The role and use of fire for biodiversity conservation in Southeast Queensland:

Fire management guidelines derived from ecological research

July 2001

Penny Watson
Project Coordinator
SEQ Fire and Biodiversity Consortium

Fire & Biodiversity CONSORTIUM

The SEQ Fire and Biodiversity Project is sponsored by Logan City Council, and funded by the Commonwealth Government's National Heritage Trust through the Bushcare Program.

Bushcare
A program of the Natural Heritage Trust

AUSTRALIA
Natural Heritage Trust
Helping Communities Helping Australia
A Commonwealth Government Initiative

The role and use of fire for biodiversity conservation in south-east Queensland: Fire management guidelines derived from ecological research

Penny Watson
SEQ Fire and Biodiversity Consortium

The South-east Queensland Fire and Biodiversity Consortium aims to gather and disseminate information on fire management practices that will support conservation of the region's biological diversity.

The project covers the area from Noosa Shire in the north to the New South Wales border, and from the islands of Moreton Bay to the Great Dividing Range in the west.

July 2001

Table of Contents

Acknowledgments

Many people contributed ideas and insights that are reflected in this report. The work by Cuong Tran and Clyde Wild, which preceded this in the Fire and Biodiversity Consortium agenda, was both useful and inspiring, as were discussions with these colleagues throughout the project. Thanks go to all who commented on the draft guidelines, particularly David McFarland, Carolyn Sandercoe, Cuong Tran, Robyn James, Rhonda Melzer, Paul Williams, Paul Donatiu, Wendy Drake, Anthony Thomas, Geoffrey Smith, David Crossley, Rod Fensham, Christine Pfitzner and Wayne Kington. Input from these reviewers, reflecting their depth of understanding of SEQ's natural heritage and their enthusiasm for the subject under discussion, has been invaluable in refining the document. The energetic and supportive FABC Steering Committee – Tanya Pritchard, Bernard Trembath, Paul Donatiu, Christine Pfitzner, Cuong Tran, Sonia Culley, Clyde Wild, Cherene Mitchell and Carmel Peacock - provided a sounding board and helped determine which way to go with some questions of content and style. The wider network of Consortium members and supporters also provided encouragement and insights through numerous discussions, and through their keen interest in the role of fire in biodiversity conservation. Thanks go to all.

Introduction

This document aims to provide native vegetation managers in south-east Queensland with guidelines to assist them in understanding and planning fire management for bushland areas. The primary audience is people employed in land management, extension, planning and policy roles who have some background in ecology and who are seeking a moderately in-depth understanding of what the literature has to say.

Fire is an integral part of many of south-east Queensland's diverse vegetation types, notably its sclerophyll and grassy ecosystems. Many plants and animals in these communities do not merely tolerate burning; rather fire plays an important and positive role in their life-cycle. However species are not adapted to fire *per se*, but to a particular regime of fire. Inappropriate fire regimes affect community structure and composition, and may even lead to species extinctions (Gill and Bradstock 1995). If we want to conserve the full range of species and communities currently found in south-east Queensland, then the role of fire cannot be ignored.

For those responsible for these fire-adapted communities, be they private landholders or public land managers, fire management is a key task. Managers must decide whether to initiate planned burns, and where, when and how to carry them out. Plans as to where and how to stop a wildfire, if one occurs, must be developed. While life and property issues always need to be considered, a sound understanding of the effects of fire on native plants and animals should also inform management plans and strategies.

While these guidelines are designed to assist managers with their fire management role, they do not offer a set formula. Fire management is a complex process and unique local factors must always be addressed. The aim is to outline key points for consideration in local planning and practice.

These Guidelines fulfil one of the objectives of the South-east Queensland (SEQ) Fire and Biodiversity project. This project, which is funded by the National Heritage Trust through the Bushcare program, is auspiced by Logan City Council on behalf of the SEQ Fire and Biodiversity Consortium (FABC). The Consortium includes representatives from local authorities throughout SEQ, the Rural Fire Service, the Queensland Parks and Wildlife Service (Parks and Forest Management divisions), the Department of Primary Industries, Landcare, Greening Australia Queensland, and universities. Other Consortium products include an individual property fire management planning kit and fact sheets targeted to private landholders (see p. 49).

The Guidelines have been informed by an earlier Consortium document, "A Review of Current Knowledge and Literature to Assist in Determining Ecologically Sustainable Fire Regimes for the Southeast Queensland Region" by Cuong Tran and Clyde Wild from Griffith University, released in August 2000. Both that document and this have been distilled through many discussions with fire and land managers, researchers and extension staff throughout SEQ and beyond. The guidelines represent the author's best understanding of the literature in the light of her own analysis and this input. However they naturally remain to some extent subjective, and readers are invited to critique, discuss and draw their own conclusions.

Assertions in the text are supported through citing research findings and expert opinion. Fire ecology is a vast field encompassing many questions, only some of which have been investigated, and very few through research in south-east Queensland. While local references have been used where possible, findings from other parts of Australia and even overseas have been cited where the author believed they would provide relevant insight. Readers are encouraged to seek out documents cited in the text, and use them to explore directly the issues most relevant for their fire management.

As research enlarges our understanding of our fire-adapted ecosystems, some of these guidelines will undoubtedly need modification. While we know enough at this stage to draw tentative conclusions as to appropriate fire management strategies for broad vegetation types, research to date has merely scratched the surface of what could be investigated (Tran and Wild 2000). The Consortium aims to encourage fire ecology research in south-east Queensland, with a view to fulfilling another project goal, that of "filling the gaps" in our knowledge of appropriate fire regimes.

The author has sought to arrange the guidelines in logical order, and to bring out important points as early as possible in the document. However inevitably some points must follow others. Managers are encouraged to read the entire document, so individual guidelines are seen in context. Guidelines take the form of summary statements followed by explanatory text.

The document is divided into three sections. The five guidelines in Part A provide a broad context for those that follow. Topics include perceptions of fire, the place of fire management for biodiversity in a multi-use landscape, management aims and background ecological information. Part B covers nine general guidelines for managing fire for biodiversity conservation. The initial points focus primarily on fire frequency, while later guidelines in this section address questions of spatial pattern and season. Appropriate fire regimes for broad vegetation types are discussed in Part C. The discussion in this section assumes consideration of issues addressed in Parts A and B.

A. Contextual guidelines

1. Perceptions of fire

Fire is an integral part of most south-east Queensland ecosystems, and therefore should not be considered unnatural or catastrophic *per se*.

Most south-east Queensland ecosystems are adapted to fire. While some communities such as rainforests and mangroves do not benefit from being burnt, many others *require* fire to maintain the diversity and vigour of the organisms and processes within them. Heathlands, grassy woodlands, shrubby forests and tall open forests all fall into this category.

For many plants in these communities, fire is a catalyst for regeneration.

- Fire can act as a trigger for **seed release**. *Banksia* and *Hakea* species are typical examples, although species within these groups differ in the extent to which they retain seed on the plant in the absence of disturbance (Cowling *et al.* 1990). Some eucalypts also release seed in response to fire (Noble 1982, Gill 1997). For example many freshly-fallen *E. planchoniana* capsules with seeds beside them were observed in Toohey Forest after a recent planned burn (P. Donatiu, pers. comm. 2000).

- Heat promotes **germination** in many species, particularly legumes (Shea *et al.* 1979, Auld and O'Connell 1991). Smoke too plays a major role for some species (Dixon *et al.* 1995, Enright *et al.* 1997, Roche *et al.* 1998).

- Fires influence **nutrient availability**, providing conditions which promote seedling growth (Williams *et al.* 1994, Cheal 1996, Florence 1996).

- Fires **open up the canopy** and remove shrubs, grass clumps and litter. Light penetration is increased and competition for water and nutrients reduced, encouraging seedling growth (Sandercoe 1989, Williams and Gill 1995, Morgan 1998).

- Fires can counteract **microbial factors** in the soil which may inhibit germination and seedling growth (Florence and Crocker 1962, cited in Florence 1996).

- Fires can act as a cue to **flowering** and subsequent seed release. *Xanthorrhoea* species are the best-known example of this phenomenon (Harrold 1979, McFarland 1990), however other plants, such as *Lomatia silaifolia* (Denham and Whelan 2000), *Blandfordia* spp. (Sandercoe 1991, Johnson *et al.* 1994), and *Telopea speciosissima* (Bradstock 1995) also flower almost exclusively in the years after fire.

Many animal and bird species depend on the habitat provided in the early and middle years of post-fire regeneration (Fox 1983, Friend 1993, Hannah *et al.* 1998, Tasker *et al.* 1999, Woinarski 1999 – see Guideline 3 for further information). Some macropod species, for example, utilise the fresh grass that flourishes after a burn, as do granivorous birds. Without periodic fire, these species may not be able to persist in an area (Kington 1997).

The character of some fire-adapted vegetation types can be lost altogether if fire is excluded for an extended period. Certainly this applies to some wet sclerophyll forests, which develop into rainforest in the absence of fire (Unwin 1989, Harrington and Sanderson 1994, Chapman and Harrington 1997), and to grassy woodlands which may become dominated by species such as *Allocasuarina littoralis* (Lunt 1998b) and *Lophostemon confertus* (Kington 1997). Even long-unburnt heathland can be vulnerable to proliferation of shrubs such as *Leptospermum, Kunzea* and *Melaleuca* (Burrell 1981, cited in Attiwill 1994; W. Drake, pers. comm. 2001).

2. Addressing multiple objectives through zoning

Fire managers need to consider both ecological and protection/production objectives. Zoning offers opportunities for addressing multiple fire management goals.

Many south-east Queensland residents live close to bushland, making property protection an essential fire management objective. Grazing, agriculture and farm forestry are important land uses. Fire management in these circumstances will generally involve either fire exclusion, or the use of fire at shorter intervals than those recommended for biodiversity conservation.

To meet protection, production and ecological objectives, managers can designate zones which are managed in different ways. If areas immediately adjacent to housing and other assets are managed so as to keep fuel loads low, then other areas can be more confidently managed through fire regimes appropriate for biodiversity conservation (Rose *et al.* 1999). Further information on property protection measures can be obtained from the Rural Fire Service, and from the Fire and Biodiversity Consortium's Individual Property Fire Management Planning Kit.

One implication of this point is that options for conservation management decrease as bushland becomes fragmented. Where houses are dotted through a fire-adapted landscape, fire management must focus increasingly on property protection, leaving proportionately less opportunities for regimes consistent with conserving biodiversity. Here is yet one more reason for maintaining large intact areas of bushland, a practice also recommended by those who have studied other effects of fragmentation on biodiversity (Diamond 1975, Janzen 1983, Saunders *et al.* 1991, Soulé 1991). This point should be noted by those involved in land use planning in both urban and rural areas.

3. Ecological background

Understanding the ecological impacts of fire involves an appreciation of:
- **plant species life history characteristics**
- **post-fire succession, and**
- **the role of abiotic factors other than fire**

Life history characteristics

Plants in fire-adapted vegetation types have two major ways of keeping their place in the community. Gill (1981) classified plants as "non-sprouters" or "sprouters", on the basis of whether mature plants subjected to 100% leaf scorch die or survive fire.

Most adults of **sprouting** species, also called "resprouters" or "fire tolerant plants", regrow from shoots after a fire. These shoots may come from root suckers or rhizomes, from lignotubers, from epicormic buds, or from active pre-fire buds (Gill 1981). Some resprouters, ie those which regrow from root suckers or rhizomes (such as blady grass and bracken), can increase vegetatively after a fire. However other resprouters cannot increase vegetatively, and therefore need to establish new plants to maintain population numbers, as adults will eventually age and die. *Banksia spinulosa, Eucalyptus planchoniana, Alphitonia excelsa, Strangea linearis* and *Boronia rosmarinifolia* are examples of resprouting species in south-east Queensland.

The adults of **non-sprouting species**, also sometimes called "obligate seeders" or "fire sensitive" plants, die when their leaves are all scorched in a fire. These species rely on seed regeneration to maintain their presence in the community. Obligate seeder species generally produce more seed (Lamont *et al.* 1998), and greater numbers of seedlings (Harrold 1979, Wark *et al.* 1987, Benwell 1998) than resprouters, and seedling growth rates tend to be faster (Bell and Pate 1996, Benwell 1998). *Allocasuarina littoralis, Pultenaea villosa, Boronia keysii, Eucalyptus grandis* and *Hakea actites* are example of south-east Queensland species which typically regenerate from seed after fire.

These categories are not invariant. Survival rates in the field for both resprouters and obligate seeders change with fire intensity (Morrison and Renwick 2000), as leaf scorch is rarely exactly the "just 100%" specified by Gill (1981). Some species exhibit different regeneration strategies in different environments (Williams *et al.* 1994, Benwell 1998). Examples from SEQ in include *Ricinocarpos pinifolius* and *Monotoca scoparia,* which act as obligate seeders in woodland habitats but resprout in dry heath (Sandercoe 1989, W. Drake, pers. comm. 2001). Differences in regeneration strategy between species found at Cooloola and at Myall Lakes have also been identified (Sandercoe 1989).

Some species can regenerate by seed in the interval between fires, while others are limited to the immediate post-fire period (Cheal 1996, Keith 1996). Noble and Slatyer (1980) have developed a "vital attributes" model which builds on Gill's classification. Their model includes factors associated with seed production and dispersal, and adds a time dimension. This model can be used to predict how species, and even communities, may be affected by particular fire regimes.

Post-fire succession - flora

In the years after a fire, plant communities change in structure, dominance, and above-ground composition. Immediately after a fire, bare ground is plentiful. Shoots of resprouting species are the first signs of life to appear, then with adequate soil moisture, seedlings can be found. In many systems, species richness peaks in the early years post-fire, while both short-lived herbs and more persistent species are present (Specht *et al.* 1958, Posamentier *et al.* 1981, McFarland 1988a). At this stage the vegetation structure is relatively simple and open (Coops and Catling 2000). Grass and herb species grow and flower (Harrold 1979, Wark *et al.* 1987, McFarland 1990, Enright *et al.* 1994), however some shrub species will not yet have reached reproductive maturity (Harrold 1979, Benson 1985). Over the years, litter builds up and structure becomes more complex (Coops and Catling 2000). Canopy cover is restored. In shrubby systems, shrub cover may become so thick as to be almost impenetrable. Some plant species decrease sharply in abundance or disappear as competition for light and other resources becomes increasingly intense (Specht *et al.* 1958, Russell and Parsons 1978, McFarland 1988a, Sandercoe 1989). Many of these species will remain present, but invisible, in the soil seedbank (Posamentier *et al.* 1981, Gill and Bradstock 1995), while others persist through underground structures such as bulbs. After a number of years overall reproductive effort declines (McFarland 1990, Lunt 1994). Even-aged stands of seeder species which developed after the fire, for example some legumes, age and eventually die. Thus when a fire has not occurred for many years, the shrub layer may thin out, although litter will still be thick.

Post-fire succession – fauna

The cyclical sequence in vegetation composition and structure is parallelled by changes in the relative abundance of fauna species with time since fire. Successional effects have been identified for mammals, birds, reptiles, and invertebrates.

