest has suggested this trend exists" (Hannah *et al*. 1997). For their study, reptile abundance, species richness and diversity (using the Shannon–Weaver Index) were all highest in the unburnt (control) plots. Amphibian abundance was highest in the periodically burnt area (every 2–5 years), species richness highest in the unburnt plots, and species diversity (Shannon–Weaver Index) highest in the annually burnt treatments (Hannah *et al.* 1997). The authors did mention that this result might be related to rainfall and not the fire regime. These results correlate with previous studies such as Boorsboom (1983) who stated that the unburnt areas were much higher in relative densities of species than the annually burnt treatments. This may correlate with the late successional stages mentioned previously by Friend (1993). The results for reptiles seems to suggests that periodic fire regimes of 2–5 years in this dry sclerophyll forest may maintain the 'composition measures' representative of unburnt areas. But, this is not a recommendation for ecologically appropriate fire regime in dry sclerophyll forests. It was also stated that high frequency prescribed burning is limiting the composition of reptiles in terms of abundance and diversity (compared to unburnt) but inconclusive for amphibians (Hannah *et al*. 1997). In agreement with other studies, no relationship between time since fire and number (and total abundance) of frogs were found, as amphibian composition was unaffected in the study (Hannah *et al*. 1997). As others (Friend 1993 and Bamford 1992) have stated, this may be more related to the climatic and geographic conditions than the fire regime, with frogs preferring 'wetter' habitats, and subsequent heavy falls at the end of the study in Baupal. Therefore, the conclusions of the work showed that: reptilian (abundance and diversity) but not necessarily amphibian diversity will be suppressed with frequent burning regimes. This reduction in amphibian diversity is due in part to the reduction in structural complexity of the habitat, transformation of habitat to more arid-adapted species, and limiting the time for re-colonisation from other areas (Hannah *et al*. 1997). For management implications, it is very impractical (especially for life and property values) to permit areas to be left unburnt for a long time, or (as the study has shown) burnt too frequently, but what is needed is a mosaic. This will provide a mixture of structural diverse habitats, in different stages of post-fire successional stages, providing for both dry and moist taxa (Hannah *et al.* 1997).

More recently, Mathieson *et al*. (1999) conducted a preliminary analysis of the effects of fire and grazing in two local Blackbutt (*Eucalyptus pilularis*) forests (Mt. Mee and Brisbane Forest Park) in southeast Queensland. Even though, it was a pilot study, the preliminary results indicate that the time since fire (approximately 3–6 years) reduced the grass and litter layers sufficiently to negatively impact on the numbers of herpetofauna and specifically the lizard, *Lampropholis delicata* within the two areas. However, the authors point out that differences between the relatively low numbers of species recorded with other studies could relate to the sites chosen as only ridge and midslopes (ie no gullies) were sampled, as this study was also a comparison of grazing pressures (Mathieson *et al.* 1999). The authors readily admit that the results should be treated with caution unless more work is completed to verify or refute the results. It is, however, a beginning to the type of much-needed research required for this area.

EFFECTS ON RODENTS

Sutherland and Dickman (1999) completed a comprehensive and extensive review of the recovery mechanisms of rodents after fire throughout the Australian continent. This is a highly recommended source of information, and it is meaningless to summarise an already comprehensive review. However, even after such an exhaustive critique of fire effects on rodents in Australia, Sutherland and Dickman (1999, page 415) stated that "it is apparent … there much remains to understand regarding the impact of fire on Australian rodents, and the mechanisms that allow survival at individual and population levels. Further, there has been little treatment of the problem at the community level…it is unrealistic to view individual organisms or species as if they are isolated from the environment as a whole." The overriding theme is that, for any valid use in predicting a species responses to fire, consideration of the wider community is an absolute necessity. As Whelan (1995, quoted in Sutherland and Dickman 1999) states, "…this is especially true (with respect to experimental procedures) because of the lack of replication and control in many previous studies." Furthermore, there is a need for experimental manipulation of habitat variables, especially in post-fire conditions. This will distinguish features of the habitat essential for the survival of that species, and once this is quantified, it should be possible to predict species responses (following a known intensity, frequency and season, ie, *fire regime*) and allow effective management (Sutherland and Dickman 1999).

Fire Regimes and Effects on Avifauna

As this report has already discussed, there has been a substantial amount of research on the effects of fire in heathlands where the fauna species most likely to be used an indicator species is the Ground Parrot. Birds also feature prominently on the list of endangered or rare species which may become potential indicator species. McFarland (1998) proposes the notion that birds species could be used as 'umbrella' species to assess the effects of disturbance (including fire) over large areas of forests, regardless of forest type. Birds are easily recognisable and most bird habitats are also easily measurable, therefore the following is a discussion about the fire and its effects on avifauna in various vegetation types.

Chapman and Harrington (1997) assessed the impacts on the local avifauna in northern Queensland from changed fire regimes in the wet sclerophyll/tropical rainforest boundary. Wet Sclerophyll forests have been identified in this study as the area requiring the most attention for research, due to a large amount of endemic and uncommon species of plants and animals dependent upon this 'intermediate' community type. In the absence of fire, rainforest will invade these areas and change the composition of vegetation, and conversely, too-frequent fire will reduce the wet-sclerophyll vegetation to drier open eucalypt vegetation. The question of what regime is required to maintain this specific type of vegetation with eucalypt overstorey and diverse understorey of grass and sclerophyllous species (for north Queensland) and sub-tropical plants (for southeast Queensland) remains largely unknown. As Chapman and Harrington (1997) state, the high shade of rainforest does not permit sclerophyll tree sto regenerate, though rainforest can regenerate under a sclerophyll canopy. This is an almost perfect definition of what constitutes a 'wet-sclerophyll' forest type. To keep the canopy from closing, fire (at some interval) is required, but with fire suppression and other management practices, Chapman and Harrington (1997) state that 48% of wet sclerophyll in far-north Queensland has been transformed into rainforest in the last 50 years. The invaded areas are quickly usurped and the loss of the highly flammable understorey (grasses) means the forest also loses it ability to ignite.

The results of the study were quite astounding. Only one species of bird (Pale Yellow Robin, *Tregallasia capito*) benefited from the expansion of rainforest, whereas the Golden Whistler (*Pachycephala pectoralis*) showed no (statistically) significant response (Chapman and Harrington 1997). The Eastern Yellow Robin (*Eopsaltria australis*), however, was clearly disadvantaged (a highly specialised species requiring wet areas and open ground to forage), and it was predicted that longer term, the White-naped *Melithreptus lunatus* and White-cheeked *Phylidonyris nigra* Honeyeaters would be threatened from the loss of habitat (Chapman and Harrington 1997). Similar to the Eastern Yellow Robin, the two honeyeaters require the wetter areas adjacent to the rainforest for refuge. *Eucalyptus grandis* (Flooded Gum), a favoured food source of the honeyeater, is particularly under threat and it has been stated that there has been 80% loss over the past half-century (Chapman and Harrington 1997). A similar fate belongs also to the local banksia species. It is apparently clear that fire is needed to maintain this unique and declining vegetation type. As Chapman and Harrington (1997) declare, burning would create suitable foraging areas for the species under threat from rainforest expansion, and facilitate regeneration of tall eucalypt and sub-canopy trees…but, the optimal fire regime is *still to be determined*. Despite the overwhelming evidence indicating fire is needed in this vegetation classification, fires are still considered detrimental to overall diversity (Chapman and Harrington 1997). The existence of endemic fauna suggests this habitat has existed and persisted for thousand of years and more widespread in the past (Chapman and Harrington 1997).

More recently, Woinarski (in Gill *et al*. 1999) completed a comprehensive review of the effects of fire regimes on the avifauna on a continent-perspective. Inappropriate fire regimes have been contributed to the extinction of two of the three bird species, and three of the four subspecies that have disappeared since settlement by the Europeans (Woinarski 1999). More disturbingly, 51 nationally recognised threatened bird taxa are now under threat (Woinarski 1999) from ecologically unsound fire practices. Only habitat clearance and fragmentation is more damaging (which threatens 52 taxa). Even though there is widespread recognition of the misuse of fire (too frequent and/or too infrequent) as one of the main threatening processes, only a few studies have researched (in detail) the relationships between birds and fire regimes, and even less commonly have any of these recommendations been used in management strategies (Woinarski 1999). In Queensland, where there are far fewer studies (compared with other states) on fire regimes and its associated effects, there has been some quite detailed work conducted by McFarland (1988) in Cooloola National Park. Despite this, it is unknown whether the recommended fire regime of 8–10 years to sustain the small populations of the Ground Parrot have been included in the management plan for the park (McFarland *pers. comm*.). Woinarski (1999) stated after the review that in all the cases which the impacts of fire regime on birds were sufficiently well documented, almost all species preferred a regime of less frequent burning than is currently prescribed. This is something worth recommending to the local agencies and authorities. This 'impact' is further compounded with habitat loss through clearing and fragmentation (Woinarski 1999). Only the broad vegetation types that occur in southeast Queensland will be included in the summary (below) of the specific recommendations resulting from Woinarski's (1999) review:

Coastal Heaths: heathlands have been burnt more frequently than is desirable, and it is suggested an interval of (minimum) 10 years is needed, and >20 years (preferably) for most threatened species. The Ground Parrot is one species of regional importance.

Temperate Eucalypt Open Forest: pre European, this vegetation type probably had grassy understoreys (from Aboriginal burning practices). However, with the exclusion of Aboriginal burning, the understorey became shrubbier, and even though many bird species benefited from the increase in understorey complexity – there was a regional homogenisation of the landscape, reducing the regional bird species diversity. The first obvious disadvantage was for birds dependent upon grassy understoreys. Species under threat in this habitat include, the Powerful Owl (*Ninox strenua*). The proposed fire regime is "fire management for bird species in eucalypt open forests should involve the flexible use of a broad range of fire regimes, with specific attention to threatened species. Until more is known (of long-term impacts) … this is the most prudent strategy." Woinarksi (1999).

Temperate Woodlands: Aboriginal land management in this forest type maintained a floristically diverse understorey dominated by *Themeda*. Many woodland birds are declining in southern Australia, through the loss of hollows (in– or directly resulting from fire regimes), changes in understorey vegetation composition, and associated impacts from grazing. The Glossy Black-Cockatoo is one specific species under threat in these woodlands. As is stated, there is an urgent need for data to determine management practices for threatened woodland birds.

Acacia Shrublands and Woodlands: only a relatively small amount of information is available in this vegetation type, and there is not enough available data for implementing appropriate fire regimes. (Refer to Gill *et al*. 1999 for more detailed explanations).

There is no mention of other vegetation types (such as wet-sclerophyll forests) and this can only be attributed to the lack of scientifically adequate information.

Baker and Whelan (1994) examined the population distribution of the Ground Parrot throughout Australia and compared the different recommended fire regimes to conserve this species. It was argued that in the population models designed for the Ground Parrot two outcomes were predicted

(1) the populations will decline to zero some years after the heath has not burnt, and

(2) Ground Parrots remain in long unburnt heath (an extension of the first prediction)

Before any model predicting the recovery of Ground Parrots is recommended, Baker and Whelan (1994) rightly mention that regular censuses (ie validation) of these populations is required before enforcing a management strategy. Reilly (1991) periodically examined six coastal areas in Victoria after the Ash Wednesday fires in 1983. As others have previously stated, the highest danger from fire is to the ground-dwelling species, since they are restricted in habitat and distribution (and moreover, have specific dietary and refuge requirements) (Reilly 1991). Furthermore, it is evident that to maintain a diverse number of species there is a need to provide a mosaic of suitable (ie-different) habitats within a specific area. The persistence of ground-dwelling species, such as the Rufous Bristlebird and the Southern Emu-wren, was attributed to the area of unburnt heath, where these bird species took refuge from the initial fire (Reilly 1991). However, as Reilly (1991) aptly states, response rate following a single fire (albeit, a very highly intense one) means there is insufficient data (especially pre-fire, as population numbers were not recorded) to determine the period necessary for complete re-establishment. What this highlights is that even after much is known about the response characteristics of a species to imposed fire regimes, each site where the species occur may differ slightly but significantly in their responses. This variation in fire regime from site-to-site will require acknowledgment and verification.

The Eastern Bristlebird has been previously mentioned as one bird species under threat from altered fire practices. The distribution of this bird is apparently influenced by the time-since-fire (Pyke *et al*. 1995). Even though some work has been completed on the effects of fire on this species, as Pyke *et al*. (1995) states, there is no information on the density of the Bristlebirds in areas not burnt for ten (or more) years, which prompted their study. Despite the small area of the study, with many adjoining areas receiving different fire regimes, it was shown that the population density of the birds is greatly influenced by the occurrence of mixed-aged (of time-since-fire) stands of heath (Pyke *et al*. 1995). That is, the birds require mature heath (13+ years, preferably longer, without fire) adjoining areas of woodland and heath. Furthermore, it was shown that the bird numbers increase up to nine years since fire and remain constant for an additional four years (Pyke *et al*. 1995).

Eastern Bristlebirds may be a potential indicator species of fire regimes in southeast Queensland. It was discussed that the most suitable habitats for these birds are subjected to periodic burning. The Western Bristlebird is mentioned to occupy heathland some three to ten (3–10) post fire, however, decline in bird populations may occur if areas are left unburnt for more than 30 years. Holmes (1989) also stated that for some Eastern Bristlebirds, recovery was observed only 18 months after a fire, but this was restricted to just one breeding pair, which used standing dead stems of plants as cover. Despite the fact that most of the evidence is circumstantial and speculative, there are some good observations for many areas in southeast Queensland. Nesting birds were located some 3–5 years after fire at Cunningham's Gap. It was also discussed by Holmes (1989) that with frequent fire (2–3 year fire intervals) which resulted in a change of the understorey vegetation would make these sites unsuitable for nesting. Hartley and Kikkawa (1994) reiterated similar comments, stating that fire frequencies greater than 5–10 years will lead to the extinction of the population.

Furthermore, Hartley and Kikkawa (1994) stated in their management plan that burns must be planned and designed to prevent intense wildfires sweeping throughout the area. The establishment of buffer zones was stated as extremely important in preventing the encroachment of wildfire. Furthermore, the use of mosaic fires within suitable refuges is also needed to maximise the protection of the birds (and their habitats) from destructive wildfires (Hartley and Kikkawa 1994). Importantly, the seasonality of the fires was stated as being quite important for the survival of the birds and the regenerative capacity of the habitat. Generally, it was decided that burning in the cooler winter months was preferred due mainly to the fact that the birds breed in the summer months (Hartley and Kikkawa 1994).

As has been previously stated, the Ground Parrot (*Pezoporus wallicus*) has regional significance (classed as 'Vulnerable' in Queensland) and under threat from both habitat fragmentation and improper fire regimes. Meredith *et al*. (1984) studied the Ground Parrot to determine if it is a 'fire-adapted' species and to determine the influences on the diet of this species. It was suggested that the birds do not respond to the increase in foliage cover many years following a fire (minimum of three years) but to seed production of sedges, a major component of their diet (Meredith *et al*. 1984). Finally it was concluded that the Ground Parrot is not a fire-adapted species, but a 'fire-requiring' one (Meredith *et al*. 1984). The distinction between the two descriptions is negligible.

On a more local scale (in Gundiah), Porter and Henderson (1983) surveyed an open forest for birds in response to three different burning regimes (annually, periodically every 2–5 years, and fire exclusion for 29 years, at the time of the study). Intriguingly, the effects of the three very different fire frequencies seemed to have no effect on the species of birds found in each of the treatments, which the authors attribute to the recording of many species on few occasions during the study, which would have biased the results (Porter and Henderson 1983). Despite this, it was stated that the abundance of birds in open forest (in Gundiah) is influenced by structural changes in the understorey as a result of the different fire regimes (Porter and Henderson 1983). This conclusion is another example of the detrimental effects of choosing just one species as an indicator organism of disturbance.

The examination of the effects of fire on birds is well documented and known for most areas in Australia, including southeast Queensland. There remains a good prospect of using birds, in addition to other species, such as ants as viable indicator species of fire regimes. This would, of course, require an extensive amount of study to evaluate each species as indicator organisms.

Assessing Fire Regimes

The ultimate test of an ecologically appropriate fire regime is the maintenance and conservation of the existing biodiversity in the region. The previous sections have outlined some biological indicators to evaluate fire regimes. Below is a section which discusses the types of tests that can be performed to evaluate fire regimes. The advantages of some of these tests are that information about the biology of a specific plant or animal is not necessarily needed and accurate assessments are possible using other methodologies. Loyn (1999) states that time scales of hundreds or (even) thousands of years are clearly needed to describe fire regimes, as opposed to the effects of single fires. Obviously this is not amenable to scientists, land managers and other disciplines. These groups, along with palaeontology and archaeology, sedimentology and even social science needs to be involved in study (Loyn 1999). Therefore, according to Loyn (1999) there are four essential approaches available to supply this type of information:

1. ***Before/after studies*:**

   This approach is used (often) for single planned fires (with replicates) and controls. Major drawback, is that long time periods are needed to determine longer-term effects. Wildfires are problematical, as it is 'impossible' to plan before/after studies in advance. However, sometimes it may be deemed useful just to do the post-fire assessment, when there may be sites already established, which are more fortuitous than planned. Multiple before/after studies fires are rare.

2. ***Designed experiments*:**

   A rigorous approach, possible for planned fires, but not wildfires. Used mainly for deliberate single fires rather than fire regimes, with one notable exception, Tolhurst *et al*. (1992) study in mixed *Eucalyptus* forest in central Victoria. The main limitation of this method is that it is only concerned with one forest type, and due to the rigorous nature of the work, only a few questions can be answered with this approach. The ongoing experiments conducted by Alan House also fit well in here.

3. ***Retrospective studies*:**

   A powerful approach which compares groups of sites with one fire history, with comparable groups of sites with another fire history at one or more points in time. Assumption of comparability is hard to test, and minimum requirements are similar ranges of altitude, slope, aspect, geology and vegetation type. Allows good insight into medium to long-term effects in addition to short-term effects. Two disadvantages exist: it is difficult to match pairs of sites with precision, so for statistical validity a large number of sites are needed, and it is difficult to eliminate bias between groups of sites. That is, gullies are less likely to be chosen as they are less likely to burn, whereas ridge-tops are not.

4. ***Inferential studies*:**

   Valuable inferences can be made from knowledge and distribution and abundance a species and the habitats they use. For example, if the life history characteristics of a rare plant can be described in detail, it may be possible to predict short-term effects and possible to predict fire regime effects in the longer term. Birds in this case are especially valuable, as their habitats are easy to observe in the field. If microhabitats are used for feeding or as a refuge, it may also be possible to estimate how these are affected over time with a particular fire regime. If sedentary species are restricted to narrow ranges, past fire regimes at these sites may show what fire regime is tolerable. Of course, the danger with this method is if the primary inference is wrong, which consequently means that the method requires verification with empirical data.

As Loyn (1999) states the limitation of all these studies is that they all apply to subsets of forest environments, when the ultimate goal is to manage fire in *all* environments. For this goal to be met there is a need for insight about effects of fire protection (such as fuel reduction burning) on ecosystems, the effectiveness of fire protection in regulating frequency or effects of wildfire, and the effects of wildfire (Loyn 1999). No one methodology will answer this 'problem', but it would seem that a sensible combination of the four above mentioned empirical and inferential studies, from a wide range of environments is needed to provide these insights (Loyn 1999).

Monitoring is also seen as a possible mechanism to evaluate the effectiveness of fire regimes. Most studies on fire ecology involve some aspect of monitoring within the study. It has many advantages and can assist in answering many questions, but should also be treated with caution. In recent discussions with fire researchers, there was the notion that too much emphasis is placed on monitoring and that its use will solve all the mysteries behind fire and biodiversity. It is definitely a very useful tool for researchers, but much like indicator species, should be used in conjunction with other techniques to provide a clearer idea of the relationships that are occurring. Below is a section outlining some aspects of monitoring specifically designed to assess fire regimes, and is worth considering for southeast Queensland.