Studies from a range of vegetation types have confirmed that different small ground-dwelling **mammal** species dominate burnt areas at different times after fire. A typical sequence begins with *Pseudomys* spp. (eg the New Holland Mouse *P. novaehollandiae*) – and the introduced House Mouse (*Mus musculus*). The dasyurids are next, with the Common Dunnart (*Sminthopsis murina*) tending to precede the Brown Antechinus (*Antechinus stuartii*). Bush Rats (*Rattus fuscipes*) and Swamp Rats (*Rattus lutreolus*) occupy later regeneration niches (Fox and McKay 1981, Fox 1982, Fox 1983, Friend 1993, Wilson 1996). A study by Catling and Burt (1995) that linked abundance with habitat complexity found Brown Antechinus and Bush Rats preferred the complex habitat which develops some years after a fire, while larger herbivorous mammals such as the Eastern Grey Kangaroo (*Macropus giganteus*) preferred an open understorey with few shrubs and good grass cover.

Immediately after or even during a fire, insectivorous and carnivorous **bird** species such as raptors, kookaburras, ibis and crows may be attracted to the area (McFarland 1988b, Woinarski 1999). Open country species such as those that feed on grass seed dominate in the early post-fire years (Woinarski 1999). McFarland (1988b) identified a number of species which favoured early regeneration in the Cooloola coastal heath, including Brown Quail (*Coturnix australis*) and Richard's Pipits (*Anthus* *novaehollandiae*). Ground Parrots (*Pezoporus wallicus*) and Brush Bronzewings (*Phaps elegans*) were most abundant in mid-aged heaths, while the White-cheeked Honeyeaters (*Phylidonyris nigra*) and Yellow-tailed Black Cockatoos (*Calyptorhynchus funereus*) reached highest densities in heaths which had not been burned for over 10 years (McFarland 2000). Black-breasted Button-quail (*Turnix melanogaster*) inhabit long-unburnt dry sclerophyll forest, where litter is thick (Hughes and Hughes 1991).

Some **reptiles** also appear to favour different post-fire habitats (Friend 1993, Wilson 1996). Hannah *et al*. (1998) found three lizard species – all from the genus *Lampropholis* – were much more abundant, in plots that had been long unburned than in plots which were recently, burned. Conversely, two species from the genus *Carlia* were more abundant in recently burnt plots. This Queensland study (from Bauple near Gympie) did not find any clear relationship between burning history and amphibian abundance.

Invertebrate species assemblages also vary with time since fire. Groups associated with leaf-litter, such as ticks and mites, are more abundant in the later post-fire years (Norris and Conroy 1999, York 1999). Species which are abundant in the early years after a fire tend to be generalists, or species adapted to drier and more open environments (York 1999).

Other abiotic factors

Factors other than fire such as soil type, rainfall, and topography, also have a major effect on the distributions of animal and plant species and communities. Interactions can occur. For example, soils are often deeper and richer in nutrients in gullies than on slopes. These factors in themselves mean that the vegetation is likely to be more mesophytic (moisture-loving), and canopies more closed, in these sheltered places (Ash 1988, Florence 1996). Soil moisture is therefore better retained. Fires that burn the vegetation on slopes will often skip or stop when they reach the wetter gully vegetation, partly because fire burns more slowly downhill. This provides more opportunity for fire-sensitive mesophytic species to grow.

The patchwork of vegetation that results from the interaction of fire with other abiotic factors has implications for biodiversity conservation. Animals seeking shelter from a fire burning through a dry vegetation type may find refuge in unburned creekside patches. Some species may utilise this habitat until post-fire regeneration renders their favoured habitat accessible. The location of different vegetation types in a landscape may shift over time, as boundaries move in response to fire frequency. "Ecotonal" plant species find a home in the movable zone between the drier and wetter systems (W.J. McDonald, pers. comm. 2000).

4. The fire regime

While the effects of a single fire are often dramatic and easily observed, it is the more subtle impact of the *fire regime* – the frequency, intensity, season, type and extent of fires *over time* – which is likely to affect biodiversity and ecosystem processes.

The concept of the fire regime is basic to the study of the effects of fire on plant and animal communities, as it identifies the components of fire which may determine vegetation change (Gill 1977), and points out the need to consider the effects of not just one, but a sequence of fires (Keith 1996). Gill (1977) initially identified four fire regime components: type, frequency, intensity and season. Extent, or patchiness, can be added to that list (Whelan 1995).

Many of these factors inter-relate. For instance, where there is a short interval between fires, fuel will not have had time to build up, leading to a greater likelihood of a low intensity burn than if the interval between fires has been long. Low intensity fire is more likely to be patchy (Whelan 1995). Season of fire may affect intensity, patchiness and type of burn. Vegetation with a high moisture content will burn less intensely than dry. Conversely, during extremely dry times intense burns are more likely and these could include both types of fire – above ground in vegetation, and below ground in peat beds.

5. Management aims in conservation areas

Management aims may influence choice of fire regimes in particular conservation areas. It is suggested that the needs of the community as a whole be considered wherever possible, rather than focussing exclusively on a single species.

Obviously, clear management objectives are a prerequisite for any level of fire management planning.

This document is based on the premise that the management aim is to retain biodiversity at a landscape level. The study of fire ecology alerts us to the fact that species complements change over time and space. It is at landscape level, rather than at a small scale, that we can realistically aim to retain the species complement of a community.

Sometimes, of course, there will be a more specific management aim. For example, the aim may be to maximise population numbers of a particular rare or vulnerable species. Fire regimes would then be tailored to that species' needs. It is important to recognise, however, that this practice may not be compatible with retaining all components of the community in that local area, given species' variable adaptations to fire (see Guideline 6). While this trade-off may be appropriate in some areas, it is generally suggested that fire management consider the needs of the community as a whole. Occasionally there may be an argument for maintaining some areas within a vegetation type on longer intervals to retain or enhance the abundance of particular species, while other areas are managed with shorter intervals to benefit other species (Lunt 1997, Bradstock 2000).

Another management aim is to reduce weed invasion. These guidelines do not attempt to address that topic. Suffice to say that the extent to which fire will control a particular weed will depend on its characteristics vis-à-vis those of other species in the community. Authors agree that fire alone is unlikely to be effective against weed invasion, but that fire may have a place when used in conjunction with other control strategies (Lunt 1990, Downey 1999). For example, fire may improve access to weedy sites, and vigorous post-fire regrowth may be particularly susceptible to herbicide. It is also important to note that in places where weed propagules are readily available, fire may encourage weed invasion. Precautions such as silt traps at the heads of gullies may be advisable (Melville 1995).

Monitoring results to assess the degree to which objectives have been achieved is also good management practice. This task will be relatively simple where the aim relates to one or two species. Assessing the effects of fire management strategies on overall biodiversity at a landscape level is more of a challenge, and will not be explored in detail here. Some clues may be obtained from indicator species, although the relationship between species characteristics and overall biodiversity would need to be very clear before much reliance could be placed on this technique.

B. General guidelines for managing fire for biodiversity conservation

6. Fire regime variability in and between communities

a. Fire adapted communities and their inhabitants are not adapted to fire *per se*, but to a particular regime of fire. Different ecological communities need different regimes of fire.

b. A range of frequencies, intensities, seasons and scales of burning, within the ecological constraints of each community, should be incorporated into fire regimes in order to optimise conservation of biodiversity.

Fire-adapted ecosystems and the organisms within them are generally resilient in the face of a single fire (Tolhurst 1996). While individual animals and plants may die, species in fire-adapted systems generally have efficient recovery mechanisms. However different species recover at different rates: for some species, post-fire recovery is rapid, while others may take quite a few years (Benson 1985; Fox and McKay 1981). Field studies have also shown that some species benefit from relatively frequent fire, while others are more abundant where fire has been less frequent. This variability in response is apparent even within one community (Fox and Fox 1986, Watson 1999), and is even more salient across different broad vegetation types. In a field study, Morrison *et al.* (1995) found variability in interfire intervals in dry sclerophyll communities around Sydney was associated with greater species richness in both resprouting and obligate seeder species. Fire intensity and season can also affect different species differently, with particular intensities and seasons favouring some species over others (Clark 1988, Birk and Bridges 1989, Auld and O'Connell 1991, McLoughlin 1998, Morrison and Renwick 2000).

Thus over time, a range of frequencies, intensities and seasons of burn is recommended, as this gives each of the varied ecosystem components an opportunity to experience at least some fires which are to their 'liking'. A constant regime, for example one of low intensity burning every five years in August, will inevitably advantage some species over others, and will eventually result in those species for which this is not a preferred regime becoming less abundant, or even locally extinct. Biodiversity is more likely to be maintained through a series of fires at intervals of, say, between three and six years encompassing a wider range of months of the year. Intensity would probably vary naturally if this more varied prescription for frequency and season were adopted.

However, this frequency range needs to lie within broad limits appropriate for the particular vegetation type. The above example, for instance, would be appropriate in a grassy woodland, but not in a heath or shrubby forest, as the interfire intervals would be too short for some species of plants and animals which inhabit these communities. The limits for each vegetation type can be determined through considering the needs of flora and fauna species most at risk of being lost from the community as a result of either very frequent, or very infrequent, burning (Gill and McCarthy 1998).

7. Inappropriate fire regimes

Both too frequent and too infrequent burning can result in the loss of species from an area.

Fire-adapted ecosystems may lose species if fire is either too frequent, or too infrequent. These terms denote different frequencies in different vegetation types (see Guidelines 6, 8 and Part C) however the general principle applies across all fire-adapted vegetation types.

Problems of too frequent burning

Frequent burning may result in the loss of obligate seeder plant species. If a second fire takes place before seedlings have reached full reproductive maturity (see Guideline 9), there may be little or no seed available to germinate after the fire. Some species with long-lived soil-stored seeds may retain viable ungerminated seed through two or more fires, eg due to variations in depth of burial and patchiness in fire intensity – *Bossiaea laidlawiana*, from south-west Western Australia is an example (Christensen and Kimber 1975). However there will obviously be limits to this strategy if burning continues to occur within the obligate seeder's juvenile period. Other species may be able to grow from seed dispersed from outside the burnt patch, though dispersal distances are probably limited to a few metres or tens of metres in the large majority of species in Australian fire-prone vegetation (Keith 1996, Hammill *et al.* 1998). Frequent fire may be particularly problematic for obligate seeder species that store relatively heavy seeds on-plant, as these species have no soil-stored seed to survive through a second fire, and dispersal distances tend to be short. These factors may explain the finding that *Banksia ericifolia* and *Hakea teretifolia* are the species most often affected by frequent burning in Sydney's Hawkesbury Sandstone heaths and woodlands (Siddiqi *et al.* 1976, Nieuwenhuis 1987, Cary and Morrison 1995, Bradstock *et al.* 1997).

Resprouting species can also be negatively affected by frequent fire. Watson (1999) found that a number of resprouting species, including *Lomatia silaifolia*, *Hakea dactyloides*, *Jacksonia scoparia* and *Hardenbergia violacea* (all species found in south-east Queensland), were much reduced in abundance in frequently burned woodlands in Girraween National Park near Stanthorpe. Keith (1996) points out that the time taken for the seedlings of resprouters to develop fire tolerance is analogous to the juvenile period in obligate seeders. Successive fires at intervals less than this duration will mean no recruitment, and populations will decline as established plants die. Frequent fires can also negatively affect resprouting adults as bark is progressively thinned or bud reservoirs are depleted (Clark and McLoughlin 1986). Frequent experimental burning can eliminate mallee eucalypts (Noble 1982), and resprouting melaleucas (S. Skull, EPA, pers. comm. 1999).

We have already seen that many mammal, bird, reptile and invertebrate species do not find suitable habitat in areas that have been recently burnt (see Guideline 3). Where fire frequency is high, the habitat needed by these species never develops, and thus over time they will be lost from the community. Frequent fire may also hasten the destruction of hollow-bearing trees (G. Smith, pers. comm. 2001). Catling (1994) names 25 native mammal species which he believes will be disadvantaged by frequent fire, which he sees as a further factor in habitat fragmentation:

> "The long-term effect of the prescribed burning regime currently used in the management of the drier forests of south-eastern Australia will reduce the habitat suitable for native fauna to islands. Suitable forest habitat will only remain along creeks and drainage lines or in places where vegetation can recover quickly after fire. Whereas the clearing of forests into disjoint islands is obvious, the reduction of faunal habitat due to prescribed burning is insidious and not readily apparent. Such restriction of the wildlife to small pockets greatly increases the risk of predation by dingoes, foxes, cats and raptors." (Catling 1994:39)

Catling (1991:353) argues for "the cessation of a monotonous application of one prescribed burning regime, and the use of a range of fires, with an occasional planned intense fire inserted in the regime."

Problems of too infrequent burning

Long-term fire exclusion can also cause species to be lost. Plant species which rely on fire for regeneration will eventually become old and die. If a second fire does not occur before the seeds of these species lose viability, then they will be lost from the community, along with any animal species reliant on them. A modelling study of an obligate seeder Banksia species found senescence was the primary mechanism causing extinction (Bradstock *et al.* 1996). In grassy ecosystems the dominant clump grasses rapidly occupy the spaces needed by the forbs, sub-shrubs and less dominant grasses which grow in this vegetation type. Without fire, regeneration of these less dominant ecosystem components may not occur (Lunt 1990, 1994; Morgan 1998).

Aging vegetation often loses productivity (McFarland 1990), reducing food availability. This is one reason why bird and animal species move out of long-unburnt vegetation (McFarland 1988b, Kington 1997).

We have already noted that some fire-adapted vegetation types may be taken over by other vegetation types in the absence of fire (see Guideline 1). Plant species can be lost in this way. For example, *Eucalyptus grandis* seedlings cannot survive in a rainforest understorey, and there are well-grounded fears that this wet sclerophyll species, which also occurs in south-east Queensland, will be lost from North Queensland forests where fire is excluded from the system (Unwin 1989, Turton and Duff 1992, Harrington and Sanderson 1994). With it will probably go a range of bird species which utilise this forest type, two mammal species endemic to the Wet Tropics, and a specialised ant fauna (Chapman and Harrington 1997). Unburnt grassy woodlands are readily taken over by opportunistic shrub species (Kington 1997; Lunt 1998a,b). Fauna which need open grassy areas, such as Pretty-face Wallabies and other macropods (Kington 1997), and many ant species (Vanderwoude *et al.* 1997), will be lost from these areas.

8. Managing within appropriate fire frequency ranges

a. Minimum and maximum interfire intervals conducive to the maintenance of biodiversity can be estimated for broad vegetation types. Planned fire should not be implemented for ecological purposes below the lower limit, while communities approaching the upper limit should be considered for ecological burning.

b. Fires can be planned or unplanned. Both affect the ecology of bushland communities, and both should be considered in fire management planning.

Minimum fire-free intervals for particular communities can be estimated from parameters such as the time taken by obligate seeder species to reach full reproductive maturity, the time for resprouter seedlings to reach fire tolerance, and the time needed for fauna populations to re-establish. Maximum intervals can be estimated from factors such as time to senescence of obligate seeder species, and from studies of fauna successional patterns and plant productivity.

While most species are likely to be resilient in the face of the occasional interval outside the recommended limits, repeated interfire intervals outside those recommended are likely to cause species to be lost. Guidelines 15 to 19 outline the minimum and maximum interfire intervals recommended for broad vegetation types in south-east Queensland, along with the reasoning behind these figures. It needs to be emphasised that these recommendations do *not* equate to endorsement of repeated burning at or near the suggested minimum. Rather, the idea is to encourage variability of interfire intervals within the upper and lower parameters (see Guideline 6). In many areas, an emphasis on intervals at the longer end of the spectrum may be warranted (see Guidelines 9 and 11).

Realistically, fire regimes in many parts of south-east Queensland will continue to encompass fires that have not been planned by those with management responsibility. Thus fire managers need to consider unplanned as well as planned fires when planning for the ecological needs of the bush. The impacts are cumulative irrespective of the source.

In some areas, fires will occur at intervals below the minimum recommended for that vegetation type, courtesy of arson or accident. Here, management will focus on finding ways to reduce the level or impact of these factors. In other areas, fires are unlikely to take place unless the manager carries them out, or mitigates factors preventing them. For example, in some areas a lower than recommended fire frequency may result from grazing pressure on fuel loads (G. Smith, C. Everson, pers. comm. 2001). Roads or clearing may block the ‘natural’ path of wildlife. Where infrequent wildfire forms part of the regime, managers must decide whether additional burning is necessary. We suggest that this should only be considered where intervals approaching or above the maximum would otherwise be the order of the day.

9. The importance of long unburnt vegetation

Longer interfire intervals provide important habitat for some wildlife, and time for plant species to reach their full reproductive potential and develop a seedbank. It is therefore important that areas of relatively long-unburnt vegetation are retained in the landscape.

Catling's contention that frequent burning further fragments an already fragmented landscape (Catling 1994) emphasises the importance of allowing some areas to remain unburned for periods up to the maximum recommended for that vegetation type. Woinarski (1999) points out that many of the threatened bird species whose relationships with fire have been documented show a clear preference for long interfire intervals – the Eastern Bristlebird is a good example (Baker 2000). Tasker *et al.* (1999) found a significantly higher abundance of small mammals in long unburnt sites than in sites burnt approximately every four years. While fauna that prefer relatively open areas may find suitable habitat in farmland and cities, those that need complex habitat are less likely to do so. Longer intervals may also help ensure the rate of tree hollow formation equals or exceeds the rate of hollow destruction (Lindenmayer *et al.* 1997). Thus retention of bushland areas subject to relatively long interfire intervals is vital for the conservation of many fauna species, particularly where other potential habitat is either cleared, or frequently burned.

Some plant species, too, need long intervals to reach their full reproductive potential. It may take several years after an obligate seeder first flowers before seed is set, particularly in quantities sufficient for population replacement in the event of a fire. In some cases, a further period is required for seed to mature (eg in *Banksia* cones). Predation may reduce seed numbers and further delay the build-up of an adequate seedbank (Auld and Denham 2001). For these reasons, fire free periods of at least double the time to first flowering of the most tardy obligate seeder species are recommended by Gill and Nicholls (1989).

While it is true that long-unburnt vegetation is likely to burn more intensely than more frequently burnt vegetation, research has shown that in at least some vegetation types, fuel loads plateau at four to six years post-fire (Sandercoe 1989). And while intense fire may kill more individual birds and animals than more moderate burns, Catling (1994) argues convincingly that the habitat benefits of long-unburnt vegetation outweigh the disadvantages of occasional intense fire for native fauna.

10. Managing fire frequency in a landscape context

Where adjacent vegetation communities are adapted to differing fire frequencies, the less flammable community will probably burn in some, but not all of the fires which affect the more flammable community. Planned burns can focus on the more flammable community.

Within a landscape, vegetation communities may occur in a mosaic pattern. In Brisbane Forest Park, for instance, grassy woodlands tend to occupy ridgetop sites, while scrubs and rainforests occur in the valleys and gullies. The forest becomes progressively more dense and damp as one moves from ridge to valley floor. In this situation, there is a strong argument that fire management should focus on maintaining fairly frequent burning on the ridges. This will not only maintain the grassland vegetation, but will provide some protection from wildfire to the valley rainforests (D. Kington, pers. comm. 2000). Uneven penetration by both planned and unplanned burns into the shrubby slopes may well provide appropriate fire frequencies for these vegetation types, which are adapted to less frequent burning than the ridge-top grassy woodlands. Moisture differentials can be exploited to achieve this objective during a planned burn (Burrows 2000, N. Gourley, pers. comm. 2000). This regime may be similar to that used by Aboriginal people, who may have kept the ridgetops clear both for hunting and for cross-country access. Aboriginal burning in the Sydney area appears to have concentrated on the ridgetops (Clark and McLoughlin 1986).

Gill and McCarthy (1998) argue that many rare species occur in landscape features that are likely to experience a lower fire frequency than the surrounding matrix, such as rocky outcrops and fire shadows along creeklines. The distribution of fire intervals in these areas "may be regarded as secondary, being dependent on the primary fire-interval distribution of the surrounding landscape" (Gill and McCarthy 1998:163). However, sometimes important refugial microhabitats adapted to lower fire frequencies to those of the surrounding matrix may not be so obvious (Wardell-Johnson 2000). Land managers need to be aware of these local landscape issues in their management of fire.

11. Fire frequency variation within a broad vegetation type

Within a broad vegetation type, different communities may be adapted to somewhat different fire frequency distributions.

The fire regimes suggested in Part C relate to very broad vegetation types. Obviously each broad classification will encompass many different species assemblages, and fire regimes may also vary (Williams *et al.* 1994). For instance, grassy woodlands may benefit from more frequent fire than grassy forests. Vegetation on relatively nutrient-poor substrates may take longer to recover, and thus require somewhat longer interfire intervals, than relatively nutrient-rich sites within the same board vegetation type. These are just suppositions: at present we do not have sufficiently detailed information to make these distinctions. Further research is needed.

12. Mosaic burning

Mosaic burning at small-scale and landscape level:

- **maximises habitat for a variety of fauna species**
- **provides opportunities for recolonisation of burnt patches by plants and animals**
- **provides options for managing threatened species populations, and**
- **can assist in meeting wildfire mitigation goals.**

Mosaic burning is frequently recommended as a way of maintaining habitat for the full range of fauna species in a community including early, middle and late successional species (Fox and McKay 1981, Fox 1983, McFarland 1988b, Masters 1993, Hannah *et al.* 1998, Woinarski 1999). Patchiness provides refugia for animals during a fire, a source of food in the early post-fire months and years, and protection from predators. It provides some defence against local extinction of plant species, as individuals and propagules can persist in unburnt remnants and recolonise nearby recently burnt areas (Williams *et al*. 1994, House 1995).

Small-scale patchiness may be useful in managing populations of rare or threatened species, through prescriptions such as burning no more than 50% of the species' home range at one time (Rose *et al.* 1999). In North Queensland, patch burning is used to increase availability of grass seeds for granivorous birds, which are becoming increasingly rare (P. Williams, pers. comm. 2001).

Mosaic burning breaks up fuel loads and thus can help slow wildfire (Rose *et al.* 1999, McCaw 2000).

To some extent, mosaics occur naturally, even in extreme wildfires. Turner *et al.* (1994) found great variability in intensity of burn in Yellowstone National Park forests after fires of unprecedented ferocity in 1988, as did Smith and Woodgate (1985) in Victorian *Eucalyptus regnans* forests after the 1983 Ash Wednesday fires. On a smaller scale, fire intensity in woodlands has been found to differ markedly between locations, species and types of litter, and is much greater under plants than on open ground (Hobbs and Atkins 1988). Some of this variability may be due to conditions during the burn, particularly wind speed and direction, and relative humidity. However, much is the result of features of the landscape itself such as creeklines and boulder outcrops, vegetation type, and slope (Turner *et al.* 1994, Gill and Bradstock 1995, Whelan 1995).

When undertaking a planned burn, there is usually room to incorporate a degree of patchiness. For example, several point ignition sources will lead to more patchiness than setting a line of fire. Mop-up operations can concentrate on safety issues but leave unburnt patches that do not pose a threat. Weather conditions and time of day can be used to vary fire intensity and spread. Experienced hands-on fire managers are able to achieve a mosaic effect even when burning a large area (N. Gourley, pers. comm. 2000).

In terms of extent of burn, there are suggestions in the literature that burning small isolated patches may bring its own problems. Bradstock *et al.* (1996), in a modelling study using species characteristics of an obligate seeder *Banksia*, found that extinction was least likely when average fire size was large. Predators may concentrate their grazing in small burnt areas, where succulent shoots abound, reducing seedling recruitment of the most palatable species (Keith 1996). Leigh and Holgate (1979) found that grazing by native animals after fire had a major effect on the survival of both resprouting plants and seedlings, and speculate that larger burned plots may be more able to support the local herbivore population, giving some individual seedlings a chance to develop to a size where they can survive.

Overall, it may be best both ecologically and practically to aim for a broad mosaic across the landscape of vegetation in different stages of post-fire regeneration, with small scale patchiness within the broader mosaic to provide animal refugia, seed sources for recolonisation, and options for managing threatened species populations. Planning tools to assist in the practical application of this concept are being developed. Rose *et al.* (1999) use a GIS-based planning system which supports the development of spatial mosaics at different scales, taking into account biodiversity, property protection and fire behaviour factors. Richards *et al.* (1999) describe a decision support model for maintaining a range of post-fire age-classes in a large, uniform reserve.

13. Season of burn

Variability in season of burn is suggested, although there is some evidence that elements of south-east Queensland's vegetation may be adapted to fire in autumn/winter.

There does not appear to be a definitive ecological argument for fire in any particular season in south-east Queensland, although a range of opinion exists.

Wildfire is most likely to occur in spring (September to November), or occasionally in early summer if storm rains arrive later than usual. Fuel which has dried out over the low rainfall months of winter is ripe for burning, westerly winds blow, and storms cause lightning strikes (Just 1978). Some argue that spring is therefore the 'natural' time for burning, while others point out that this is *not* the time to set planned fires, as they are more likely to escape and cause problems to life and property. Graziers tend to burn in spring to take advantage of the rapid post-fire grass regrowth that occurs with early summer rains. Others also advocate burning when soil moisture is high, in order to maximise post-fire regrowth and minimise damage to plant root systems, soil-stored seeds, soil invertebrates and micro-organisms. Appropriate soil moisture conditions can, however, be achieved in seasons other than spring (R. Melzer, A. Thomas, pers. comm. 2001). Spring fires in the Townsville area promote greater recruitment of grasses and legumes than autumn fires (P. Williams, pers. comm. 2001).

Biological arguments against spring burning include the likely loss of a year's seed crop from shrubs that have flowered in late winter and early spring, although this should not be an issue if sufficient time has elapsed since the last fire to allow seed banks to develop. Spring burning may also be problematic for young birds and other animals not yet able to move around sufficiently to get out of the path of the flames. Some dasyurid species may be particularly susceptible at this time as males have died and population numbers are low (Wilson 1996). However appropriate mosaic burning could be used to provide refuges and sources for recolonisation.

There is some support from local research for autumn and winter burns, particularly in grassy woodlands. A detailed review of explorers' records for Queensland suggests most aboriginal burning took place at that time of year (Fensham 1997) – although similar research by McLoughlin (1998) suggests that in the Sydney region where the fire season is only slightly different to that in SEQ, aborigines mostly burnt in the spring. Odgers (1999), who studied grasses on Mt Coot-tha in Brisbane, found most native species flowered in summer, with the maximum number of soil-stored germinable seeds for 11 out of 14 native species occurring in autumn or winter. Most native species (though not *Themeda triandra*) had transient seedbanks, suggesting adaptation to disturbance at this time of the year. McFarland (2000) recommends winter burning to maximise Ground Parrot survival in the heathlands where it occurs, as this avoids both the nesting period in spring and summer, and the autumn dispersal of juveniles.

On the other hand, winter burning may be detrimental to invertebrates, which tend to be dormant in winter (D. Sands, pers. comm. 2000). Trees and shrubs may also be at risk due to low sap flow (R. Melzer, pers. comm. 2001).

For some purposes, season of burn may be a means to a particular end. For example storm burning in late summer has been used to create relatively intense burns to discourage lantana in Brisbane Forest Park (N. Gourley, D. Kington, pers. comm. 2000).

There are also reasons to believe that season may either have little effect on biodiversity in south-east Queensland, or that variability in season of burn may be the most desirable approach. McFarland (1990) found no differences in plant reproductive activity in Cooloola heath by the second spring after autumn and spring burns. Even in more seasonal climates, ecological research has not identified clear preferences. Clark (1988) found some Hawkesbury sandstone species benefited from spring burns, while some preferred winter fire. A study of regeneration in five West Australian *Banksia* species found that although autumn burning produced more than twice the density of seedlings as did spring burning at the end of the first winter, differential mortality over the subsequent summer equalized the seedling-to-parent ratio (Enright and Lamont 1989). The main message from a study of the effects of spring and autumn burns in Victorian dry sclerophyll forest was one of resilience. Plant species differed in their strategies for maintaining a place in the community in the face of fire, but all had strategies which functioned effectively in either season, at least after a single fire (Tolhurst 1996).

The principle of variability probably has its place in relation to season of burn, as it does with other elements of the fire regime. Variability provides an opportunity for species that seed at different seasons, or which have varied seasonal vulnerabilities, to co-exist in the long-term. A mix of late summer, autumn and winter planned burning, together with some spring wildfire, would provide this variability.

14. Habitat protection during a burn

Leave escape routes for animals during burns. Where feasible, reduce fuel around key habitat features such as trees with hollows and logs. Minimise habitat fragmentation to ensure sources for recolonisation.

Where possible, planned burns should avoid creating a 'ring of fire' which will trap animals and increase the likelihood that they will be killed. Breaks in fire lines may provide avenues of escape for mobile fauna species (D. McFarland, pers. comm. 2001).

Fires both destroy, and over time help create, habitat features such as hollows and logs. Unfortunately, these features are in much shorter supply than they once were, due to clearing, logging, and gathering of firewood (Lindenmayer *et al.* 1997). Lost tree hollows take time to replace: a study in northern NSW found hollows appropriate for larger gliders and possums did not form in blackbutt trees until they reached at least 144 years of age (Mackowski 1984). Bushcare and other community groups may be able to assist in raking fuel away from trees and logs prior to a burn (T. Fensom, pers. comm. 2001). This approach will maximise the life of hollows in old trees, thus helping bridge the gap until younger trees reach hollow-bearing age.

The importance of unburnt patches in providing refugia for animals during a burn has already been discussed, as has the natural patchiness that occurs during fires (Guideline 12). The ability of fauna species to recolonise burnt patches may, however, be compromised by habitat fragmentation through clearing or other forms of disturbance. Fire in a small remnant may be the final blow that leads to local extinction, if avenues for recolonisation are not available. This problem is one of fragmentation, not fire *per se*. Here is yet another argument for the retention of large intact and diverse areas of bushland.

C. Guidelines for broad vegetation types

15. Rainforests and vine scrubs

Rainforests and vine scrubs are not fire adapted, and fire should generally be excluded from these areas.

Although there is evidence that some rainforest species can make an impressive recovery from the effects of a single fire (Unwin *et al.* 1985, Williams 2000), these mesic vegetation types are not fire adapted, and thus where possible fire should be excluded from these areas.