FIRE REGIMES AND MONITORING

To answer some of the many questions relating to fire and biodiversity, Gill (1998) stated that monitoring must form an integral part of any plan to assess the regional effects of fire on biodiversity. According to Gill (1998) two forms of monitoring are possible:

1. *Scientific* – where a hypothesis is formed; there is a defined and replicable sampling strategy; with statistical analysis of data, and
2. *Management* – which can be seen as:
   - *State-of-the-Environment* monitoring: which involves measuring climatic factors, fire occurrences and visitor numbers; is rarely replicable (unless photo-point monitoring is used); involves no experimentation; and may involve aerial photography and remote sensing.
   - *Assisting Decision-Making*: as is suggested, this leads to an eventual decision but needs information regarding timing. "When to prescribe burn?" is a classic example of this monitoring form. May involve knowledge of fuel conditions, proximity to life and property assets and status of the biota, where formal fire-threat analysis is possible, and
   - *To Gain Information*: this is possible in *lieu* of scientific monitoring, where the resources may be lacking to permit exhaustive analysis (eg site replication). There are many thousand species of plants, animals, invertebrates, fungi etc. of which nothing is known about their response to fire. Only simple experimental designs are required and may lead to more formal experimental procedures.

In relation to managing fire to avoid the local extinction of a species, Gill provided in the following figure (5):

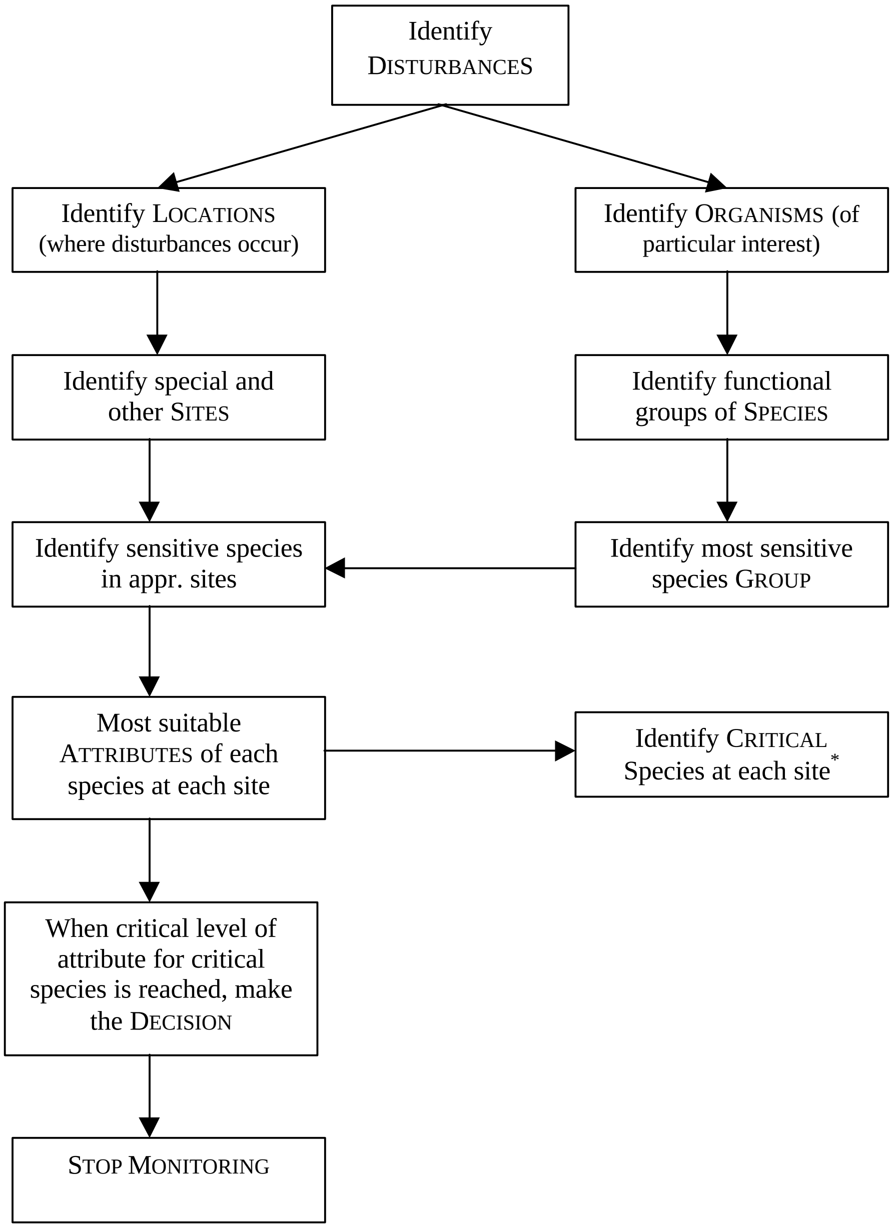

Figure 5: Outline of monitoring system for plant species preservation given an impending fire event or other acute disturbance (Gill 1998).
***In next round of monitoring, only require monitoring of attributes of the critical species**

On the basis of Figure 3, this would involve an extensive amount of work and resources to initiate, but if one area contained a number of 'critical species' then it may be justifiable. Choosing a minimum set of species and sites may also be feasible. It is interesting to note that in recent discussions with the Department of Natural Resources (who are currently working on the Queensland Strategic Fire Management Plan) there is a new proposal for the zoning of areas with respect to life and property assets (Baker *pers. comm*.). Furthermore, in this new classification system fire effects would be concentrated at the community level, rather than at each individual species level. This has particular reference to the Species Management Profiles, where individual species fire regimes are included (though with some conjecture). Therefore it would be apparent that consideration and selection of groups of organisms is a more (cost–) effective solution than choosing individual species. With that in mind, we certainly endorse the examination of Community-level responses (and evaluations thereof).

Sieler and Read (1999) presented a departmental (Victoria Department Natural Resources and Environment, DNRE) perspective on the role of agencies in charge of conservation and the novel methods used in their Fire Protection Plans to maximise conservation values. As opposed to Parks Victoria, the DNRE, has responsibility for conservation across all land tenures, and remains the lead agency for planning, policy development and state-wide coordination (Sieler and Read 1999). Importantly for Queensland, as Sieler and Read (1999) state, the development of partnerships and relationships with DNRE, land managers and the community is central to Parks Victoria's approach to environmental management. This coordinated approach is critical for similar agencies in southeast Queensland to find answers to the numerous questions about fire ecology and environmentally sensitive fire regimes. In Victoria's case, the use of Fire Protection Plans as the name suggests, involves all facets of the planning regime that will include the fire protection *priorities* for the next five year period (Sieler and Read 1999). Similar to Rose *et al*. (1999) use of zoning in NSW, Victoria has a system of management zones for biodiversity, heritage and Indigenous cultural conservation through four stages:

1. Development of clear process and understanding of environmental, heritage and cultural values and importance is established;

2. Identification and analysis of the risks or threats is quantified in terms of potential loss of environmental value (such as fire frequency, fire protection or inappropriate zoning);

3. Management strategies are identified to reduce risks to such values, including examination of the proposed fire protection zones and comparison of these to known minimum and maximum inter-fire periods and scientific literature; and

4. Communication of proposed zones to DNRE for appropriate zoning of these values in the protection plans or at the very least, their identification as spot values requiring protective works during fuel reduction programs or wildfire suppression.

(Adapted from Sieler and Read 1999)

As part of the process there is an initial step to an integrated approach, it considers the delineation of zones with respect to biodiversity, heritage and indigenous values in a systematic manner using state-wide data (Sieler and Read 1999). This type of approach is needed for our state, but will require coordination and commitment from all agencies, which would mean re-allocation of resources, something that seems to be lacking at this point in time.

Recommendations

This report has presented a lot of information on general, and in most cases, specific effects of fire on the landscape. In all cases, the relevance of each research study under review to southeast Queensland has been of utmost importance. With this in mind, there are some specific recommendations for the southeast Queensland region. These are outlined below:

1. Use fire regimes outlined in Table 1 only as a guide. These are very general and as discussed, each site will be different.

2. Maintain and maximise variability in fire regimes. This seems to be crucial for biodiversity. Creating mosaics in any forested area, irrespective of vegetation type is also very important.

3. Adapt and initiate research into the prominent indicators (summarised below) as outlined by Burrows *et al*. (1999) for southeast Queensland:

    - *Minimum intervals between fire to sustain biodiversity,*
    - *Minimum interval between fires for wildfire control,*
    - *Maximum interval between fires to sustain biodiversity,*
    - *Season of fire,*
    - *Fire intensity, and*
    - *Scale and patchiness of fire*

4. Expand on the work completed at the Beerwah, Baupal and Peachester long-term experimental sites. This will be particularly useful in assessing the long-term effects of prescribed burning. Measure more variables and if permitted, establish new sites and examine the effect of changing season of burn on biodiversity.

5. There is an urgent need to initiate research on the wet-sclerophyll–rainforest boundaries within southeast Queensland. Particularly important is the evaluation of ‘follow-up’ burning soon after a prescribed fire. Intensity of prescribed burning is also quite important to study. Very little is known about the complex relationships of the biological species in this habitat. While the presence of wet-sclerophyll forests is limited to only a few regions in the FABC, it holds enough biological value for the whole of southeast Queensland to warrant more work.

6. The response of Eucalyptus open forest is relatively well understood. However, even though it seems that 7–8 years is the minimum inter-fire interval required for shrubby understoreys and 4–5 years is an appropriate minimum for grassy understoreys, there is little idea on the maximum period. This will require some intensive work to determine. Fortunately, the long-term sites mentioned in point-4 provide us with an opportunity to do this. This should be encouraged and instigated.

7. Melaleuca woodlands and coastal (Casuarina) woodlands remain poorly studied ecosystems. Much is unknown about species dependent on these types of forests and the lack of available information is unlikely to be alleviated in the near future as little to no work has been identified to be currently performed in these regions. It is a matter of urgency that research into the aspects of fire regimes (point-3) is commenced immediately.

8. Fire in heathlands, especially in southeast Queensland is quite well understood. Recent discussions have discovered that there is actual implementation of previous work (McFarland 1990 and Sandercoe 1989) relating to the fire ecology of the Ground Parrot found in Cooloola National Park. However, similar to Eucalyptus open-forest, the maximum fire frequency requires validation-using manipulated experiments.

9. Indicator species have many advantages and this report covers all aspects of using organisms as viable measures of fire regimes. For many species, there is a well researched and documented

pattern of recovery post-fire and for groups such as ants, a pattern exists that seems to be match regardless of location. The authors are, however, wary of devoting complete attention of using just one species (or one group) as the indicator species. More work is definitely required to validate these responses post-disturbance and there is also the requirement for some longer term studies to examine any changes in species structure in the absence of disturbance.

Conclusions

It is clear that fire is a significant feature of the Australian landscape and many characteristics of plants have evolved to adapt to fire (Whelan and Brown 1998). It has been demonstrated that repeated high frequency fire intervals have an overall detrimental impact on biodiversity. Furthermore, it has been shown that despite the relatively similar community structure of vegetation throughout Australia, each site has intrinsic differences which makes predicting generalised fire regimes (especially to maintain biodiversity) particularly difficult and fraught with potential threats to maintaining this diversity.

What is clear is that in recent years there has been recognition that variability and mosaics is *critical* for effective fire management. Further, the use of flexible management strategies, zoning of regions with respect to (human) asset hazard and an increase in long-term monitoring of the impacts of repeated fire frequencies is required for a clearer picture of the ecology of fire. Patterns of mosaic burning are recognised as the fire patterns, which naturally occurred in Australia, preceding human settlement. The arrival of the Australian Aborigines extended this burning regime. It is not viable to study the response of just one organism to fire – community level responses is clearly the path that any fire research program must develop. The lack of rigorous, statistically robust studies in the local southeast Queensland region does not permit the confident recommendation of suitable fire regimes for any community type. What is needed is an initiation of long-term monitoring and other research projects. Despite the many decades of studies in other parts of the continent, much remains unknown about the responses to fire.

The recommended fire regimes for different vegetation types should only be used as a guide. It should be noted that these figures were drawn from a variety (of quality) of literature, and should not be treated as ‘hard-and-fast’ regimes. Each region within southeast Queensland will differ in its response (sometimes markedly) to an imposed fire regime, which only shows the need to initiate research programs to thoroughly investigate such impacts.

Possible indicator species are extremely difficult to pinpoint. There is a strong tendency to focus on floral and fauna species under threat as a direct impact of imposed fire frequencies (The Glossy Black Cockatoo *Calyptorhynchus lathami* is an excellent example). Viable population assessment of many species is confounded by the very notion that they are already quite problematical to locate, define and study. While the proposal of using these species is a sound one, the option of using other, more easily determinable populations, which have been shown to provide an excellent indicator of ecosystem health must also be considered. Invertebrates are an option not considered previously in many studies. Many previous studies have shown that ants exhibit a regular pattern of recovery from disturbance. This recovery process seems to be quite uniform throughout Australia and some work completed in southeast Queensland already indicates this.

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Appendix 1: Distribution of Key Endemic Centres in southeast Queensland (QDEH 1998)

KEY NP/ SF LOCALITIES AND ECOSYSTEM TYPES	INDICATIVE SPECIES (10 KM GRID)
Great Sandy NP(south) • wet sclerophyll forest • shrublands • rainforest • tall open forest	*Acacia attenuata, Acacia hubbardiana, Archidendron lovelliae, Argyrodendron* sp., *Astrotricha glabra, Boronia keysii, Boronia rivularis, Callistemon pachyphyllus* var. *viridis, Eucalyptus conglomerata, Grevillea leiophylla, Macadamia ternifolia, Macarthuria complanata, Petrophile shirleyae, Pultenaea paleacea* var. *pauciflora, Rhodamnia acuminata, Schoenus ornithopodioides, Symplocos harroldii, Tecomanthe hillii, Westringia tenuicaulis, Xanthorrhoea fulva, Xanthostemon oppositifolius, Xylomelum salicinum*
Mt Barney NP	
+ SF 745 FTY1263 (Palen) • shrubland • shrubby woodland • tall open forest • heath	*Acacia acrionastes, Acacia bruniodes* ssp. *brunioides, Acacia saxicola. Arundinella grevillensis, Arundinella montana, Astrotriche pauciflora, Banksia conferta* ssp. *conferta, Comesperma breviflorum, Coopernookia scabridiuscula, Eriostemon* sp., *Eucalyptus dura, Helichrysum lindsayanum, Keraudrenia hillii* var. *velutina, Pandorea* sp., *Plectranthus alloplectus, Pultenaea whiteana, Rapanea* sp., *Rulingia salvifolia, Scaevola* sp., *Syncarpia verecunda, Tetramolopium vagans, Westringia blakeana*
Triunia NP	
+ SF368 FTY1612 (Maroochy) **+ SF249 FTY1673 (Maroochy)** **+ Ferntree NP** • rainforest	*Argyrodendron* sp., *Austromyrtus inophloia, Austromyrtus* sp., *Austormyrtus* sp., *Caesalpinia subtropica, Corynocarpus rupestris* ssp. *arborescens, Cupaniopsis serrata, Graptophyllum reticulatum, Jasminum* sp., *Macadamia ternifolia, Mallotus* sp., *Nothoalsomitra suberosa, Pouteria eerwah, Romnalda strobilacea, Tapeinosperma* sp., *Triunia robusta, Zieria* sp.
SF 283 FTY1067 (Colinton) **+ Main Range NP** • tall open forest • heath • shrubby woodland	*Acacia bruniodes ssp. brunioides, Aneilema biflorum, Arundinella grevillensis, Arundinella montana, Austromyrtus* sp., *Bertya pinifolia, Bossiaea prostrata* var., *Brachyscome ascendens, Erythrina* sp., *Eucalyptus dura, Helichrysum lindsayanum, Hovea* sp., *Marsdenia longiloba, Pandorea* sp., *Plectranthus alloplectus, Tetramolopium vagans, Wahlenbergia glabra, Xanthorrhoea* sp.
Burrum Coast NP • closed sedgeland • tall woodland • grassy/ shrubby woodland - OF	*Acacia hubbardiana, Alyxia sharpei, Callistemon pachyphyllus* var. *viridis, Eucalyptus hallii, Leucopogon sp., Melaleuca cheelii, Micromyrtus littoralis, Petrophile shirleyae, Pultenaea paleacea var. pauciflora, Rapanea* sp., *Schoenus ornithopodioides, Westringia tenuicaulis, Xylomelum salicinum*

KEY NP/ SF LOCALITIES AND ECOSYSTEM TYPES CONTAINED IN GRID CELL(S)	INDICATIVE SPECIES (10 KM GRID)
SF 997 FTY 1671 **SF 959 FTY 1295**	*Acacia attenuata, Acacia hubbardiana, Callistemon pachyphyllus* var. *viridis, Cupaniopsis serrata, Eriostemon myoporoides* ssp. *queenslandicus, Eucalyptus conglomerata, Haemodorum tenuifolium, Petrophile shirleyae, Prostanthera* sp., *Rhodamnia acuminata, Strangea linearis, Symplocos harroldii, Xanthostemon oppositifolius*
Mt Walsh NP • tall open forest • montane heath	*Arytera foveolata, Arytera microphylla, Backhousia* sp., *Callistemon* sp., *Coopernookia scabridiuscula, Corynocarpus rupestris ssp. arborescens, Eucalyptus decolor, Eucalyptus dura, Eucalyptus montivaga, Leucopogon rupicola, Leucopogon* sp., *Micromyrtus vernicosa, Triplarina volcanica* ssp. *borealis, Parsonsia leichhardtii, Cassinia collina*
SF840 FTY1633 7 (Bingera) • open forest • grassy/shrubby woodland	*Acacia hubbardiana, Alxyia sharpei, Callistemon pachyphyllus* var. *viridus, Eucalyptus hallii, Glochidion ferdinandi* var. *pubens, Grevillea leiophylla, Lepyrodia* sp., *Leucopogon sp.* , *Melaleuca cheelii, Micromrytus littoralis, Petrophile shirleyae, Pultenaea paleacea* var. *pauciflora, Stackhousia nuda, Westringia tenuicaulis, Xanthorrhoea fulva, Xylomelum salicinum*
Mt Coolum NP	
+ Noosa NP • woodland • rainforest • open forest • heath	*Allocasuarina emuina, Allocasuarina thalassoscopica, Argyrodendron* sp. *Austromyrtus inophloia, Austromyrtus* sp., *Callistemon pachyphyllus* var. *rubrolilacinus, Chamaecrista* sp., *Cupaniopsis serrata, Eucalyptus conglomerata, Gompholobium virgatum* var. *emarginatum, Goodenia* sp., *Leptospermum oreophilum, Marsdenia coronata, Petrophile shirleyae, Rhodamnia acuminata, Schoenus ornithopodioides, Triunia robusta, Westringia tenuicaulis*
SF589 FTY1657 (Beerwah) **+ Glasshouse Mts NP**	*Allocasuarina filidens, Arundinella montana, Banksia conferta* ssp. *conferta, Dodenaea rupicola, Leptospermum luehmannii, Leptospermum oreophyllum, Westringia grandifolia,*
+ SF611 FTY1687 (Beerwah) **+ SF561 FTY1655 (Beerwah)** • open forest • rainforest • shrubland	*Acacia attenuata, Acacia hubbardiana, Eucalyptus conglomerata, Eucalyptus kabiana, Gonocarpus effusus, Grevillea hodgei, Grevillea leiophylla, Lepyrodia* sp., *Petrophile shirleyae, Poranthera* sp., *Triplarina volcanica* ssp. *volcanica, Xylomelum salicinum*