Fully developed rainforest in south-east Queensland is generally fire resistant, even in extreme fire weather (M. Hall, pers. comm. 2000). Vine scrubs, however, will burn if conditions are sufficiently dry, particularly where weeds such as green panic and lantana have increased their flammability. This also applies to rainforest which has been degraded through logging and weed invasion, where a vicious cycle of increased weediness and accompanying increased flammability may set in after an initial fire (House *et al.* 1998). It follows that one way to protect this vegetation type from fire is to minimise disturbance and weediness. For this reason, and because the ecotone between rainforest and drier forest types may house significant species, fire trails along scrub margins should only be used as a protection measure where other options are not available. Planning burning in ecologically appropriate nearby areas is one such option – for example relatively frequent burning on grassy ridgetops can help stop wildfires penetrating into rainforest gullies (D. Kington, pers. comm. 2000).

16. Wet sclerophyll forests

a. High intensity fire catalyses regeneration in eucalypt-dominated wet sclerophyll forests. Wildfire in its natural season should adequately fulfil this role in most cases. Intervals are likely to range between 20 and 100+ years.

b. Less intense understorey fires may also play a role in some tall eucalypt forests (eg those with an understorey of sclerophyllous shrubs), along with occasional 'stand replacing' wildfires. Research to determine where and when this may be the case is urgently needed. Intervals below 6 years for grassy systems, and 12 years for shrubby systems, are not recommended.

c. *Lophostemon confertus* (brush box) dominated forests are probably adapted to very long interfire intervals, and may not need fire at all.

Wet sclerophyll forest in SEQ encompasses a range of tall eucalypt and other myrtaceous overstorey species. Its understorey can be grassy (Taylor 1992), it can consist of sclerophyllous shrubs (Guinto *et al.* 1999), or it can be made up of rainforest species. It shelters a variety of bird and animal species, including eucalypt-dependent flying foxes and gliders (M. Fingland, pers. comm. 2000).

Of all the broad vegetation types in SEQ, wet sclerophyll forest poses the most intriguing questions and dilemmas in terms of appropriate fire regimes. The above guidelines are particularly tentative: the literature on this topic is explored in some detail, in order to give readers the opportunity to draw their own conclusions. This vegetation type is a top priority for fire ecology research in south-east Queensland (Tran and Wild 2000).

Wet sclerophyll forests occur in a discontinuous distribution along mainland Australia's east coast, in Victoria, Tasmania and south-west Western Australia (Ashton 1981). Research has primarily focused on the Victorian montane ash forests, and the *E. grandis* forests in the North Queensland highlands, although some studies from SEQ and northern NSW are available.

The Victorian wet sclerophyll forests feature ash eucalypt species, the foremost of which is the majestic mountain ash *E. regnans.* Eucalypt species and the dominant understorey species in these forests regenerate through seedling establishment after severe fire in which adults are killed (Ashton 1976), leading to even-aged forest stands. These forests thus demonstrate the classic obligate seeder strategy, and are vulnerable to loss of overstorey species if severe fire occurs before seedlings reach reproductive maturity at about 20 years, or if severe fire does *not* occur before even-aged stands senesce and die at 350-500 years (McCarthy *et al.* 1999). Although rainforest may occur in gullies adjacent to Victorian wet sclerophyll forests (Melick 1990), rainforest 'takeover' is not seen as a major threat unless fire-free intervals stretch to several hundred years (Ashton 1981, Attiwill 1994), an unlikely occurrence in an area which boasts some of the most severe fire weather in the world (Cheney 1994). McCarthy *et al.* (1999) estimate that interfire intervals over the past few centuries have averaged between 75 and 150 years for intense "tree killing" fires, and between 37 and 75 years for all fires. These intervals have retained the wet sclerophyll forest, although in a form which will not provide sufficient "old growth" trees for hollow-dependent fauna (Lindenmayer *et al.* 1997). Pre-European fire regimes would probably have featured longer intervals between tree-killing fires.

The North Queensland wet sclerophyll forests occur in a narrow strip between rainforest and woodland, generally on the western slopes of the Great Dividing Range. Stands are either dominated by the flooded gum *E. grandis*, or feature a mixture of species including *E. grandis*, *E. resinifera*, *Corymbia intermedia* and *Syncarpia glomulifera*. The understorey is generally grassy (Harrington and Sanderson 1994). Rainforest invasion of these forests is occurring at a rapid rate (Unwin 1989, Harrington and Sanderson 1994), to the point where a total loss of the *E. grandis* forests, and a severe reduction or total loss of the mixed eucalypt forests, is predicted within the next century (Harrington and Sanderson 1994). This loss is of concern, as a number of animal and bird species are at risk of local extinction if the wet sclerophyll habitat disappears (Chapman and Harrington 1997). However while there is widespread agreement that the problem is one of changed fire regimes, the nature of regimes which will conserve these forests is not clear. Low intensity fire, even at fairly short intervals, doesn't seem to be effective. Many of these forests *are* regularly burned, for grazing green pick, however this has not halted the rainforest advance. Williams (2000) found invading rainforest species regenerated well through both resprouting and seedling establishment after low to moderate intensity planned burns at two NQ wet sclerophyll sites, confirming previous findings (Unwin *et al.* 1985). It also seems unlikely that severe fire at the lengthy intervals found in the Victorian wet sclerophyll forests would halt NQ rainforest expansion, given the rapid rate of takeover. It has been suggested that fires that occur today may be of lower intensity, and thus less able to penetrate to the rainforest edge, than those when the forests were under Aboriginal management. This may be due to reduction in grass cover through grazing (Unwin 1989), and to a shift from late to early dry season burning (Harrington and Sanderson 1994). It has also been suggested that Aboriginal burning may have maintained the wet eucalypt forests from the days of the last glacial period, through firing at least every 10-20 years (Ash 1988). If this is the case, perhaps a threshold has been passed which will not allow a reversion to sclerophyll forest in some areas, no matter what fire regime is applied.

Lying between the cool temperate and the tropical wet sclerophyll forests are those of south-east Queensland. SEQ wet sclerophyll forests are much more extensive than those in NQ, covering many hectares on a range of substrates, particularly on slopes in upland areas. Key species include *E. pilularis* (blackbutt), *E. grandis, E. microcorys, E. acmenoides, Lophostemon confertus* and *Syncarpia glomulifera*, ie there is some overlap in terms of species with NQ, but none with Victoria. In SEQ the various wet sclerophyll species form communities which intergrade with each other and with rainforest and dry sclerophyll along gradients determined by substrate, nutrient levels, topography and rainfall, as well as fire – in fact, these factors often interact (Florence 1996).

The work from NQ and Victoria tells us that fire in wet sclerophyll forests may be vital for two reasons: to prevent rainforest take-over, and to allow regeneration. It also gives us some clues as to regimes suitable for meeting these objectives. To what extent are these objectives, and regimes, relevant in SEQ?

Concern has been expressed that SEQ wet sclerophyll forests may be lost to rainforest invasion in the absence of an appropriate fire regime. There is some support in the literature for this contention. Herbert (1936) describes the advance of young *Northofagus moorei* (Antarctic beech) into eucalypt forest in the Macpherson Range in SEQ. Smith and Guyer (1983) clearly demonstrate the replacement of *E. saligna* and *E. microcorys* tall eucalypt forest by rainforest east of Tenterfield in north-eastern NSW. Both papers contend that fire defines the rainforest boundary. Anecdotal accounts of rainforest invasion in SEQ also abound. For instance, by the early 1990s many specimens of the rare cycad *Macrozamia peroffskiana* in Springbrook National Park appeared to be dying due to colonisation by rainforest crowsnest ferns and vines. A planned burn in 1993 successfully eliminated the ferns and vines, adult cycads regenerated, and cycad seedlings established. Eight years later, the cycads are still healthy, however some rainforest pioneer species have established. Parks staff now believe they should have aimed for a more intense burn (M. Hall, S. Millington, pers. comm. 2000). There are, however, no SEQ studies equivalent to those in NQ which document the extent or speed of rainforest incursion. Perhaps it is an issue in some areas but not others. Perhaps incursions are temporary, and will be reversed by the next wildfire. These questions should be a primary focus of future research in SEQ.

Florence (1996), a forestry researcher of considerable experience in SEQ and elsewhere, believes that site factors other than fire will limit rainforest spread in south-east Queensland. He explicitly rejects the Clementian doctrine which regards rainforests as the "climax vegetation" of the east coast, and critiques the concept of wet sclerophyll forest as "transitional", arguing for a community view of the association of eucalypts with a rainforest understorey. Florence (1996) suggests that a mature and healthy wet sclerophyll overstorey may “control” the extent to which the rainforest understorey is able to expand. Under this scenario, rainforest expansion might be more a function of logging in wet sclerophyll forests, than of changed fire regimes. Perhaps a balance between wet sclerophyll and rainforest exists in some communities within this broad vegetation type, but not in others.

The issue of rainforest ‘invasion’ also raises values issues. Apart from the obvious question as to whether rainforest or tall eucalypt forest is preferable, it has also been pointed out that these ‘transitional’ areas shelter a number of species whose habitat has been lost due to clearing (W.J. McDonald, pers. comm. 2000).

What about the role of fire in regenerating SEQ’s wet sclerophyll species? The majority of eucalypts in the SEQ tall open forests are lignotuberous (Ashton 1981), indicating an ability of most adults to survive wildfire, and to create a pool of fire-tolerant suppressed seedlings which can respond to canopy gaps when they occur (Florence 1996). Fire may assist these species’ regeneration through the creation of gaps and an ash bed, however it may also generate overwhelming competition for young eucalypts, due to massive germination and growth of understorey species such as *Dodonaea triquetra, Acacia irrorata, Kennedia rubicunda* and other leguminous shrubs (Floyd 1966). Fire may be of more assistance to SEQ’s two major non-lignotuberous wet sclerophyll eucalypts, *E. pilularis* and *E. grandis*. *Eucalyptus pilularis* has the ability to regenerate from epicormic shoots, and can exhibit “wheatfield germination” reminiscent of that in the Victorian mountain ash forests after fire (Florence 1964). Experimental work has demonstrated severe inhibition of *E. pilularis* seedlings in undisturbed soil from mature blackbutt forest (Florence 1996), but it has also shown, in a field experiment in the Sunshine Coast region, that this inhibition is not necessarily broken by planned fire (Florence 1964). *Eucalyptus grandis*, a fire-sensitive gum-barked tree, does not regenerate well in the presence of overstorey competition. Its presence in mixed eucalypt stands in SEQ argues for the existence of fires of sufficient intensity to create canopy gaps through killing adult trees. However, too-frequent fire will eliminate this species, as young trees will be lost before they reach reproductive maturity (Florence 1996). Abundant post-fire eucalypt seedling regeneration has been observed in ashbeds left by burnt logs in SEQ and nearby areas (B. Trembath, W.J. McDonald, pers. comm. 2001), again suggesting the importance of intense fire.

It is possible that, for the long-lived eucalypt species of SEQ’s wet sclerophyll forests, it is not vital that perfect regeneration conditions occur with every fire. Some fires may favour particular species, with fires at a variety of seasons, intervals, and intensities providing conditions suitable for sufficient recruitment over a period of 100 or more years. The result may be a range of tree ages, with small patches of regrowth scattered through mature forest. Patchiness within fires may add to this effect (Florence 1996).

What fire frequency and intensity may hold SEQ rainforest at bay, and allow fire sensitive species to regenerate? Given the NQ findings on the tolerance of rainforest pioneers toward low and moderate intensity burns, it seems reasonable to assume that high intensity fires will be needed in SEQ to keep rainforest from taking over in wet sclerophyll areas. Fire will need to be sufficiently frequent to prevent the rainforest canopy from reaching a density which will exclude fire, but sufficiently infrequent to allow fire-sensitive species to reach reproductive maturity. Assuming that the rate of rainforest invasion in SEQ is considerably lower than that in NQ, perhaps a frequency of 20-50 years might be appropriate to arrest rainforest incursion here. A study in *E. pilularis/E. grandis* forest near Coffs Harbour in northern NSW found less seed of two common rainforest pioneer species in a site which had had two fires at approximately 15 year intervals, than in a matched site which had burned after a 30-year interfire interval. Note that fire was still able to penetrate the site after 30 years (Floyd 1976). Russell and Roberts (1996) found no rainforest invasion in a blackbutt/Sydney blue gum forest near Toowoomba after 14 years without fire, although lantana was becoming problematic. A minimum interval of 20 years should allow the fast-growing young *E. grandis* saplings to reach reproductive maturity. A minimum interfire interval of this order would also be appropriate for another fire sensitive wet sclerophyll species, *Eucalyptus oreades*, the Blue Mountains Ash (Glasby *et al.* 1988), which is found in the Scenic Rim area. Scarring on *Xanthorrhoea* trunks suggests *E. oreades* stands in Springbrook National Park have burned at approximately 50 year intervals over the past 200 years (C. Sandercoe per. comm. 2001).

The role of fire in the creation and destruction of tree hollows should also be considered in discussions of appropriate fire frequency in wet sclerophyll forests. We have noted that hollows do not form in blackbutt trees for well over 100 years (Mackowski 1984), and that these trees are more easily killed by intense fire than their lignotuberous counterparts. Some interfire intervals above 50 years may be desirable for retention of hollow-bearing trees in SEQ, although the less fire sensitive nature of SEQ wet sclerophyll eucalypts may mean that fire does not destroy hollows to the same extent here as in Victorian wet sclerophyll forests. Research to clarify this question is needed.

A carbon-dating study from northern NSW suggests that SEQ's wet sclerophyll forests may be adapted to very long interfire intervals indeed, at least in terms of "stand replacing" fires. Turner (1984) counted the number of layers of charcoal in the soil of three vegetation types: gully rainforest, adjacent *Lophostemon confertus*-dominated wet sclerophyll forest, and blackbutt forest on the slopes above the brush box. One layer, which was dated to 1110 years BP, was found in the rainforest. Eight layers were consistently found in the brush-box, giving an average interfire interval of 325-380 years. The twelve layers found in the blackbutt forest were less distinct, but assuming each layer represents a single fire, the average interfire interval in this vegetation type was 280 years! These three vegetation types have apparently co-existed for over 1000 years on this fire regime. It is not known whether the fires detected in this study were interspersed with fires of lower intensity which did not show up in the charcoal record.

It is likely that different communities within the broad wet sclerophyll forest vegetation type are adapted to somewhat different fire regimes. While brush box dominated forest will burn and recover, this species can regenerate in the absence of fire (Guinto *et al.* 1999). Perhaps fire is not a necessary component of this ecosystem, even if it can be tolerated. Tall eucalypt forests with a sclerophyll or grassy understorey may be adapted to shorter interfire intervals than the wetter forests. Less intense fires may play a role in understorey regeneration in these forests, along with infrequent "stand replacing" fires. Patches where the understorey is younger than the overstorey due to the effects of moderate intensity fire can be found in Victorian ash forests (Ashton 1981, Ashton and Martin 1996), and Harrington (1995) speculates that both planned burns and occasional intense wildfire may play a role in maintaining NQ's tall open forests.