KEY NP/ SF LOCALITIES AND ECOSYSTEM TYPES CONTAINED IN GRID CELL(S)	INDICATIVE SPECIES (10 KM GRID)
Moogerah Peaks NP – Mt Moon	
+Moogerah Peaks NP - Mt Greville • rainforest • open forest • shrubland	*Acacia acrionastes, Acacia brunioides* ssp. *brunioides, Arundinella grevillensis, Arundinella montana, Comesperma breviflorum, Cupaniopsis tomentella, Eucalyptus dura, Helichrysum lindsayanum, Hovea* sp.*, Marsdenia coronata, Notalaea lloydii, Phebalium gracile, Plectranthus alloplectus, Syncarpia vercunda, Wahlenbergia glabra, Westringia sericea*
Lamington NP+	*Austromyrtus* sp.*, Banksia conferta* sp. *conferta, Brachyscome ascendens, Caesalpinia subtropica, Cassia marksiana, Corynocarpus rupestris subsp. arborescens, Cupaniopsis serrata, Doryanthes palmeri, Eucryphia jinksii, Jasminum* sp.*, Genoplesium sigmoideum, Helmholtzia glaberimma, Hovea* sp.*, Macadamia integrifolia*
SF 316 FTY1328 (Kroombit Tops)	*Eucalyptus montivaga, Parsonsia kroombitensis, Asperula conferta* var. *scoparioides, Senecio* sp.*, Persoonia volcanica, Parsonsia lilacina, Muellerina myrtifolia, Eriostemon myoporoides* ssp. *leichhardtii, Bouchardatia neurococca*
Plunkett CP – TR 766 FTY1420	*Macadamia integrifolia, Microcitrus australis, Eucalyptus dura, Fontainea venosa, Symplocos harroldii, Cassytha muelleri, Persoonia* sp.*, Fasciculochloa sparshottiorum, Pouteria eerwah, Dissilliaria baloghioides*
Blue Lake NP	*Syncarpia hillii, Marsdenia fraseri, Petrophile shirleyae, Xylomelum salicinum, Strangea linearis, Genoplesium psammophilum, Haemodorum tenuifolium, Oleria hygrophilum, Schoenus ornithopodioides*
SF 471 FTY 863	*Alyxia sharpei, Argyrodendron* sp.*, Arytera dictyoneura, Bouchardatia neurococca, Caesalpinia subtropica, Capparis* sp.*, Choricarpia subargentea, Goodenia* sp.*, Hernandia bivalvis, Parsonsia leichhardtii, Xylosma terrae-reginae*
SF 391 FTY 1007	*Acacia bakeri, Argyrodendron* sp.*, Arytera dictyoneura, Bosistoa transversa, Bouchardatia neurococca, Capparis* sp.*, Cupaniopsis serrata, Cupaniopsis shirleyana, Dissiliaria baloghioides, Medicosma cunninghamii, Medicosma elliptica, Parsonsia leichhardtii, Parsonsia lilacina, Persoonia volcanica, Phyllanthus sauropodoides, Phyllanthus* sp.
Mt Bauple NP	*Argyrodendron* sp.*, Arytera foveolata, Arytera microphylla, Bouchardatia neurococca, Caesalpinia subtropica, Choricarpia subargentea, Cossinia australiana, Cupaniopsis serrata, Dissiliaria muelleri, Macadamia integrifolia, Medicosma cunninghamii, Parsonsia leichhardtii, Xylosma terrae-reginae*

KEY NP/ SF LOCALITIES AND ECOSYSTEM TYPES CONTAINED IN GRID CELL(S)	INDICATIVE SPECIES (10 KM GRID)
SF 309 FTY 1307	*Arytera foveolata, Bouchardatia neurococca, Capparis sp., Choricarpia subargentea, Dissiliaria baloghioides, Hypoestes floribunda* var. *pubescens, Medicosma cunninghamii, Microcitrus australis, Parsonsia lilacina, Rapanea* sp., *Swainsona fraseri, Symplocos harroldii*
SF 135 FTY1638	*Bosistoa transversa, Bouchardatia neurococca, Capparis* sp., *Cupaniopsis serrata, Erythrina* sp., *Macadamia ternifolia, Medicosma cunninghamii, Microcitrus australis, Parsonsia lilacina, Sarcochilus dilatatus*

Appendix 2: Further Rare and Threatened Species for consideration (adapted and modified from Novello and Klohs 1998)

Distribution:

N – northern limit of biogeographical range
En – endemic to south-east Queensland & north-east New South Wales
D – disjunct distribution
U – uncommon species

Status:

E – endangered species in serious risk of disappearing
V – vulnerable species not presently endangered but at risk
R – rare species but not currently considered vulnerable or endangered
K – poorly known species which probably fits one of the above categories

Structure	Community Type	Noteworthy Species	Distri-bution	Status	Location	Life Form	Flowering Time	Fruiting Time	Regenerating Method & Fire Regimes
Closed Forest	1A Cool Subtropical Rainforest	*Acradenia euodiiformis*	N		Lamington NP, Springbrook NP	Shrub	October–December	Fruit Ripe – January	Poor seed germination, known to sucker
		Acronychia baeuerlenii	En	R	Lamington NP, Springbrook NP	Tree	October–February	March–April	Poor seed germination, possibly suckers
		Akania bidwillii (syn. *Akania lucens)*	D		Chinghee, Mt Barney Complex, Lamington NP, Springbrook NP	Tree	Spring	February–May	Very slow growing. Possibly very intolerant to fire
		Anopterus macleayanus	N		Mt Barney Complex, Lamington NP, Springbrook NP	Shrub	October–December		Seed viability is unknown but appears to be for a short time. Intolerant to fire.
		Choricarpia subargentea		R	Moogerah Peaks	Tree	Spring–Summer	April–June	Possibly intolerant to low intensity fires
		Cinnamomum virens	N		Main Range, Lamington, Mt Barney Complex, Springbrook	Tree	Autumn	December–January	Relatively slow growing
		Clematis fawcettii		V	Main Range, Lamington NP, Mt Lindesay (along Rf edges)	Vine	Spring	January–December	Rainforest edge species. Obligate seeder. Seed viability unknown. Seeds: wind dispersed. Juvenile period: 2 years. Killed by fire.
		Cryptocarya foveolata	N		Main Range, Mt Barney Complex, Lamington, Springbrook (up to 150 m altitude)	Tree	November–December	April–October (fruit ripe)	Seed dispersal probably by birds
		Dendrobium schneiderae		R	Main Range NP, Lamington NP	Epiphyte Orchid	Summer–Autumn		
		Dicksonia youngiae	D		Lamington NP	Tree Fern			
		Eucryphia jinksii	En.	Pending E.	Springbrook NP (Natural Arch section on hillslope 770–800 m altitude)	Tall Tree			Obligate seeder with long lived seed bank exhausted by disturbance. Seeds viable (1 yr). Juvenile period (8 years)
		Geissois benthamii	N		Main Range, Springbrook, Mt Barney, Lamington NP, Chinghee	Tree	Winter–Spring	May–August	
		Helmholtzia glaberrima	En.	R	Lamington NP, Springbrook NP	Herb	Spring–Autumn		
	1A Cool Subtropical Rainforest	*Lastreopsis silvestris*		R	Main Range NP, Lamington NP, Springbrook NP	Epiphyte Fern	–		

		Myoporum betcheanum	N		Main Range NP	Shrub	Summer–Autumn		
		Orites excelsa	N		Main Range, Springbrook, Mt Barney, Lamington, Chinghee	Tree	Winter–Spring	February–July	
		Ozothamnus vagans (syn. *Helichrysum vagans)*		V	Mt Barney Complex, Lamington	Shrub	Spring–Autumn		Obligate seeder with long-lived seed bank. Seeds: wind dispersed. Juvenile period: 2+ years.
		Pimelia ligustrina	N		Main Range, Lamington, Springbrook	Shrub	Spring–Summer		
		Quintinia sieberi	N		Main Range, Mt Barney Complex, Springbrook	Tree	Spring	December–January	
		Rhodamnia whiteana	N		Main Range, Lamington	Tree	Summer		
		Sarcochilus weinthalii	D	E	Main Range & Lamington NP on upper branches of rainforest trees	Epiphyte Orchid	June–October	Persists 6–8 weeks	Resprouter. Seeds: wind dispersed and viable for 2–6 weeks. Juvenile period: 2 years.
		Solanum callium		R	Mt Barney Complex, Chinghee	Shrub	Summer		
		Sophora fraseri	D	V	Main Range NP (Mt Mistake), Lamington NP	Shrub	April–November	January–November	Resprouter. Long lived seed bank is exhausted by disturbance. Juvenile period greater than 4 years. Responds poorly to frequent fire.
		Triunia youngiana	D		Lamington, Springbrook	Shrub	November–December	March–May	
		Trimenia moorei	N		Mt Barney Complex		Early Summer		
Closed Forest	1AA Warm Subtropical Rainforest	*Acronychia baeuerlenii*	En.	R	Lamington, Springbrook	Tree	October–February	March–April	Poor seed germination, possibly suckers
		Amorphospermum whitei	D	V	Springbrook MP (Mt Cougal), Upper Tallebudgera	Tree	September–October	Ripe in Spring	Obligate seeder. Seed viability (1–3 months). Seeds dispersed by mammals. Juvenile period is greater than 6 years, responds poorly to fire.
		Antrophyum sp. ('Blue Pool')		K	Lamington NP	Epiphyte Orchid			
		Archidendron muellerianum (syn. *Pithecellobium muellerianum)*		R	Springbrook NP (Natural Bridge)	Tree	November–December	March–July	Fire tolerance (unknown) but known to sucker
		Ardisia bakeri	En.	R	Springbrook NP	Shrub, Tree	Spring	March–May	
		Argophyllum nullumense	D	R	Lamington NP (Warrie section), Springbrook NP (Mt Cougal)	Shrub, Tree	October–January	May–September	Probably tolerate low intensity fires. Known to sucker
		Baloghia marmorata	En.	V	Springbrook	Shrub. Tree	July–September	July–August	Obligate seeder. Seed viability (2–3 weeks). Seed dispersal (mechanical, explosive mechanism). Juvenile period: 5 years, tolerates frequent fire.
		Bulbophyllum argyropus		R	Lamington NP	Epiphyte Orchid	Mainly in Autumn		