However even those wet sclerophyll forests without a rainforest understorey should not be subject to very short interfire intervals. Two SEQ studies add weight to this conclusion. Both utilised experimental sites maintained by DPI Forestry in Peachester State Forest near the Glasshouse Mountains. Three treatments in this blackbutt mixed forest have been maintained since 1972: burning every two years, burning every four years, or no burning. Eucalypt tree recruitment was negligible on all three treatment sites, although the more mesophytic *Syncarpia glomulifera* and *Lophostemon confertus* did recruit quite extensively on the unburnt plots (Guinto *et al.* 1999). Presumably the relatively low-intensity fires in the burnt plots were not hot enough to create the light and soil conditions necessary for establishment, and/or killed off any young eucalypt seedlings, while lack of disturbance prevented eucalypt establishment on the unburnt plots. The second study, which looked at nutrient losses and gains under the various regimes, found a 40% reduction in topsoil organic carbon and total nitrogen levels in plots burned on the two-year cycle, and a 10% reduction on the four-year cycle (Guinto *et al.* 1998).

Overall, we still have much to understand in relation to this forest type. Further research in SEQ may well lead to modification of the regimes suggested above.

17. Dry sclerophyll forests and woodlands

Appropriate fire frequency for dry sclerophyll forests and woodlands depends on the nature of the understorey. Fires at a range of intervals between 3 and 6 years are suggested for areas which support a grassy understorey. Fires at a range of intervals between 7 and 25 years are suggested for areas which support a shrubby understorey. The variable fire intensity and season which is likely to result from a mix of planned burns and wildfire should be appropriate for maintaining biodiversity in this forest type.

Any discussion of appropriate fire regimes for dry sclerophyll forests and woodlands must consider the nature of, and the management aims for, the understorey.

The ecosystems within this broad vegetation type vary in their degree of tree cover, and in the extent to which shrubs or grasses dominate the understorey. Some areas are open, with a grassy understorey often dominated by kangaroo grass (*Themeda triandra*). Others have a shrubby understorey. To some extent, the distribution of these understorey types will be a function of soil nutrient status, with more nutrient-rich sites tending to support a grassy understorey, and poorer sites tending to support shrubs (Christensen *et al.* 1981). However fire frequency also influences the degree to which shrubs or grasses dominate a site, with frequent burning tending to favour grasses over shrubs, and vice versa (Bradfield 1981, Birk and Bridges 1989, Lamb *et al.* 1992, House 1995).

As with wet sclerophyll/rainforest, there can be debate over what constitutes a 'natural', or desirable, state for dry eucalypt ecosystems. On one hand, there is strong evidence that some places in Australia which supported open grassy woodland under Aboriginal management have 'thickened' in terms of tree and/or shrub cover since European settlement. Many authors (eg Clark and McLoughlin 1986, Pyne 1991, Flannery 1994) quote the writings of European explorers, which repeatedly mention open woodland with a rich grassy understorey. Lunt (1998a,b) has documented, and decried, the change from open grassy woodland to increasingly impenetrable *Allocasuarina littoralis* forest in the absence of burning on the Bellarine Peninsula in Victoria. In south-east Queensland, Kington (1997) has noted the increased abundance of shrubs and *Lophostemon confertus* in once-grassy woodlands in Brisbane Forest Park in the absence of frequent burning, and the detrimental effect on some macropods, particularly Pretty-face Wallabies. In the Bunya Mountains, the area occupied by the grassy "balds" shrunk by 26% in the forty years to 1991, although not in one frequently-burned area. Exhaustive investigation was unable to uncover any explanation for the maintenance of the balds other than Aboriginal burning (Fensham and Fairfax 1996). On the other hand, there are convincing arguments to support the contention that some dry sclerophyll forests and woodlands were shrubby prior to European settlement, and that Aboriginal people did not burn these areas so frequently. The presence of shrubs which need a relatively long fire-free interval to maintain their presence in the community is one such argument (Clark and McLoughlin 1986, Benson & Redpath 1997), as is the existence of bird and animal species which need long-unburnt habitat (Fox 1983, Catling 1991, Woinarski 1999). Clark and McLoughlin (1986) argue that Aboriginal fire management around Sydney would have varied according to vegetation type, with frequent burning on the shale ridges, and less frequent burning on the sandstone hillsides. The fact that current Aboriginal fire management practices in areas such as Kakadu and Uluru involve different approaches for different vegetation types (Lewis 1989, Breeden 1994) supports this contention.

If we wish to preserve south-east Queensland's biodiversity, we need to maintain both grassy and shrubby dry sclerophyll forests and woodlands within our landscapes. This means a diversity of fire regimes, applied with due regard to the likely 'natural' understorey, given substrate considerations and what is known of past Aboriginal management in local areas.

The following discussion draws heavily on work from outside south-east Queensland, as few studies have been conducted locally.

Grassy forests and woodlands

Active management of grassy woodlands is essential. These ecosystems have suffered disproportionately from the effects of European settlement. They were generally the first areas to be cleared or grazed, with the result that those remnants which remain tend to be degraded (Lunt 1991, Prober and Thiele 1993, Yates and Hobbs 1997). Changed fire regimes have also threatened these ecosystems, which are adapted to pulse disturbance at relatively frequent intervals (Jones 2001). The understorey consists of a matrix of a limited number of tussock grass species, amongst which grow forbs and other grasses. It is this interstitial component which gives grassy ecosystems their diversity. Its maintenance depends on disturbance, which reduces the stature of the dominant grasses, removes litter, and creates bare space in which the forbs and small grasses can germinate and grow (Tremont and McIntyre 1994). Recent work by Morgan (1998) in productive *Themeda*-dominated grasslands in Victoria found gaps suitable for seedling recruitment had disappeared by three years after fire. Morgan recommends burning these grasslands at intervals of one to three years, in order to ensure the gaps needed for recruitment. He also contends that these intervals will enhance the likelihood that recruitment opportunities will occur when native grass seeds, which are more likely to be present in early than later post-fire years, will encounter suitable climatic conditions.

Other evidence that very frequent burning enhances biodiversity in *Themeda*-dominated grasslands comes from a classic study by Stuwe and Parsons (1977), again from basalt country in Victoria. A comparison of three management regimes found that the patchy annual burning undertaken on railway reserves was associated with a higher species richness of native plants than was grazing or fire exclusion – although this study has some limitations due to confounding of time-since-fire and fire frequency.

How relevant are these Victorian recommendations for south-east Queensland's grassy ecosystems? Our grassy woodlands may not be as productive as those of Victoria's basalt plains – if this is so, then tussock-grass gaps may not close as quickly. Very frequent burning, eg at yearly intervals, is said to favour blady grass at the expense of kangaroo and other native grasses in SEQ. Also, these Victorian studies focus on grasslands, rather than woodlands, so eucalypt recruitment is not an issue. Burning in grassy woodlands needs to be sufficiently frequent to prevent large-scale recruitment of trees and shrubs, but not so severe as to altogether prevent seedling establishment and recruitment of eucalypts into the canopy. The resprouting eucalypts found in dry sclerophyll forests tend to maintain a bank of suppressed saplings which resprout after fire (Florence 1996), but do not necessarily enter the canopy at that time (P. Williams, pers. comm. 2001). Work on fire regimes for savanna woodlands in the Northern Territory has recommended a fire frequency of three to five years, partly to ensure appropriate tree regeneration (Andersen 2000, Setterfield 2000). Some longer intervals would also provide habitat for birds and animals which need relatively dense understorey cover. The suggested frequency of three to six years seems reasonable when all these factors are taken into account. Research into the regeneration needs of all components of grassy woodland ecosystems in south-east Queensland, and into the habitat requirements of grass-dependent fauna species such the Pretty-face Wallaby, would help refine this estimate.

A word about weeds. Grassy ecosystems often contain exotic grasses. As well as their direct effect on biodiversity, these species can affect fire regimes through increased fuel loads. Unfortunately, the literature provides little hope that fire alone can provide an effective means of tipping the balance back towards native species. A post-fire study in a Victorian *Themeda* grassland found that exotic species thrived after fire (Lunt 1990). In a local study of grass species in "savanna open forest" in Mt Coot-tha reserve, Odgers (1999) found that most native species had transient seedbanks, while the seedbanks for co-occurring exotic grasses were persistent. Lunt (1990) recommends a search for integrated methods of weed control, with fire as one element in an arsenal which may include weeding, poisoning, and perhaps some manipulated grazing by native or introduced herbivores.

Shrubby forests and woodlands

In SEQ, diverse and colourful shrubby forests and woodlands occur in coastal sandy soils, as well as in low nutrient soils further inland, for example in the Helidon Hills and Toohey Forest. Attempts to create a grassy understorey through frequent burning in these ecosystems may lead to a “simple, degraded, species poor and erosion prone system” (A. Thomas, pers. comm. 2001).

The fire ecology of shrubby open forests and woodlands has been extensively studied in New South Wales, particularly, but not only, in the Hawksbury sandstone country around Sydney. Some work has taken place in and around south-east Queensland, notably at the DPI sites at Beerwah (which includes both heath and shrubby forest) and at Bauple near Gympie (Taylor 1992, Hannah *et al.* 1998), and in Girraween National Park near Stanthorpe (Watson 1999).

In Sydney, many shrub species found in the heathlands also occur in the shrubby forests and woodlands. In fact, research studies and attendant management recommendations from this area frequently cover heath and shrubby woodland together. There is strong evidence that a number of plant species in the Hawksbury sandstone vegetation need a fire-free interval in excess of 6-8 years to regenerate to the point where they can maintain their place in the community. Benson (1985), in a careful study of the primary juvenile period of Hawksbury sandstone shrubs, found some species did not flower until the eighth year after fire, while many species took from four to six years. We have already noted that it may take several more years until an obligate seeder species builds up sufficient seed reserves to successfully replace itself if another fire were to occur (Gill and Nicholls 1989). Resprouter species vulnerable to frequent fire have also been identified, with minimum intervals of up to 16 years recommended for population stability (Bradstock and Myerscough 1988, Bradstock 1990). Community-level studies and simulations have confirmed the risks of plant extinctions under frequent burning in this vegetation type. Nieuwenhuis (1987) found a number of obligate seeder species consistently missing from sites burned at less than five years intervals. Cary and Morrison (1995) found drastic effects in areas burnt every 1-3 years, while fires at 4-6 year intervals also caused changes. A simulation study which took account of the ameliorating effects of long-lived seedbanks and patchiness, still found high levels of extinction in obligate seeders at interfire intervals of less than six years (Bradstock *et al.* 1998). Interfire intervals of between 6-8 and 30 years have been recommended for this vegetation type in NSW (Bradstock *et al.* 1995).

Closer to south-east Queensland, a study by Fox and Fox (1986) in shrubby woodland near Myall Lakes in NSW found some obligate seeder species missing from sites burned twice in twelve years, while other species, mostly resprouters, were absent from sites which had had a twelve year interfire interval. Watson (1999) found a number of species reduced in abundance on sites in Girraween National Park which had burnt approximately every four years. A smaller number of species was disadvantaged by interfire intervals of 20-28 years. Variable intervals, with an emphasis on those over 10 years, were recommended.

What can these studies tell us about minimum interfire intervals for SEQ shrubby forests? Most of the obligate seeder species which have been found to be particularly vulnerable to frequent burning in the Hawkesbury sandstone are not found here, perhaps reflecting a history of shorter interfire intervals in SEQ than around Sydney. However, a number of vulnerable species identified at Myall Lakes and Girraween *are* found in SEQ. They include *Dillwynia retorta* and *Gompholobium virgatum* (Fox and Fox 1986), and *Lomatia silaifolia* and *Hakea dactyloides* (Watson 1999). These studies suggest at least a 7-8 year interval between fires. However they, as well as the studies from Sydney, come from somewhat cooler climates than SEQ. Here, shrub seedlings may mature more rapidly. The minimum interfire interval of seven years recommended for shrubby SEQ dry eucalypt forests and woodlands recognises the above similarities and differences. Studies of the length of juvenile periods of SEQ obligate seeder shrub species would be invaluable in refining this estimate.

Determining an appropriate maximum interfire interval value is more difficult. The figure needs to be sufficiently large to allow for variability in actual intervals, as the findings on the importance of variability relate particularly to this vegetation type. The need to ensure that areas of long-unburnt vegetation are available to provide habitat for those fauna species which need dense cover, hollows, litter, etc is also an important factor. Studies at Myall Lakes have found that later entrants in the small mammal succession in dry sclerophyll forests can take up to eight years to establish a thriving breeding population (Fox and McKay 1981, Fox 1983). Habitat complexity, a defined variable which correlates with the number of native small mammals captured in dry sclerophyll forest (Catling and Burt 1995) levels out at 8-10 years post-fire (Coops and Catling 2000). Catling (1991) argues strongly for maintaining a good proportion of vegetation in the later post-fire successional stages, and identifies 25 native mammal species in south-eastern Australia which will suffer if burning is too frequent. In northern NSW, long unburnt sites with a dense multi-layered shrubby understorey have been found to house a significantly higher abundance of small mammals than grazed sites burned approximately every four years (Tasker *et al.* 1999). Similar arguments and findings have been advanced in relation to bird species (Woinarski 1999). For example, Black-breasted Button-quail numbers in a forest near Gympie expanded when frequent burning ceased, allowing the build up of the deep litter layer that these birds need (Hughes and Hughes 1991). Reptiles, too, have been found to favour long-unburned sites. A study by Hannah *et al.* (1998) in the DPI plots at Bauple found abundance, species richness and species diversity were all highest in the unburnt treatment relative to plots burnt either annually, or every 2-5 years. The maximum interfire interval of 25 years recommended above will allow patches of dry shrubby forest with complex habitat structure to persist in the landscape to a reasonable extent.

At the same time, interfire intervals should not be so long as to lead to the loss of plant species dependent on fire for regeneration. The finding that some shrub species in open forests and woodlands at Girraween were significantly disadvantaged by a 28 year interfire interval (Watson 1999) suggests the SEQ maximum should not exceed 25 years.

18. Heathlands

a. In coastal treeless heath and *Banksia aemula* woodlands, fires at a range of intervals between 7 and 20 years, with an emphasis on intervals in the 8-12 year range, are recommended to maintain overall biodiversity. Planned burns in wet heaths should be conducted when the substrate is wet, to avoid the risk of peat fire.

b. Heathlands of rocky inland areas are probably adapted to a range of fire regimes depending on their relationship to the surrounding matrix. Intervals between 15 and 50 years are suggested.

Coastal and sub-coastal heath

We are fortunate in having a body of research that directly addresses the question of ecologically appropriate fire regimes in coastal heaths in or near the area covered by this report. Much of this research was carried out from 1983 to 1991 in the Cooloola area using chronosequences with sites differing in time since fire. Other studies have focussed on the DPI experimental plots at Beerwah. This concentrated work means that the recommendations for this vegetation type are well-founded, and draw only marginally on studies from outside the region.

As with the other broad vegetation types discussed here, coastal heathlands include a range of plant communities. Some heaths are treeless, and may be "wet" or "dry" depending on their relationship to the water table. Other areas have an overstorey dominated by *Banksia aemula*, the wallum banksia. These heathy woodland areas may be adapted to slightly longer interfire intervals than the treeless heath (C. Sandercoe, pers. comm. 2001). Wet heath, however, probably burns as frequently as treeless dry heath. These two vegetation types often adjoin, and reports indicate fires do not stop when they reach the wetter areas (eg Newsome *et al.* 1975, Fox 1983, Benwell 1998), which build up high fuel loads (Sandercoe 1989). Wet heaths can dry out after prolonged drought, and may be vulnerable to peat fires which can cause major shifts in species composition (Brown and Podger 1982). For this reason, planned burns in wet heath should take place when the substrate is saturated (A. Thomas, P. Donatiu, pers. comm. 2001).