		Callerya australis (syn. *Milletia australia)*		R	Lamington, Springbrook and Chinghee	Vine	Spring	March–May	Seeds moderately viable. Seeds falls to ground and probably dispersed by mammals
		Cassia marksiana		R	Lamington, Springbrook	Shrub	Early Summer	June–August	
		Citrus australasica (syn. Microcitrus australasica)	U, En.		Mt Barney Complex, Lamington, Springbrook	Shrub, Tree	Autumn	May–September	Relatively slow growing
		Clematis fawcettii		V	Main Range NP, Mt Dumeresq, Mt Barney NP, Lamington NP (along rainforest edges)	Vine	Spring	January–December	Obligate seeder. Seed viability (unknown), wind dispersal. Juvenile period (2 years), killed by fire
		Cordyline congesta		R	Lamington NP	Shrub	Spring–Summer		
		Corynocarpus rupestris	D	R	Moogerah Peaks, Lamington NP, Natural Arch	Tree	October–November		Can resucker from base
		Crytopcarya foetida		V	Lamington, Springbrook	Tree	December–February	June–August (fruit ripen)	Resprouts after mild fire. Repeated fire are a threat. Seed viability (2–3 weeks). Seeds bird dispersed. Juvenile period is greater than 10 years
		Cupaniopsis newmanii		R	Lamington, Springbrook	Tree	August–October	October–November	Slow in seedling stage. Suckers following fire. Seed germinates readily but possibly short viability
		Dendrobium schneiderae		R	Main Range, Lamington	Epiphyte Orchid	Summer–Autumn		
		Endiandra hayesii	En., D	V	Springbrook NP (Warrie section)	Tree	Spring	August–March	Obligate seeder. Seed bank short lived. Birds disperse seeds. Juvenile period is greater than 10 years
		Floydia praealta	D	V	Springbrook NP	Tree	Summer–Autumn	January–April	Seed drops to ground. Can resucker from roots. Juvenile period: 10 years
		*Fontainea australis (*syn. *F. sp.* Tweed Valley*)*	N	V	Springbrook NP (Natural Bridge)	Shrub, Tree	December–January	July	Obligate seeder with short-lived seed bank (seed drops to ground). Juvenile period: 10 years
		Harpullia alata	N		Lamington, Springbrook	Shrub	Autumn	July–September	
		Helicia ferruginea	N		Lamington NP	Tree	Spring–Summer	January–February	Slow growing
		Helmholtzia glaberrima	En.	R	Lamington NP, Springbrook NP	Herb	Spring–Autumn		
		Hicksbeachia pinnatifolia	D	V	Springbrook NP (Mt Cougal section)	Tree	September–October	September–January	Resprouter. Seed viability (1–2 weeks). Possibly bird and flying fox dispersed. Killed by fire
		Lepiderema pulchella	N, En.	R	Lamington, Springbrook	Tree	September–October	December–January (fruit ripens)	Fruit often infertile, containing few seeds. Suckers following low/moderate intensity fires
		Macadmia tetraphylla	D	V	Lamington NP (north end of Lower Bellbird), Springbrook NP (Natural Arch)	Tree	Late Winter–Spring	January–March	Resprouts when damaged. Seed viability 3–6 months and seed bank exhausted by predation. Seeds drop to ground and maybe stream

									dispersed. Juvenile period: greater than 6 years
		Meiogyne stenopetala subsp. *stenopetala*	N		Lamington, Springbrook	Shrub, Tree	February–April	August–October	
		Notelaea johnsonii	U		Lamington NP	Shrub, Tree	Spring–Summer	November–December	Most likely bird dispersed
		Ochrosia moorei	D, En.	E	Springbrook NP (Natural Arch section)	Shrub, Tree	Summer	December–January	Obligate seeder. Seed drops to the ground and viable for 2–3 months, dispersed by birds and ground-dwelling mammals. Juvenile period around 5 years
		*Ozothamnus vagans (*syn. *Helichrysum vagans)*		V	Mt Barney Complex, Lamington	Shrub	Spring–Autumn		Obligate seeder with long-lived seed bank. Seeds: wind dispersed. Juvenile period: 2+ years.
		Papillilabium beckleri		R	Main Range, Lamington (along water-courses)	Epiphyte Orchid	September–October		Protect from fire
		*Pararistolochia laheyana (*syn. *Aristolochia deltantha* var. *laheyana)*	D	R	Lamington, Springbrook	Vine	Mainly Summer		Seed dispersal: ground dwelling birds, eg. Brush-turkey when scratching litter
		Petermannia cirrosa	En.	V	Springbrook NP	Vine	Summer		Possibly tolerant to cool fires as it grows in wet-sclerophyll forest
		Pomaderris notata		R	Lamington NP	Shrub	Spring–Summer		
		Rhodamnia maideniana		R	Springbrook NP–Mt Cougal	Tree			
		Sarcochilus weinthalii	D	E	Lamington	Epiphyte Orchid	June–October	6–8 weeks	Resprouter. Seeds: wind dispersed and viable for 2–6 weeks, with juvenile period of 2 years
		Symplocus baeuerlenii	En.	V	Springbrook NP (Canyon area) and Mt Cougal	Epiphyte Orchid	Winter–Spring	Ripe in February	Seeds: possibly bird dispersed. Could resucker, with juvenile periods extending more than 10 years
		Syzygium hodgkinsoniae	D	V	Springbrook NP (Natural Bridge) and Mt Cougal	Tree	Late Summer–Autumn	Fruit ripe in Spring	Obligate seeder and roots could resprout. Seeds drop to ground and possibly stream dispersed. 3-4 week seed viability with a juvenile period of 10+ years
		Syzygium moorei		V	Springbrook NP (Mt Cougal section)	Tree	November–January	June–August	Seeds drop to ground and maybe stream/bird dispersed. Possible resucker, with juvenile period of 15-20 years
		Zieria collina	En.	V	Mt Tamborine NP, Mt Barney Comple	Shrub	Mainly Spring		Obligate seeder with short-lived seed bank. Seeds drop to ground and possibly wind dispersed. Opportunistic plant requiring disturbance for reproduction

Closed Forest	9D Cool Temperate *Acmena smithii* communities with 10i and 11i assoc. shrublands	*Atalaya multiflora*	U		Moogerah Peaks, Lamington, Chinghee	Tree		Summer	Under threat from *Lantana* invasion
		Bulbophyllum globuliforme	En.	V	Lamington NP (nr Cainbable Falls), on mature hoop pines	Epiphyte Orchid	Autumn		Obligate seeder. Seed viability: 2-6 weeks, with wind dispersal. Juvenile period: 4 years. Habitat: mature hoop pines
		Cassia marksiana		R	Lamington, Springbrook	Tree	September–November	June–August	Usually poor seed viability from grub attack. Suckers from moderately intense fires
		Croton stigmatosus	U		Lamington, Chinghee	Tree	September–December	December–January	
		Citrus australasica (syn. Microcitrus australasica)	U, En.		Mt Barney Complex, Lamington, Springbrook	Shrub, Tree	August–November	May–September	Relatively slow growing
		Cupaniopsis tomentella		V	Moogerah Peaks (Mt French and Mt Edwards)	Tree			Possible resprouter. Seed viability: 1–2 weeks, with 10 year juvenile period
		Dendrobium schneiderae		R	Main Range, Lamington	Epiphyte Orchid	Summer–Autumn		
		Marsdenia longiloba (possible upgrade to Vul.)	D	R	Main Range (Mt Bangalora), Lamington NP (north side of Shipstern Range)	Vine	November–January		Seeds require insect pollination. Fruiting is rare & seasonal. Seed dispersal: wind, with viability up to one year. Resprouts from rootstock, and a 2 year juvenile period
	Parasitic on *Croton*, *Parsonsia* and *Pandorea jasminoides*	*Muellerina myrtifolia*		R	Main Range NP	Parasitic Vine	Mainly Spring–Summer		Seed dispersed by Mistletoe bird (*Diaceum hirundinaceum*) and possibly other fruit-eating birds
		Owenia cepiodora	D	V	Lamington NP	Tree	November	Fruit ripe January–March	Seed bank exhausted by disturbance. Seeds drop to ground with viability for 6 months. Known to resucker. Juvenile period can exceed 100 years and trees live for more than 200 years
		Papillilabium beckleri		R	Main Range, Lamington (along water-courses)	Epiphyte Orchid	September–October		Protect from fire
		Sarcochilus weinthalii	D	E	Lamington	Epiphyte Orchid	June–October	6–8 weeks	Resprouter. Seeds: wind dispersed and viable for 2–6 weeks, with juvenile period of 2 years
		Sophora fraseri	D	V	Main Range NP (Mt Mistake), Lamington NP	Shrub	April–November	January–November	Resprouter. Long-lived seed bank is exhausted by disturbance. Juvenile period greater than 4 years. Responds poorly to frequent fire. Fire intervals of at least 8 years are suggested
		Uromyrtus sp. "McPherson Range'		R	Lamington NP	Shrub Tree	November–December	March–July	

Closed Forest	5H *Nothofagus moorei* Forest with 10i and 11i assoc. shrublands	*Cassinia compacta*	N		Main Range, Mt Barney Complex, Lamington and Springbrook	Shrub	Spring–Summer		Seed: Predominantly wind dispersed
		Cassinia trinervia	N		Mt Barney Complex	Shrub Tree	Summer–Autumn		Seed: Predominantly wind dispersed
		Cryptandra sp. 'Q1'	D		Mt Barney Complex				
		Dendrobium falcorostrum	N		Mt Barney Complex, Springbrook, Lamington	Herb	Late Winter-Spring		
		Dendrobium schneiderae		R	Main Range, Lamington	Epiphyte Orchid	Summer–Autumn		
		Euphrasia bella	En.		Lamington NP	Herb	Spring–Summer		
		Helmholtzia glaberrima	En.	R	Lamington, Springbrook	Herb	Spring–Summer		
		Lastreopsis silvestris		R	Main Range, Lamington, Springbrook	Epiphyte Fern	–		
		Nothofagus moorei	N		Mt Barney Complex, Lamington, Springbrook	Tree	Spring–though not every year	December–February	Seed germination probably triggered by disturbance, eg. storm damage, mild fire or drought to open up canopy. Mass germination occurred following 1983 tornado
		Pararistolochia laheyana (syn. *Aristolochia deltantha* var. *laheyana)*	D	R	Lamington, Springbrook	Vine	Mainly Summer		Seed dispersal: ground dwelling birds, eg. Brush-turkey when scratching litter
		Parsonsia tenulis	En.	R	Lamington, Springbrook	Vine	All year		
		Pittosporum oreillyanum	En.	R	Lamington Np	Shrub	Spring	March–April	
		Pneumatopteris pennigera	N, D	R	Lamington NP	Fern	–		
Closed Forest	51 Warm Temperate Simple–Complex Microphyll Forest	*Alloxylon pinnatum* (syn. *Ocercallis pinnata)*	N	R	Lamington, Springbrook	Tree	Spring–Autumn	February–June	Low seed viability due to borer attack. May require mycorrhizal association to survive to maturity–low seedling survival rate
		Cyathea cunninghamii	N	R	Lamington NP	Tree	–		
		Dicksonia youngiae	D		Lamington NP	Tree	–		
		Ozothamnus vagans (syn. *Helichrysum vagans)*		V	Mt Barney Complex, Lamington	Shrub	Spring–Autumn		Obligate seeder with long-lived seed bank. Seeds: wind dispersed. Juvenile period: 2+ years.
		Pneumatopteris pennigera	N, D	R	Lamington NP	Fern	–		
Closed Forest	5J *Ceratopetalum apetalum* Warm Temperate Rainforest	*Acradenia euodiiformis*	N		Lamington, Springbrook	Tree	October–December	Fruit ripe in January	Poor seed germination, known to sucker
		Alloxylon pinnatum (syn. *Ocercallis pinnata)*	N	R	Lamington, Springbrook	Tree	Spring–Autumn	February–June	Low seed viability due to borer attack. May require mycorrhizal association to survive to maturity–low seedling survival rate
		Anopterus macleayanus	N		Mt Barney Complex, Lamington, Springbrook	Tree	October–December		Seed viability is unknown but appears to be for a short time. Intolerant of fire
		Austrobuxus swainii	En.	R	Springbrook	Tree	April–June	February–	

							March	
		Ceratopetalum apetalum	N		Lamington, Springbrook	Tree	November–December	January–March	
		Dendrobium schneiderae		R	Main Range, Lamington	Epiphyte Orchid	Summer–Autumn		
		Helicia ferruginea	En.	R	Lamington NP	Tree	Spring–Summer	January–February	Slow growing in establishment
		Helmholtzia glaberrima	En.	R	Lamington NP, Springbrook NP	Herb	Spring–Autumn		
		Lastreopsis silvestris		R	Main Range, Lamington, Springbrook	Epiphyte Fern	–		
		Ozothamnus vagans (syn. *Helichrysum vagans)*		V	Mt Barney Complex, Lamington	Shrub	Spring–Autumn		Obligate seeder with long-lived seed bank. Seeds: wind dispersed. Juvenile period: 2+ years.
		Pararistolochia laheyana (syn. *Aristolochia deltantha* var. *laheyana)*	D	R	Lamington, Springbrook	Vine	Mainly Summer		Seed dispersal: ground dwelling birds, eg. Brush-turkey when scratching litter
Open Forest & Woodland	2a1, 2a2, & 4a *Lophostemon confertus* communities	*Alloxylon pinnatum* (syn. *Ocercallis pinnata)*	N	R	Lamington, Springbrook	Tree	Spring–Autumn	February–June	Low seed viability due to borer attack. May require mycorrhizal association to survive to maturity–low seedling survival rate
		Alyxia ilicifolia spp. magnifolia		R	Springbrook				
		Anopterus macleayanus	N		Mt Barney Complex, Lamington, Springbrook	Tree	October–December		Seed viability: unknown but appears to be short-lived. Intolerant to fire
		Argophyllum nullumense	D	R	Lamington NP (Warrie section), Springbrook NP (Mt Cougal)	Shrub Tree	October–January	May–September	Probably tolerates low intensity fires. Suckers.
		Cyperus semifertilis	En., D	V	Springbrook on rainforest edge	Sedge			Perennial, requiring disturbance to regenerate Seeds short lived: dispersed by wind, water and rats. Resprouter
		Helicia ferruginea	En.	R	Lamington NP	Tree	Spring–Summer	January–February	Slow growing in establishment
		Marsdenia longiloba (possible upgrade to Vul.)	D	R	Main Range (Mt Bangalora), Lamington NP (north side of Shipstern Range)	Vine	November–January		Seeds require insect pollination. Fruiting is rare & seasonal. Seed dispersal: wind, with viability up to one year. Resprouts from rootstock, and a 2 year juvenile period
		Sophora fraseri	D	V	Main Range NP (Mt Mistake), Lamington NP	Shrub	April–November	January–November	Resprouter. Long-lived seed bank is exhausted by disturbance. Juvenile period greater than 4 years. Responds poorly to frequent fire. Fire intervals of at least 8 years are suggested
	4b *Eucalyptus eugenoides* Communities	*Eucalyptus quadrangulata*	D, N		Main Range, Lamington	Tree	Summer–Autumn		Usually grows adjacent to rainforest and may therefore be intolerant to high intensity fire conditions

		Santalum obtusifolium	N		Lamington	Shrub	Early Summer		
	2c, 4c1 and 4c2 *Eucalyptus campanulata* Communities	*Acacia orites*	En.	R	Lamington, Springbrook	Tree	August–September	November–January	Fast growing pioneer species which is disturbance opportunistic
		Acacia paradoxa	D, U		Lamington	Shrub	Late Spring	November–February	
		Acianthus amplexicaulis	U	R	Lamington	Herb	Autumn–Winter		
		Eucalyptus banksii	N, D		Lamington NP, Main Range NP	Tree	Summer–Autumn		
		Dodonaea megazyga	U		Lamington, Springbrook	Shrub Tree	Late Spring–Summer	Late Spring–Summer	Probably an obligate seeder. Responds to disturbance. Possibly promoted by moderate fires. Seeds require fir or scarification for germination
		Lecopogon lanceolatus var. *lanceolatus*	D		Lamington	Shrub	Late Winter–Spring		Seeds may be animal dispersed, most drop to ground
		Marsdenia coronata	D	V	Main Range NP (Steamers, east of Emu Vale), Moogerah Peaks (Mt Moon)	Vine	November–March	3–4 months	Obligate seeder. Seed viability is short. Seeds widely dispersed by wind. Juvenile period of one year
		Olearia heterocarpa	En.	R	Lamington, Springbrook	Shrub	Spring–Summer		
		Pandorea baileyana	En.	R	Mt Barney Complex, Lamington, Springbrook	Shrub	Spring		
		*Persoonia volcanica (*syn. *P. attenuata)*	D	R	Mt Barney NP, Lamington, Springbrook NP (Mt Cougal)	Vine	Much of the year		Seed drop to ground and may be animal dispersed
		Poranthera corymbosa	U		Mt Barney Complex	Shrub	Spring–Autumn		
		Rhizanthella slateri		K	Lamington	Herb	Spring		
		Santalum obtusifolium	N		Lamington	Shrub	Early Summer		
		Tristaniopsis collina	N		Mt Barney Complex, Lamington, Springbrook	Tree	Summer		
		Zieria granulata var. *adenodonta*	En., D	R	Lamington	Shrub	Late Winter–Spring		An obligate seeder. Seeds drop to the ground and possibly bird and wind dispersed. Seed viability 2-3 years, with juvenile period up to 6 years
	4d *Eucalyptus viminalis* Communities	*Eucalyptus viminalis*	N		Main Range NP	Tree	Summer–Autumn		Mainly occurring along drainage systems. Regeneration largely threatened by *Lantana camara* & other weed invasions
	2e *Eucalyptus*	*Eucalyptus dunnii*	N, En.	R	Main Range, Lamington	Tree	Autumn		Seeds shed throughout the year,

	dunnii								though viability is low. Trees bear seed at 20 years. Young trees not very fire tolerant. Requires fire intervals greater than 20 years
	4f *Eucalyptus acmenoides* Communities	*Marsdenia coronata*	D	V	Main Range NP (Steamers, east of Emu Vale), Moogerah Peaks (Mt Moon)	Vine	November–March	3–4 months	Obligate seeder. Seed viability is short. Seeds widely dispersed by wind. Juvenile period of one year
		Persoonia volcanica (syn. *P. attenuata)*	D	R	Mt Barney NP, Lamington, Springbrook NP (Mt Cougal)	Vine	Much of the year		Seed drop to ground and may be animal dispersed
	4g *Eucalyptus obliqua* Communities	*Eucalyptus obliqua*	N		Main Range NP	Tree	Summer		
	2h *Eucalyptus laevopinea*	*Eucalyptus laevopinea*	N		Main Range NP	Tree	Late Summer–Autumn		
		Eucalyptus quadrangulata	D, N		Main Range, Lamington	Tree	Summer–Autumn		Usually grows adjacent to rainforest and may therefore be intolerant to high intensity fire conditions
	6l *Eucalyptus tereticornis*	*Podolepis monticola*	En.	R	Lamington	Herb	Spring–Summer		
	7j, 4l & 7l 2h *Eucalyptus crebra & Corymbia citriodora–E. crebra*	*Thesium australe*	D	V	Main Range NP	Herb	Spring–Summer		Resprouter. Annual or biennial. Seed viable for around one year. Require regular burns to maintain open conditions. Summer burns may prevent regeneration
	4k & 6k *Eucalyptus oreades* and 10ii and 11ii assoc. shrublands	*Acacia saxicola*		R	Mt Barney NP, Mt Maroon	Shrub			Seed likely to be long lived
		Allocasuarina rigida	D		Main Range, Mt Barney Complex, Lamington, Springbrook	Tree			Obligate seeder. Dispersal: by wind or drop to ground. Juvenile period of 3–5 years
		Baeckea linifolia			Lamington, Springbrook	Shrub	Summer		Suckers following fire
		Banksia conferta	D	R	Mt Barney Complex, Lamington	Tree	April–July		Seed follicles remain closed until burnt. May be fire tolerant, resprouting from trunk
		Bauera rubioides	N		Moogerah Peaks, Mt Barney	Shrub	Spring–Summer		
		Callistemon comboynensis	En.		Main Range, Moogerah Peaks, Mt Barney Complex, Lamington, Springbrook	Shrub	Summer–Autumn		Suckers
		Callistemon montanus	En.		Mt Barney Complex, Lamington, Springbrook	Shrub	Mainly in Spring		Suckers
		Callistemon pallidus	N		Mt Barney Complex, Lamington, Springbrook	Shrub	Spring–Summer		Suckers
		Callitris monticola		R	Mt Barney Complex, Lamington, Springbrook, Main Range	Shrub			An obligate seeder. Possibly very intolerant of fire. Seed prodcued 7 years after fire at Ships Stern: viability unknown
		Comesperma esulifoilum	En.	R	Mt Barney Complex, Lamington,	Shrub	Spring–early		Possibly obligate seeder