The percentage of obligate seeder species in coastal heath is of the order of 24-40% (Sandercoe 1989). These species are at risk from too frequent fire, although they vary in time to reproductive maturity. In *Banksia aemula* heathy woodlands at Mt Bilewilam in Cooloola, nine obligate seeder species did not produce mature fruits by 3 years post-fire (Taylor 1992). At least one obligate seeder species, *Hakea actites*, has been lost from the Beerwah experimental plot burned every three years, and populations in the five-year burn plot fluctuate widely (Taylor 1992, Sandercoe 1989). Harrold (1979) found that while most obligate seeder species were fruiting by five years after a fire in *Banksia aemula* woodland near Noosa, *Persoonia virgata* had not yet produced mature fruits, while *Allocasuarina littoralis* and *Ricinocarpos pinifolius* had only a small number of fruits per plant. It thus appears that six years or more should be allowed to elapse between fires if all obligate seeder species are to be retained in this community.

Several other findings in relation to flora suggest a minimum interfire interval of six to eight years. Seedlings of eight resprouting species in the Mt Bilewilam study died if burnt within 36 months of germination (Taylor 1992). In coastal treeless heath reproductive effort, in terms of both number of species seeding and the standing crop of seeds, peaks at 4-8 years post-fire (McFarland 1990). Many plant structural characteristics in this community, including vegetative cover, plant height and vertical foliage density peak at six to seven years post-fire, then plateau or decline (McFarland 1988a). McFarland (1988a:543) speculates that "In Queensland, where there is potential for growth during more months of the year… maximum biomass may be reached more quickly. Consequently, growth may cease sooner and degeneration begin earlier in subtropical heathlands compared with those in the south." Thus fires in SEQ coastal heathlands may be on a shorter average cycle than those 1000km to the south, where longer interfire intervals are often recommended (eg Bradstock and Myerscough 1988, Bradstock *et al.* 1995).

The Cooloola work included bird usage of different-aged heaths. Initial study of the vulnerable Ground Parrot found bird density peaked at 5-8 years after fire (McFarland 1992), while nests were fairly evenly distributed in heaths 4 to13 years old (McFarland 2000). (Resurvey work in 2001 found Ground Parrots still inhabiting heath 20-30 years after fire, albeit at low density - D. McFarland, pers comm. 2001). Other bird species varied in their use of the post-fire habitat - some preferred recently burnt and others long-unburnt heathlands. For example, post-fire flowering of the grasstrees attracted lorikeets and large honeyeaters, while in the older heaths White-cheeked Honeyeaters and Yellow-tailed Black Cockatoos used maturing *Hakea* shrubs for nest sites and food respectively (McFarland 1992, 2000). The number of breeding residents peaked at 3-8 years post-fire (McFarland 2000). Here we have an argument for both variety in interfire intervals, within limits, and variety in post-fire age classes across the landscape. This should be possible at a landscape level, however the continuous fuel loads in this vegetation type make it a highly skilled task to obtain patchiness within a single burn (McFarland 2000, C. Sandercoe, pers. comm. 2001).

The fact that it is the relatively young heaths – those between 3 and 8 years post-fire – which play the greatest role in providing breeding habitat for birds (McFarland 1992) fits with findings in relation to peaks in plant reproductive effort (4-8 years post fire) and structural characteristics (6 –7 years post-fire)(McFarland 1988a, 1990). It is in order to ensure a good supply of this very productive habitat in the heathland landscape that an emphasis on intervals in the 8-12 year range is recommended.

However some longer interfire intervals are obviously important to those bird species which favour long-unburnt habitat. In addition, longer intervals allow obligate seeder plant species which take a long time to mature and build up a seedbank, to reach the point where they can easily replace or increase their population post-fire. What then might be an appropriate maximum interfire interval?

The published Cooloola work did not extend to sites over 12 years in age. The Beerwah experiment, however, includes a plot unburnt since 1972. This is dominated by large and prickly *Hakea actites*. McFarland (2000) notes that these plots have changed from heath to shrubland, and that Ground Parrots and many other bird species will not use this habitat. While not all heathlands in SEQ will be dominated by *Hakea actites* in the absence of fire, plant biodiversity may also be at risk if fire is excluded for long periods. Grasses and forbs, which flourish in the first couple of years after a fire, decrease sharply in abundance or disappear altogether from aging heaths. They remain in the soil as seeds, bulbs or corms (McFarland 1988a, 2000), however there must be limits to their ability to survive in these states. Further south, long-unburnt heath can experience possibly irreversible changes (Burrell 1981, cited in Attiwell 1994; Brown and Podger 1982). The 20 year maximum interfire interval recommended above takes these factors into account, while also allowing for the needs of plants and animals that benefit from long-unburnt habitat. Further research into the effects of long interfire intervals will assist in refining this figure.

Montane heath

In some parts of SEQ, heath is also found on rocky mountain summits and terraces. Rocky outcrops often show high landscape-generated species diversity (Hopper *et al.* 1997), and host endemic, rare and threatened species (Hunter and Clarke 1998). Some species appear only after a fire (Hunter 1995,1998a; Hunter *et al.* 1998). For example, Hunter (1995) found the rare obligate seeders *Muehlenbeckia costata* and *Acacia latisepala* in large numbers on granite outcrops in Girraween and Bald Rock National Parks on the NSW/Queensland border, after an intense fire in 1994. There are indications that the flora of rocky heaths tends to contain a higher proportion of obligate seeder species than the surrounding matrix (Hunter *et al.* 1998), suggesting that these communities are both fire sensitive and fire dependent.

Hunter (1998b) cautions against assuming, however, that these vegetation communities will benefit from frequent fire. He points out that only some fires from the surrounding matrix are sufficiently intense to move into the rocky habitats, citing his personal observation that even in the widespread fires of 1994 at Girraween and Warra State Forest, up to half the outcrop communities were unburnt (Hunter *et al.* 1998). In these inland mountain communities, Hunter believes that interfire intervals could be as long as 100 years (J. Hunter, pers. comm. 1998).

In SEQ, montane heaths vary greatly in floristic composition (W. Drake, pers. comm. 2001), reflecting their relationship to local weather patterns, substrates and the surrounding vegetation as well as the tendency for ‘islands’ to develop distinctive species complements. Appropriate fire regimes are also likely to vary, and to be linked to the degree of continuity of each community with its surrounding matrix. Research into the vital attributes of *Callitris monticola* and *Pultenaea pycnocephala*, two obligate seeder species found in heath and related vegetation types in Lamington National Park, found that mature fruits were first produced about eight years after fire (Flenady 2000), suggesting these species need relatively but not excessively long interfire intervals. Wildfires in this area seem to occur at about 20 year intervals (C. Sandercoe, pers. comm. 2001). The 15-50 year range suggested for this vegetation type in SEQ takes into account the differences in climate between here and New England, and the fact that some communities will experience more frequent fire than others depending on their proximity to the surrounding matrix. Further research into the vital characteristics of montane heath obligate seeder species would help refine estimates for different communities.

19. *Melaleuca quinquenervia* wetlands

Fires at a range of intervals between 15 and 30 years are suggested for this vegetation type. Planned burns should be conducted when the substrate is wet, to avoid the risk of peat fire.

In SEQ *Melaleuca quinquenervia*-dominated communities adjoin heaths and dry eucalypt forests and woodlands. Their capacity to burn along with these neighbours is readily apparent from blackened trunks.

Once again there is variability in understorey floristics within this broad vegetation type. Wetter sites support sedges, while drier sites have a grassy or shrubby understorey. Appropriate fire regimes may also vary, however this is a moot point as very little research is available in relation to Melaleuca wetlands. We do know, however, that *Melaleuca quinquenervia* forests and woodlands can be dynamic, both spatially and temporally. Aerial photos taken between 1941 and the 1980s in the Cooloola area show Melaleuca patches coming and going, to be replaced by, or replacing, wet heaths and sedgelands (C. Sandercoe, pers. comm. 2001). The timing of fire and of extreme wet and dry events probably plays a key role in these dynamics. Like wet heath, these wetlands are vulnerable to peat fires when dry. Planned burns should therefore be limited to times when the substrate is wet.

Only one study was located which might provide a clue as to appropriate fire frequency for this vegetation type. Bartareau and Skull (1994) found frequent burning hindered seedling establishment of *M. viridiflora* near Cardwell in north Queensland. Greater numbers of trees over 1.5m were found in plots burned at 8-10 year intervals, and where there had been no fire for 20 years, than in plots burned every three years or less. The authors recommend a fire frequency of between 10 and 20 years. Observations on SEQ's Russell Island of *Melaleuca quinquenervia* wetlands subject to firing at 1-3 year intervals supports the contention that very frequent fire is not conducive to regeneration. These forests consist of large old trees, some dead, with no young Melaleuca regrowth other than very small seedlings which have almost certainly germinated since the last fire, and which are likely to be eliminated by the next. This vegetation type, like the rocky heaths, may burn in a subset of fires which affect the surrounding heath or forest matrix.

This vegetation type is a high priority for research. Until then, the frequency suggested above remains very speculative.

Conclusion

As noted in the introduction, these guidelines are intended as an aid to management. Readers are encouraged to apply them with discretion in their particular situation, and to supplement this information with their own reading, observation and research. Those seeking further information are referred to the literature review carried out at the beginning of the Fire and Biodiversity project (Tran and Wild 2000). This document is supplemented by a database, which lists over 400 references, with abstracts (Tran 2000). Further details of these and other Consortium products can be found on p. 49.

For those with an interest in carrying out research, the Consortium has produced a list of potential projects to further our understanding of the ecology of south-east Queensland's fire-adapted ecosystems. Ideally, in five years' time the Consortium will be seeking to update these guidelines in the light of findings from studies conducted by enthusiastic researchers and managers throughout the south-east corner.

In the meantime, we sincerely hope this document provides useful guidance to the many people in south-east Queensland working to maintain the richness of our region's diverse natural heritage.

References

Andersen, A. 2000. "Synthesis and management implications." P. 13 in *Fire and Biodiversity. CSIRO's Kapalga Fire Experiment, Kakadu National Park. Workshop Notes.* Darwin: CSIRO.

Ash, J. 1988. "The location and stability of rainforest boundaries in north-eastern Queensland, Australia." *Journal of Biogeography* 15:619-30.

Ashton, D. H. 1976. "The development of even-aged stands of *Eucalyptus regnans* F. Muell. in Central Victoria." *Australian Journal of Botany* 24:397-414.

Ashton, D.H. 1981. "Fire in tall open forests (wet sclerophyll forests)." Pp. 339-66 in *Fire and the Australian Biota*, editors A. M. Gill, R. H. Groves, and I. R. Noble. Canberra: Australian Academy of Science.

Ashton, D. H. and D. G. Martin. 1996. "Regeneration in a pole-stage forest of *Eucalyptus regnans* subjected to different fire intensities in 1982." *Australian Journal of Botany* 44:393-410.

Attiwill, P. M. 1994. "Ecological disturbance and the conservation management of eucalpyt forests in Australia." *Forest Ecology and Management* 63:301-46.

Auld, T. D. and A. J. Denham. 2001. "The impact of seed predation by mammals on post-fire seed accumulation in the endangered shrub *Grevillea caleyi* (Proteaceae)." *Biological Conservation* 97:377-85.

Auld, T. D. and M. A. O'Connell. 1991. "Predicting patterns of post-fire germination in 35 eastern Australian Fabaceae." *Australian Journal of Ecology* 16:53-70.

Baker, J. 2000. "The Eastern Bristlebird: cover-dependent and fire-sensitive." *Emu* 100:286-98.

Bartareau, T. and S. Skull. 1994. "The effects of past fire regimes on the structural characteristics of coastal plain *Melaleuca viridiflora* Sol. ex Gaert. woodlands and the distribution patterns of epiphytes (*Dendrobium canaliculatum* R. Br., *Dischidia nummularia* R. Br.) in northeastern Queensland." *Biotropica* 26:118-23.

Bell, T. L. and J. S. Pate. 1996. "Growth and fire response of selected Epacridaceae of south-western Australia." *Australian Journal of Botany* 44:509-26.

Benson, D. H. 1985. "Maturation periods for fire-sensitive shrub species in Hawkesbury sandstone vegetation." *Cunninghamia* 1:339-49.

Benson, J. S. and P. A. Redpath. 1997. "The nature of pre-European native vegetation in south-eastern Australia: a critique of Ryan, D.G., Ryan, J.R. and Starr, B.J. (1995) *The Australian Landscape - Observations of Explorers and Early Settlers*." *Cunninghamia* 5:285-328.

Benwell, A. S. 1998. "Post-fire seedling recruitment in coastal heathland in relation to regeneration strategy and habitat." *Australian Journal of Botany* 46:75-101.

Birk, E. M and R. G. Bridges. 1989. "Recurrent fires and fuel accumulation in even-aged blackbutt (*Eucalyptus pilularis*) forests." *Forest Ecology and Management* 29:59-79.

Bradfield, G. E. 1981. "Component analysis of fire patterns in open eucalypt forest." *Australian Journal of Ecology* 6:99-109.

Bradstock, R. A. 1990. "Demography of woody plants in relation to fire: *Banksia serrata* Lf. and *Isopogon anemonifolius* (Salisb.) Knight." *Australian Journal of Ecology* 15:117-32.

Bradstock, R. A. 1995. "Demography of woody plants in relation to fire: *Telopea speciosissima*." *Proceedings of the Linnean Society of New South Wales* 115:25-33.

Bradstock, R. A. 2000. "Fire regimes and biodiversity: reconciling cycles of fire with cycles of life in landscapes." Paper presented at the *Fire and Biodiversity in SEQ* seminar, Brisbane, Queensland, 16 November 2000.

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

Bradstock, R. A., M. Bedward, J. Scott, and D. A. Keith. 1996. "Simulation of the effect of spatial and temporal variation in fire regimes on the population viability of a *Banksia* species." *Conservation Biology* 10:776-84.

Bradstock, R. A., D. A. Keith, and T. D. Auld. 1995. "Fire and conservation: imperatives and constraints on managing for diversity." Pp. 323-33 in *Conserving Biodiversity: Threats and Solutions*, editors R. A. Bradstock, T. D. Auld, D. A. Keith, R. T. Kingsford, D. Lunney, and D. P. Sivertsen. Chipping Norton: Surrey Beatty and Sons.

Bradstock, R. A. and P. J. Myerscough. 1988. "The survival and population response to frequent fires of two woody resprouters *Banksia serrata* and *Isopogon anemonifolius*." *Australian Journal of Botany* 36:415-31.

Bradstock, R. A., M. G. Tozer, and D. A. Keith. 1997. "Effects of high frequency fire on floristic composition and abundance in a fire-prone heathland near Sydney." *Australian Journal of Botany* 45:641-55.

Breeden, S. 1994. *Uluru. Looking after Uluru-Kata Tjuta - the Anangu Way*. East Roseville: Simon and Shuster Australia.

Brown, M. J. and F. D. Podger. 1982. "Floristics and fire regimes of a vegetation sequence from sedgeland-heath to rainforest at Bathurst Harbour, Tasmania." *Australian Journal of Botany* 30:659-76.

Burrows, N. 2000. "A fire for all reasons." Pp. 15-20 in *Landscope special edition: Fire the Force of Life*, editor D. Gough. Perth: Department of Conservation and Land Management.