					Springbrook		Summer		
		Coopernookia scabridiuscula	En., D	V	Mt Barney NP, Mt Maroon	Shrub	Late Winter–Spring		Ecotonal between rainforest and heath beside cliff. Obligate seeder. Seed drop to ground and are short-lived. Juvenile period of three years
		Epacris longiflora	N		Mt Barney, Mt Lindesay, Lamington, Springbrook	Shrub	Winter–early Summer		
		Eriostemon myoporoides	N		Mt Barney Complex	Shrub	Spring–Autumn		
		Eucalyptus approximans	D	R	Mt Barney Complex, Lamington, Springbrook	Shrub	Autumn–Winter		Probably quite tolerant of high intensity fire but regular successional fire may reduce seed production dramatically
		Eucalyptus notabilis	D, N		Mt Barney Complex, Lamington	Shrub	Summer		Probably quite tolerant of high intensity fire but regular successional fire may reduce seed production dramatically
		Eucalyptus oreades	D		Mt Barney Complex, Lamington, Springbrook	Tree	Summer		
		Gahnia insignis	En.	R	Mt Barney Complex, Lamington, Springbrook	Sedge			
		Helichrysum lindsayanum		R	Main Range, Moogerah Peaks, Springbrook	Shrub	Late Winter–Spring		
		Leptospermum sp.	En.		Moogerah Peaks, Mt Barney Complex, Lamington	Shrub			
		Leucopogon lanceolatus var. *lanceolatus*	D		Main Range, Mt Barney Complex, Lamington, Springbrook	Shrub	Late Winter–Spring		Most seed drop to ground, and may be animal dispersed
		Leucopogon melaleucoides	En.		Mt Barney Complex, Lamington Springbrook	Shrub	Late Winter–Spring		Suckers. Observed: seed regeneration 6 years following fire
		Leucopogon spatheceus	En.	R	Springbrook NP	Shrub	Spring		
		Lycopodium deuterodensum	U		Main Range, Mt Barney Complex, Lamington, Springbrook	Fern	–		
		Ozothamnus whitei		R	Mt Barney Complex, Lamington	Shrub	Spring–Autumn		
		Plectranthus alloplectus	En.	R	Moogerah Peaks, Mt Barney	Fern	Late Winter–mid Summer		
		Pomaderris ledifolia	En.		Mt Barney summit	Shrub	Spring		
		Prostanthera phylicifolia	D		Main Range, Mt Barney Complex, Lamington, Springbrook	Shrub	Spring		
		Pultanaea daphoides	N		Mt Barney Complex, Springbrook	Shrub	Spring–Summer		Long lived seeds
		Pultanaea pycnocephala	D	R	Lamington	Shrub	Spring		Obligate seeder, Long lived seeds
		Pultanaea whiteana	En.	R	Mt Barney and Mt Maroon	Shrub	Spring–early Summer		Seeds probably long lived
		Rulingia salviifolia	En.	R	Mt Barney Complex, Lamington	Shrub	Spring		
		Thelionema grande	N	R	Mt Barney Complex	Herb	Early Summer		

		Wahlenbergia scopulicola		R	Mt Barney Complex, Springbrook	Shrub	All year		
		Westringia blakeana	En.	R	Mt Barney Complex, Lamington, Springbrook	Shrub	Spring		
		Westringia rupicola	En.	V	Lamington NP (Caves Circuit, W. of Binna Burra) Springbrook NP	Shrub	Spring–Autumn		Reprouter (with short lived seed bank), and juvenile period of 4–5 years
		Xanthosia diffusa	En.		Mt Barney Complex, Lamington, Springbrook	Shrub	Spring		Seed regeneration
		Zieria granulata var. *adenodonata*	En.	R	Lamington NP	Shrub	Late Winter–Spring		Obligate seeder. Seeds fall to ground and possibly bird and wind dispersed. Seed viability 2–3 years, with juvenile period of 6 years
	6m & 7m *Eucalyptus dura – E. acmenoides* & 11 assoc. shrublands	*Acacia floydii*	En. N	R	Mogerah Peaks, Mt Barney Complex	Tree			Long lived seeds
		Arundeinella grevillensis	En.	R	Mt Greville	Grass			
		Astrotricha biddulphiana	D		Mt Barney Complex	Shrub	Spring		
		Bossiaea rupicola	D		Main Range, Moogerah Peaks, Mt Barney	Shrub	Winter–Spring		Seeds long lived. Can resprout after 'mild' fire
		Callistemon montanus	En.		Mt Barney Complex, Lamington, Springbrook	Shrub	Mainly in Spring		Suckers
		Comesperma breviflorum	En.	R	Moogerah Peaks, Mt Barney	Shrub	Early Summer		Seeds may be long lived
		Eucalyptus notabilis	D, N		Mt Barney Complex, Lamington	Tree	Summer		Probably quite tolerant of high intensity fire, but regular successional fire may reduce seed production significantly
		Grevillea linsmithii (possible upgrade to Endan.)	En.	R	Moogerah Peaks on Mt Greville & Mt Moon, main Range NP (Mt Bangalora)	Shrub	Winter–Spring		Obligate seeder, regeneration pulsed. Seed viability of 2 years. Wind dispersed seeds and juvenile period of 2 years
		Hakea sp. 1	En.		Mt Barney & Mt Maroon	Shrub	Late Winter		
		Hibbertia hexandra		R	Moogerah Peaks, Mt Barney, Lamington, Springbrook	Shrub	Spring–early Summer		
		Hibbertia monticola	En.	R	Mt Barney Complex	Shrub			
		Hibbertia sericea	N		Main Range, Mt Barney Complex	Shrub	Most of year		
		Melaleuca groveana		R	Moogerah Peaks	Tree	Spring		Resprouts
		Phelabium gracile	En.	R	Mt Greville	Shrub	Autumn–Spring		
		Plectranthus suaveolens		R	Mt Barney Complex (Rocky Outcrops)	Herb	May–December	January--July	Avoid high intensity burns and minimise wildfire
		Pomaderris lanigera	D, N		Mt Barney Complex, Lamington	Shrub Tree	Spring		
		Ricinocarpus speciosus	D, U		Mt Barney Complex	Shrub	July–October	October–December	Avoid high intensity burns and minimise wildfire
		Westringia sericea	En.	R	Mt Edwards & Mt Greville	Shrub	Spring and Autumn		
		Zieria fraseri	En.		Mt Maroon & Mt Ernest	Shrub	Spring		

	2n *Eucalyptus grandis* communities	*Archidendron muellerianum (syn. Pithecellobium mullerianum)*	En.	R	Springbrook	Tree	Summer	March–October	Fire tolerance is unknown, but known to sucker
		Argophyllum nullumense	N, D	R	Lamington, Springbrook	Shrub Tree	Summer	May–September	Probably tolerates low intensity fire, and suckers
		Cassia marksiana	D	R	Lamington, Springbrook	Tree	Early Summer	June–August	Usually poor seed viability from grub attack. Suckers follow moderate intensity fire
	4o *Eucalyptus seeana* communities	*Eucalyptus seeana*		D	Mt Barney Complex	Tree	Spring–early Summer		
	2p *Eucalyptus deanei* communities	*Eucalyptus deanei*	D, N		Main Range NP	Tree	Late Summer–Autumn		
	4q *Eucalyptus amplifolia*	*Eucalyptus amplifolia*	N		Main Range NP	Tree	Spring–Summer		
	6r *Eucalyptus racesmosa, E. tindaliae* and *Corymbia gummifera* communities	*Eucalyptus notabilis*	D, N		Mt Barney Complex, Lamington	Tree	Summer		Probably quite tolerant to high intensity fire but regular successional fires may reduce seed production dramatically
		Eucalyptus racemosa			Lamington	Tree	Winter–Spring		Resprouts after fire
		Pultanaea pycnocephala	D, N	R	Lamington	Shrub	Spring		Obligate seeder, with long lived seeds
Shrubland	Other Shrubland species	*Bertya pinifolia*		V					
		Bertya sp. “Mt Ernest”		V	Mt Barney NP (Mt Ernest)	Shrub			Obligate seeder. Seeds drop to ground and possibly ant dispersed. Short lived seed bank, with a juvenile period of 2–3 years
		Callitris rhomboidea	D		Main Range NP	Tree			Obligate seeder, wind dispersed. Has very limited tolerance to fire
		Helichrysum lindsayanum		R	Moogerah Peaks, Mt Barney		Late Winter-_Spring		

		Phelabium elatius subsp. *beckleri*	En.	E	Mt Barney NO (Mt Lindesay)	Shrub	Spring		Obligate seeder, with seed banks exhausted by disturbance. Seed viability: 2–3 years, with a juvenile period of 3 or more years
		Pomaderris crassifolia	En.	V	Main Range NP (Steamers), Mt Barney NP (Mt Ernest)	Tree			
Rock Pavements	8 & 13 Rock pavements, open woodlands of steep, rocky areas	*Botriochloa bunyensis*	D	V	Main Range NP	Grass			Resprouter. Seeds falls to ground and short lived. A **3–5 year fire regime** is suggested
		Brachyscome ascendens		R	Lamington	Herb			
		Cyperus rubicola		R	Lamington	Sedge			
		Deyeuxia rodwayi	U		Lamington	Grass	Spring–Summer		
		Doryanthes palmeri	N		Main Range, Springbrook	Herb	Spring–Summer		
		Euphrasia bella	En.	V	Lamington NP (Mt Merino), Mt Barney (Double Peak & Mt Ballow)	Grass	Spring		Seeder, but seed longevity is unknown. Lifespan < 10 years
		Helichrysum lindsayanum		R	Moogerah Peaks, Mt Barney		Late Winter–Spring		
		*Ozothamnus whitei (*syn. *Helichrysum whitei)*		R	Mt Barney Complex, Lamington	Shrub	Spring–Autumn		
		Plectranthus argentatus	U, En.		Main Range, Lamington	Shrub	Late Summer–Autumn		
		Podolepsis monticola	En.	R	Lamington NP	Herb	Spring–Summer		
		Sarcochilus fitzgeraldii	D	E	Lamington NP & Springbrook NP (rocks and cliffs)	Epiphyte Orchid	Spring		Obligate seeder, seeding after disturbance. Seeds: wind dispersed and viable for 2–6 weeks. Juvenile period of 3 years
		Sarcochilus hartmanii	D	V	Main Range NP (Spicers Peak), Lamington NP, Springbrook NP, Mt Barney NP (Mt lindesay)	Epiphyte Orchid	Spring		Obligate seeder. Seeds: wind dispersed and viable for 2–6 weeks. Juvenile period of 3 years
		Wahlenbergia glabra		R	Main Range NP, Lamington	Herb	All year		
		Westringia blakeana	En.	R	Mt Barney, Lamington, Springbrook	Shrub	Spring		
		Westringia rupicola	En.	V	Lamington NP, Springbrook NP	Shrub	Spring–Autumn		Resprouter with short-lived seed bank. Juvenile period 4–5 years
Other		*Arundinella montana*		R	Moogerah Peaks	Grass	–		
		Pimelea umbratica		R	Main Range NP	Shrub	–		
Eucalypt forests w. heath understorey, on rocky outcrops above 800 m		*Pomaderris crassifolia*	En.	V	Main Range NP (Steamers), Mt Barney (Mt Ernest)	Tree	–		Resprouter. Regenerates from seed every 2–3 years, seeds drop to ground. Juvenile period of 3 years