Cary, G. J. and D. A. Morrison. 1995. "Effects of fire frequency on plant species composition of sandstone communities in the Sydney region: combinations of inter-fire intervals." *Australian Journal of Ecology* 20:418-26.

Catling, P. C. 1991. "Ecological effects of prescribed burning practices on the mammals of southeastern Australia." Pp. 353-63 in *Conservation of Australia's Forest Fauna*, editor D. Lunney. Mosman: Royal Zoological Society of NSW.

Catling, P. C. 1994. "Bushfires and prescribed burning: protecting native fauna." *Search* 25:37-40.

Catling, P. C. and R. J. Burt. 1995. "Studies of the ground-dwelling mammals of eucalypt forests in south-eastern New South Wales: the effect of habitat variables on distribution and abundance." *Wildlife Research* 22:271-88.

Chapman, A. and G. N. Harrington. 1997. "Responses by birds to fire regime and vegetation at the wet sclerophyll/tropical rainforest boundary." *Pacific Conservation Biology* 3:213-20.

Cheal, D. C. 1996. "Fire succession in heathlands and implications for vegetation management." Pp. 67-79 in *Fire and Biodiversity: the Effects and Effectiveness of Fire Management* (Melbourne, 8 October 1994), editor J. R. Merrick. Canberra: Biodiversity Unit, Department of the Environment, Sport and Territories.

Cheney, N. P. 1994. "Bushfires - an integral part of Australia's environment." *Year Book of Australia 1995*, editor I. Castles. Canberra: Australian Bureau of Statistics.

Christensen, P. E. and P. C. Kimber. 1975. "Effect of prescribed burning on the flora and fauna of south-west Australian forests." *Proceedings of the Ecological Society of Australia* 9:85-106.

Christensen, P., H. Recher, and J. Hoare. 1981. "Responses of open forests (dry sclerophyll forests) to fire regimes." Pp. 367-93 in *Fire and the Australian Biota*, editors A. M. Gill, R. H. Groves, and I. R. Noble. Canberra: Australian Academy of Science.

Clark, S. S. 1988. "Effects of hazard-reduction burning on populations of understorey plant species on Hawkesbury sandstone." *Australian Journal of Ecology* 13:473-84.

Clark, S. S. and L. C. McLoughlin. 1986. "Historical and biological evidence for fire regimes in the Sydney region prior to the arrival of Europeans: implications for future bushland management." *Australian Geographer* 17:101-12.

Coops, N. C. and P. C. Catling. 2000. "Estimating forest habitat complexity in relation to time since fire." *Austral Ecology* 25:344-51.

Cowling, R. M., B. B. Lamont, and N. J. Enright. 1990. "Fire and management of south-western Australian banksias." *Proceedings of the Ecological Society of Australia* 16:177-83.

Denham, A. J. and R. J. Whelan. 2000. "Reproductive ecology and breeding system of *Lomatia silaifolia* (Proteaceae) following a fire." *Australian Journal of Botany* 48:261-9.

Diamond, J. M. 1975. "The island dilemma: lessons of modern biogeographic studies for the design of natural reserves." *Biological Conservation* 7:129-46.

Dixon, K. W., S. Roche, and J. S. Pate. 1995. "The promotive effect of smoke derived from burnt native vegetation on seed germination of Western Australian plants." *Oecologia* 101:185-92.

Downey, P. O. 1999. "Fire and weeds: a management option or Pandora's box?" Paper presented at the Australian Bushfire Conference, Albury, July 1999. http://life.csu.edu.au/bushfire99/papers/downey/index.html.

Enright, N. J., D. Goldblum, P. Ata, and D. H. Ashton. 1997. "The independent effects of heat, smoke and ash on emergence of seedlings from the soil seed bank of a heathy *Eucalptus* woodland in Grampians (Gariwerd) National Park, western Victoria." *Australian Journal of Ecology* 22:81-88.

Enright, N. J. and B. B. Lamont. 1989. "Seed banks, fire season, safe sites and seedling recruitment in five co-occurring *Banksia* species." *Journal of Ecology* 77:1111-22.

Enright, N. J., B. P. Miller, and A. Crawford. 1994. "Environmental correlates of vegetation patterns and species richness in the northern Grampians, Victoria." *Australian Journal of Ecology* 19:159-68.

Fensham, R. J. 1997. "Aboriginal fire regimes in Queensland, Australia: analysis of the explorers' record." *Journal of Biogeography* 24:11-22.

Fensham, R. J. and R. J. Fairfax. 1996. "The disappearing grassy balds of the Bunya Mountains, south-eastern Queensland." *Australian Journal of Botany* 44:543-58.

Flannery, T. 1994. *The Future Eaters*. Sydney: Reed Books.

Flenady, B. 2000. *Interim Findings from Fire Regeneration Monitoring in Lamington National Park*. Draft paper prepared for Queensland National Parks and Wildlife Service.

Florence, R. G. 1964. "Regeneration burning with seed trees in two Blackbutt (*Eucalyptus pilularis* Sm) forests." *Australian Forestry* 28:157-65.

Florence, R. G. 1996. *Ecology and Silviculture of Eucalypt Forests*. Collingwood: CSIRO Publishing.

Floyd, A. G. 1966. "Effect of fire upon weed seeds in the wet sclerophyll forests of Northern New South Wales." *Australian Journal of Botany* 14:243-56.

Floyd, A. G. 1976. "Effect of burning on regeneration from seeds in wet sclerophyll forest." *Australian Forestry* 39:210-20.

Fox, B. J. 1982. "Fire and mammalian secondary succession in an Australian coastal heath." *Ecology* 63:1332-41.

Fox, B. J. 1983. "Mammal species diversity in Australian heathlands: the importance of pyric succession and habitat diversity." Pp. 473-89 in *Mediterranean-Type Ecosystems: The Role of Nutrients*, editors F. J. Kruger, D. T. Mitchell, and J. U. M. Jarvis. Berlin: Springer-Verlag.

Fox, B. J. and G. M. McKay. 1981. "Small mammal responses to pyric successional changes in eucalypt forest." *Australian Journal of Ecology* 6:29-41.

Fox, M. D. and B. J. Fox. 1986. "The effect of fire frequency on the structure and floristic composition of a woodland understorey." *Australian Journal of Ecology* 11:77-85.

Friend, G. R. 1993. "Impact of fire on small vertebrates in mallee woodlands and heathlands of temperate Australia: a review." *Biological Conservation* 65:99-114.

Gill, A. M. 1977. "Management of fire-prone vegetation for plant species conservation in Australia." *Search* 8:20-26.

Gill, A. M. 1981. "Adaptive responses of Australian vascular plant species to fires." Pp. 243-72 in *Fire and the Australian Biota*, editors A. M. Gill, R. H. Groves, and I. R. Noble. Canberra: Australian Academy of Science.

Gill, A. M. 1997. "Eucalypts and fires: interdependent or independent?" Pp. 151-67 in *Eucalypt Ecology: Individuals to Ecosystems*, editors J. Williams and J. Woinarski. Cambridge: Cambridge University Press.

Gill, A. M. and R. Bradstock. 1995. "Extinction of biota by fires." Pp. 309-22 in *Conserving Biodiversity: Threats and Solutions*, editors R. A. Bradstock, T. D. Auld, D. A. Keith, R. T. Kingsford, D. Lunney, and D. P. Sivertsen. Chipping Norton: Surrey Beatty and Sons.

Gill, A. M. and M. A. McCarthy. 1998. "Intervals between prescribed fires in Australia: what intrinsic variation should apply?" *Biological Conservation* 85:161-9.

Gill, A. M. and A. O. Nicholls. 1989. "Monitoring fire-prone flora in reserves for nature conservation." Pp. 137-51 in *WA Department of Conservation and Land Management Occassional Paper 1/89.* Perth: CALM.

Glasby, P., P. M. Selkirk, D. Adamson, A. J. Downing, and D. R. Selkirk. 1988. "Blue Mountains Ash (*Eucalpytus oreades* R.T. Baker) in the western Blue Mountains." *Proceedings of the Linnean Society of New South Wales* 110:141-58.

Guinto, D. F., A. P. N. House, Z. H. Xu, and P. G. Saffigna. 1998. *Changes in soil chemical properties and forest floor nutrients under repeated prescribed burning in eucalypt forests of south-east Queensland*. Research Note No. 49. Brisbane: Queensland Forestry Research Institute.

Guinto, D. F., A. P. N. House, Z. H. Xu, and P. G. Saffigna. 1999. "Impacts of repeated fuel reduction burning on tree growth, mortality and recruitment in mixed species eucalypt forests of southeast Queensland, Australia." *Forest Ecology and Management* 115:13-27.

Hammill, K. A., R. A. Bradstock, and W. G. Allaway. 1998. "Post-fire seed dispersal and species re-establishment in proteaceous heath." *Australian Journal of Botany* 46:407-19.

Hannah, D. S., G. C. Smith, and G. Agnew. 1998. "Reptile and amphibian composition in prescribed burnt dry sclerophyll production forest, southern Queensland." *Australian Forestry* 61:34-39.

Harrington, G. 1995. "Should we play God with the rainforest?" *Wildlife Australia* Winter 1995:8-11.

Harrington, G. N. and K. D. Sanderson. 1994. "Recent contraction of wet sclerophyll forest in the wet tropics of Queensland due to invasion by rainforest." *Pacific Conservation Biology* 1:319-27.

Harrold, A. 1979. "Heathland regeneration after fire at Noosa." *Queensland Naturalist* 22:88-96.

Herbert, D. A. 1936. "An advancing Antarctic Beech forest." *Queensland Naturalist* 10:8-10.

Hobbs, R. J. and L. Atkins. 1988. "Spatial variability of experimental fires in south-west Western Australia." *Australian Journal of Ecology* 13:295-9.

Hopper, S. D., A. P. Brown, and N. G. Marchant. 1997. "Plants of Western Australian granite outcrops." *Journal of the Royal Society of Western Australia* 80:141-58.

House, A. 1995. "Fire ecology research in Queensland native forests - current status and new directions." Pp. 86-97 in *Working Papers - Sixth Queensland Fire Research Workshop, 8-10 March 1995, Bargara, Queensland* . Toowoomba: Land Use Study Centre, University of Southern Queensland.

House, A., J. Cook, and T. Barrett. 1998. "Crediton burning: rainforest recovery after fire in the Clarke Range, Queensland." Pp. 39-48 in *Working Papers - Seventh Queensland Fire Research Workshop, Cairns, 1-3 July 1998* . Toowoomba: Land Use Study Centre, University of Southern Queensland.

Hughes, P. and B. Hughes. 1991. "Notes on the Black-breasted Button-quail at Widgee, Queensland." *Australian Bird Watcher* 14:113-8.

Hunter, J. T. 1995. "Some observations on the fire responses of two rare species in the Girraween and Bald Rock National Parks." *Queensland Naturalist* 33:146-7.

Hunter, J. T. 1998a. "Notes on the occurrence of *Monotaxis macrophylla* Benth. (Euphorbiaceae), with particular reference to New South Wales." *Queensland Naturalist* 36:21-24.

Hunter, J. T. 1998b. "Granite outcrop vegetation of Wilsons Promontory." *The Victorian Naturalist* 115:322-5.

Hunter, J. T. and P. J. Clarke. 1998. "The vegetation of granitic outcrop communities on the New England Batholith of eastern Australia." *Cunninghamia* 5(3):547-618.

Hunter, J. T., E. Fallavollita, and V. H. Hunter. 1998. "Observations on the ecology of *Muehlenbeckia costata* m.s. (Polygonaceae), a rare fire-ephemeral species occurring on the New England Batholith of northern New South Wales and southern Queensland." *Victorian Naturalist* 115:9-17.

Janzen, D. H. 1983. "No park is an island: increase in interference from outside as park size decreases." *Oikos* 41:402-10.

Johnson, K. A., D. A. Morrison, and G. Goldsack. 1994. "Post-fire flowering patterns in *Blandfordia nobilis* (Liliaceae)." *Australian Journal of Botany* 42:49-60.

Jones, C. 2001. "The great salinity debate: Part III. A way forward." *Stipa Newsletter* No. 15.

Just, T. E. 1978. *Extreme Fire Weather in Queensland*. Brisbane: Department of Forestry, Queensland. Technical Paper No. 9.

Keith, D. 1996. "Fire-driven extinction of plant populations: a synthesis of theory and review of evidence from Australian vegetation." *Proceedings of the Linnean Society of New South Wales* 116:37-78.

Kellman, M. 1986. "Fire sensitivity of *Casuarina torulosa* in North Queensland, Australia." *Biotropica* 18:107-10.

Kington, D. 1997. *Where have the Pretty Face gone?* Unpub. manuscript.

Lamb, D., D. Ash, and J. Landsberg. 1992. "The effect of fire on understorey development and nitrogen cycling in *Eucalyptus maculata* forests in south east Queensland." Pp. 496-506 in *Fire Research in Rural Queensland: Selected Papers from the Queensland Fire Research Workshop Series 1980-1989*, editor B. R. Roberts. Toowoomba: Land Use Study Centre, University of Southern Queensland.

Lamont, B. B., J. M. Olesen, and P. J. Briffa. 1998. "Seed production, pollinator attractants and breeding system in relation to fire response - are there reproductive syndromes among co-occurring proteaceous shrubs?" *Australian Journal of Botany* 46:377-85.

Leigh, J. H. and M. D. Holgate. 1979. "The responses of the understorey of forests and woodlands of the Southern Tablelands to grazing and burning." *Australian Journal of Ecology* 4:25-45.

Lewis, H. T. 1989. "Ecological and technological knowledge of fire: Aborigines versus park rangers in Northern Australia." *American Anthropologist* 91:940-61.

Lindenmayer, D. B., R. B. Cunningham, and C. F. Donnelly. 1997. "Decay and collapse in eastern Australian forests: impacts on arboreal marsupials." *Ecological Applications* 7:625-41.

Lunt, I. D. 1990. "Impact of an autumn fire on a long-grazed *Themeda triandra* (Kangaroo Grass) grassland: implications for management of invaded, remnant vegetations." *Victorian Naturalist* 107:45-51.

Lunt, I. D. 1991. "Management of remnant lowland grasslands and grassy woodlands for nature conservation: a review." *Victorian Naturalist* 108:56-66.

Lunt, I. D. 1994. "Variation in flower production of nine grassland species with time since fire, and implications for grassland management and restoration." *Pacific Conservation Biology* 1:359-66.

Lunt, I. D. 1997. "Effects of long-term vegetation management on remnant grassy forests and anthropogenic native grasslands in south-eastern Australia." *Biological Conservation* 81:287-97.

Lunt, I. D. 1998a. "Two hundred years of land use and vegetation change in a remnant coastal woodland in southern Australia." *Australian Journal of Botany* 46:629-47.

Lunt, I. D. 1998b. "*Allocasuarina* (Casuarinaceae) invasion of an unburnt coastal woodland at Ocean Grove, Victoria: structural changes, 1971-1996." *Australian Journal of Botany* 46:649-56.

Mackowski, C. M. 1984. "The ontogeny of hollows in Blackbutt *(Eucalyptus pilularis)* and its relevance to the management of forests for possums, gliders and timber." Pp. 553-67 in *Possums and Gliders*, editors A. P. Smith and I. D. Hume. Sydney: Australian Mammal Society.

Masters, P. 1993. "The effects of fire-driven succession and rainfall on small mammals in spinifex grassland at Uluru National Park, Northern Territory." *Wildlife Research* 20:803-13.

McCarthy, M. A., A. M. Gill, and D. B. Lindenmayer. 1999. "Fire regimes in mountain ash forest: evidence from forest age structure, extinction models and wildlife habitat." *Forest Ecology and Management* 124:193-203.

McCaw, L. 2000. "Wildfires: living with the threat." Pp. 31-34 in *Landscope special edition: Fire the Force of Life*, editor D. Gough. Perth: Department of Conservation and Land Management.

McFarland, D. C. 1988a. "Fire and the vegetation composition and structure of subtropical heathlands in south-eastern Queensland." *Australian Journal of Botany* 36:533-46.

McFarland, D. C. 1988b. "The composition, microhabitat use and response to fire of the avifauna of subtropical heathlands in Cooloola National Park, Queensland." *Emu* 88:249-57.

McFarland, D. C. 1990. "Flower and seed phenology of some plants in the subtropical heathlands of Cooloola National Park, Queensland, Australia." *Australian Journal of Botany* 38:501-9.

McFarland, D. C. 1992. "Fire and the management of ground parrot habitat." Pp. 483-95 in *Fire Research in Rural Queensland: Selected Papers from the Queensland Fire Research Workshop Series 1980-1989*, editor B. R. Roberts. Toowoomba: Land Use Studies Centre, University of Southern Queensland.

McFarland, D. C. 2000. "Fire in the Cooloola heathlands." Paper presented at the *Fire and Biodiversity in SEQ* seminar, Brisbane, Queensland, 16 November 2000.

McLoughlin, L. C. 1998. "Season of burning in the Sydney region: the historical records compared with recent prescribed burning." *Australian Journal of Ecology* 23:393-404.

Melick, D. R. 1990. "Ecology of rainforest and sclerophyllous communities in the Mitchell River National Park, Gippsland, Victoria." *Proceedings of the Royal Society of Victoria* 102:71-87.

Melville, S. 1995. "The effect of fire on Casuarina (*Allocasuarina littoralis*) regrowth communities: an urban bushland case study." Pp. 69-78 in *Working Papers - Sixth Queensland Fire Research Workshop, 8-10 March 1995, Bargara, Queensland* . Toowoomba: Land Use Study Centre, University of Southern Queensland.

Morgan, J. W. 1998. "Importance of canopy gaps for recruitment of some forbs in *Themeda triandra*-dominated grasslands of south-eastern Australia." *Australian Journal of Botany* 46:609-27.

Morrison, D. A., G. J. Cary, S. M. Pengelly, D. G. Ross, B. J. Mullins, C. R. Thomas, and T. S. Anderson. 1995. "Effects of fire frequency on plant species composition of sandstone communities in the Sydney region: inter-fire interval and time-since-fire." *Australian Journal of Ecology* 20:239-47.

Morrison, D. A. and J. A. Renwick. 2000. "Effects of variation in fire intensity on regeneration of co-occurring species of small trees in the Sydney region." *Australian Journal of Botany* 48:71-79.

Newsome, A. E., J. McIlroy, and P. C. Catling. 1975. "The effects of an extensive wildfire on populations of twenty ground vertebrates in south-east Australia." *Proceedings of the Ecological Society of Australia* 9:107-23.

Nieuwenhuis, A. 1987. "The effect of fire frequency on the sclerophyll vegetation of the West Head, New South Wales." *Australian Journal of Ecology* 12:373-85.

Noble, I. R. and R. O. Slatyer. 1980. "The use of vital attributes to predict successional changes in plant communities subject to recurrent disturbances." *Vegetatio* 43:5-21.

Noble, J. C. 1982. "The significance of fire in the biology and evolutionary ecology of mallee *Eucalyptus* populations." Pp. 153-9 in *Evolution of the Flora and Fauna of Arid Australia*, editors W. R. Barker and P. J. M. Greenslade. Adelaide: Peacock Publications.

Norris, P. and B. Conroy. 1999. "Fire ecology of soil and leaf litter invertebrates." Paper presented at the Australian Bushfire Conference, Albury, July 1999. http://life.csu.edu.au/bushfire99/papers/norris/index.html.

Odgers, B. M. 1999. "Seasonal variation in buried germinable seed banks of grass species in an urban eucalypt forest reserve." *Australian Journal of Botany* 47:623-38.

Posamentier, H. G., S. S. Clark, D. L. Hain, and H. F. Recher. 1981. "Succession following wildfire in coastal heathland (Nadgee Nature Reserve N.S.W.)." *Australian Journal of Ecology* 6:165-75.

Prober, S. M. and K. R. Thiele. 1993. "The ecology and genetics of remnant grassy white box woodlands in relation to their conservation." *Victorian Naturalist* 110:30-36.

Pyne, S. J. 1991. *Burning Bush: A Fire History of Australia*. New York: Henry Holt.

Richards, S. A., H. P. Possingham, and J. Tizard. 1999. "Optimal fire management for maintaining community diversity." *Ecological Applications* 9:880-92.

Roche, S., K. W. Dixon, and J. S. Pate. 1998. "For everything a season: smoke-induced seed germination and seedling recruitment in a Western Australian *Banksia* woodland." *Australian Journal of Ecology* 23:111-20.

Rose, R., G. Wiltshire, and S. Lang. 1999. "The importance and application of spatial patterns in the management of fire regimes for the protection of life and property and the conservation of biodiversity." Paper presented at the Australian Bushfire Conference, Albury, July 1999. http://life.csu.edu.au/bushfire99/papers/rose/index.html.

Russell, M. J. and B. R. Roberts. 1996. "Effects of four low-intensity burns over 14 years on the floristics of a Blackbutt (*Eucalyptus pilularis*) forest in southern Queensland." *Australian Journal of Botany* 44:315-29.

Russell, R. P. and R. F. Parsons. 1978. "Effects of time since fire on heath floristics at Wilson's Promontory, southern Australia." *Australian Journal of Botany* 26:53-61.

Sandercoe, C. 1991. "*Blandfordia grandiflora* R. Br. (Christmas bell) in Queensland - a distribution survey." *Proceedings of the Royal Society of Queensland* 101:119-23.

Sandercoe, C. S. 1989. "A review of fire research in Queensland heathlands." Paper presented at the *Bushfires and Shrublands* Conference, Canberra, 8-10 May 1989.

Saunders, D. A., R. J. Hobbs, and C. R. Margules. 1991. "Biological consequences of ecosystem fragmentation: a review." *Conservation Biology* 5:18-32.

Setterfield, S. 2000. "Plant reproduction." P. 7 in *Fire and Biodiversity. CSIRO's Kapalga Fire Experiment, Kakadu National Park. Workshop Notes.* Darwin: CSIRO.

Shea, S. R., J. McCormick, and C. C. Portlock. 1979. "The effect of fires on regeneration of leguminous species in the northern Jarrah (*Eucalyptus marginata* Sm) forest of Western Australia." *Australian Journal of Ecology* 4:195-205.

Siddiqi, M. Y., R. C. Carolin, and P. J. Myerscough. 1976. "Studies in the ecology of coastal heath in New South Wales. III. Regrowth of vegetation after fire." *Proceedings of the Linnean Society of New South Wales* 101:53-63.

Smith, J. M. B. and I. J. Guyer. 1983. "Rainforest-eucalypt forest interactions and the relevance of the biological nomad concept." *Australian Journal of Ecology* 8:55-60.

Smith, R. B. and P. W. Woodgate. 1985. "Appraisal of fire damage and inventory for timber salvage by remote sensing in mountain ash forests in Victoria." *Australian Forestry* 48:252-63.

Soulé, M. E. 1991. "Land use planning and wildlife maintenance: guidelines for conserving wildlife in an urban landscape." *Journal of the American Planning Association* 57:313-23.

Specht, R. L., P. Rayson, and M. E. Jackman. 1958. "Dark Island heath (Ninety-Mile Plain, South Australia). VI. Pyric succession: changes in composition, coverage, dry weight, and mineral nutrient status." *Australian Journal of Botany* 6:59-88.

Stuwe, J. and R. F. Parsons. 1977. "*Themeda australis* grasslands on the Basalt Plains, Victoria: floristics and management effects." *Australian Journal of Ecology* 2:467-76.

Tasker, E., R. Bradstock, and C. Dickman. 1999. "Small mammal diversity and abundance in relation to fire and grazing history in the eucalypt forests of northern New South Wales." Paper presented at the Australian Bushfire Conference, Albury, July 1999. http://life.csu.edu.au/bushfire99/papers/tasker/index.html.

Taylor, M. 1992. "Fire in subtropical forest management." Pp. 172-84 in *Fire Research in Rural Queensland: Selected Papers from the Queensland Fire Research Workshop Series 1980-1989*, editor B. R. Roberts. Toowoomba: Land Use Study Centre, University of Southern Queensland.

Tolhurst, K. 1996. "Effects of fuel reduction burning on flora in a dry sclerophyll forest." Pp. 97-107 in *Fire and Biodiversity: the Effects and Effectiveness of Fire Management* (Melbourne, 8 October 1994), editor J. R. Merrick. Canberra: Biodiversity Unit, Commonwealth Department of Environment, Sport and Territories.

Tran, C. 2000. *CD-Rom of Fire Regime Literature*. Southeast Queensland: Griffith University and the Fire and Biodiversity Consortium.

Tran, C. and C. Wild. 2000. *A Review of Current Knowledge and Literature to Assist in Determining Ecologically Sustainable Fire Regimes for the Southeast Queensland Region*. Southeast Queensland: Griffith University and the Fire and Biodiversity Consortium.

Tremont, R. M. and S. McIntyre. 1994. "Natural grassy vegetation and native forbs in temperate Australia: structure, dynamics and life histories." *Australian Journal of Botany* 42:641-58.

Turner, J. 1984. "Radiocarbon dating of wood and charcoal in an Australian forest ecosystem." *Australian Forestry* 47:79-83.

Turner, M. G., W. W. Hargrove, R. H. Gardner, and W. H. Romme. 1994. "Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming." *Journal of Vegetation Science* 5:731-42.

Turton, S. M. and G. A. Duff. 1992. "Light environments and floristic composition across an open forest-rainforest boundary in northeastern Queensland." *Australian Journal of Ecology* 17:415-23.

Unwin, G. L. 1989. "Structure and composition of the abupt rainforest boundary in the Herberton Highland, North Queensland." *Australian Journal of Botany* 37:413-28.

Unwin, G. L., G. C. Stocker, and K. D. Sanderson. 1985. "Fire and the forest ecotone in the Herberton Highland, North Queensland." *Proceedings of the Ecological Society of Australia* 13:215-24.

Vanderwoude, C., A. N. Andersen, and A. P. N. House. 1997. "Ant communities as bio-indicators in relation to fire management of spotted gum (*Eucalyptus maculata* Hook.) forests in south-east Queensland." *Memoirs of the Museum of Victoria* 56:671-5.

Wardell-Johnson, G. 2000. "Managing for refugia in fire-prone landscapes." Paper presented at the *Fire and Biodiversity in SEQ* seminar, Brisbane, Queensland, 16 November 2000.

Wark, M. C., M. D. White, D. J. Robertson, and P. F. Marriott. 1987. "Regeneration of heath and heath woodland in the north-eastern Otway Ranges following the wildfire of February 1983." *Proceedings of the Royal Society of Victoria* 99:51-88.

Watson, P. J. 1999. *The effect of fire frequency and time-since-fire on shrub flora in Girraween National Park*. Masters thesis, Griffith University.

Whelan, R. J. 1995. *The Ecology of Fire*. Cambridge: Cambridge University Press.

Williams, J. E. and A. M. Gill. 1995. *The Impact of Fire Regimes on Native Forests in Eastern New South Wales*. Sydney: NSW National Parks and Wildlife Service.

Williams, J. E., R. J. Whelan, and A. M. Gill. 1994. "Fire and environmental heterogeneity in southern temperate forest ecosystems: implications for management." *Australian Journal of Botany* 42:125-37.

Williams, P. R. 2000. "Fire-stimulated rainforest seedling recruitment and vegetative regeneration in a densely grassed wet sclerophyll forest of north-eastern Australia." *Australian Journal of Botany* 48:651-8.

Wilson, B. A. 1996. "Fire effects on vertebrate fauna and implications for fire management and conservation." Pp. 131-47 in *Fire and Biodiversity: the Effects and Effectiveness of Fire Management* (Melbourne, 8 October 1994), editor J. R. Merrick. Canberra: Biodiversity Unit, Commonwealth Department of Environment, Sport and Territories.

Woinarski, J. C. Z. 1999. "Fire and Australian birds: a review." Pp. 55-180 in *Australia's Biodiversity - Responses to Fire. Plants, Birds and Invertebrates*, by A. M. Gill, J. C. Z. Woinarski, and A. York. Canberra: Department of the Environment and Heritage.

Yates, C. J. and R. J. Hobbs. 1997. "Temperate eucalypt woodlands: a review of their status, processes threatening their persistence and techniques for restoration." *Australian Journal of Botany* 45:949-73.

York, A. 1999. "Long-term effects of repeated prescribed burning on forest invertebrates: management implications for the conservation of biodiversity." Pp. 181-266 in *Australia's Biodiversity - Responses to Fire. Plants, Birds and Invertebrates*, by A. M. Gill, J. C. Z. Woinarski, and A. York. Canberra: Department of the Environment and Heritage.

Consortium publications

The SEQ Fire and Biodiversity Consortium is producing a suite of materials to support land managers and those who work with them. Materials completed, or nearing completion as of July 2001 include:

A comprehensive literature review

Tran, C. and C. Wild. 2000. *A Review of Current Knowledge and Literature to Assist in Determining Ecologically Sustainable Fire Regimes for the Southeast Queensland Region.* 106pp.

A database of fire ecology literature

Tran, C. 2000. *CD-Rom of Fire Regime Literature.*

A database of SEQ fire research and monitoring projects

Tran, C. and P. Maidens. 2000. *Southeast Queensland Fire and Biodiversity Research Studies Database.*

Ecological guidelines, for professionals who want a moderately in-depth summary of the management implications of the fire ecology literature (this document)

Watson, P. 2001. *The Role and Use of Fire for Biodiversity Conservation in South-east Queensland: Fire Management Guidelines Derived from Ecological Research.* 49pp.

A list of potential research projects

List of potential fire ecology research projects and contacts. 14pp.

An introductory fact sheet for private landholders and the general public

Watson, P. 2001. *Fire and Nature Conservation in Southeast Queensland: an Introduction.* 4pp.

A more comprehensive fact sheet for landholders and community group members

Fire in Bushland Conservation. 12-16pp.

A fact sheet for Land for Wildlife landholders, produced in conjunction with Land for Wildlife

Moran, C. and P. Watson. 2000. *Fire as a Wildlife Habitat Management Tool.* Land for Wildlife Note No. 14. Land for Wildlife Program South-east Queensland. 8pp.

A kit to assist landholders develop a fire management plan for their property

Individual Property Fire Management Planning Kit: Balancing Fire Safety with Conservation of Bushland Plants and Animals. 41pp.

Copies of these materials can be obtained from the South-east Queensland Regional Bushcare Facilitator, Queensland Parks and Wildlife Service, phone (07) 3202-0223.