

# **A review of fire regimes of the forested region of south-western Australia with selected examples of their effects on native biota**

**A. Malcolm Gill<sup>1</sup>**

Centre for Plant Biodiversity Research  
CSIRO Plant Industry

---

<sup>1</sup> Retired August 31, 2000; presently an Honorary Research Fellow, CSIRO Plant Industry.

## **Abstract**

New methods for the detection of fire regimes hold great promise. Added to the limited use of charcoal in sediments, tree rings and fire scars, plant life histories, statistical records and maps, are the stains on stems of *Xanthorrhoea*. The last of these suggest that fires have occurred on average every three years in jarrah forest during the period immediately prior to white settlement but that the interval between fires increased subsequently. Statistical records for the past 40-50 years show that prescribed burning has dominated the between-fire interval. The interval between fires decreased during the 1960s but increased thereafter. In a comparison of two modern periods, data from Departmental maps in the Collie District suggested that there has been a decrease in the spread of burning season and in the variation in the between-fire interval. Fire intensities have varied throughout history but, with a predominance of prescribed fires, fire intensities will be predominately low in jarrah forest. What the effects of the fire regimes have been in the southwest is difficult to say due to the lack of data from any systematic monitoring program and the historically changing effects of settlement, silviculture and *Phytophthora* disease. However, available case histories demonstrate the value of very long intervals (or no fires at all) to some species (*e.g.* to maximize populations of the noisy scrub bird) and the neutral effect of current fire regimes on studied plant communities in jarrah forest. Thicket-generating fires have been seen as part of the management for woylie populations but, with the control of the exotic fox, the need for thickets may be less than it was. It is suggested that further study be undertaken on the effects of fire regimes on orchids and certain species of frogs. A concerted effort should be made to find species which are the most vulnerable to shifts in fire regimes and to quantify their vulnerability. Any evaluation of the significance of fire regimes and their historical changes should take place within the context of explicit management aims. In the present case-history, the conservation of the native biota has been the focus, not wood products, water or recreation.

## **Introduction**

The south-western Australian region is an isolated and important region of eucalypt forest but: "Within the outer boundary of the forest .. areas of woodland, heath, swamp, shrubland, sedgeland and lithic complex occur" (Abbott 1999). This region is the scene for the present regional case-history on fire regimes and biodiversity in Australia. Companion case histories, of vastly larger areas, address issues in savannas of the tropical north (Dr J. Russell-Smith) and mulga shrublands of the arid zone of Australia (Dr. J.E. Williams). These reports, and a study of remotely sensed fire occurrences in Australia (Dr R. Smith and Mr R. Craig, Western Australian Department of Land Management), result from a contract with *Environment Australia*. Collectively, these reports reflect the need for the continuity of the monitoring of fires on an Australia-wide basis if we are to understand the effects of fire regimes on our diverse landscapes.

The south-west forest region has a 170-year history of association with white settlement. During this time there have been large areas cleared of forest and what forest remains has been the focus of many human activities. In this case-history, the nature of the fire regimes of the south-west forest region and their effects on the

native biota is the topic. Here we examine the evidence for fire regimes and their changes over decades and centuries together with the effects of fire regimes on various taxa or communities of organisms.

The eucalypt forests of south-western Australia are a valuable resource for recreation, water supply, minerals, wood products and biodiversity. Forest management there, including fire management, is controversial (Abbott and Christensen 1994, Calver *et al.* 1996, Hobbs 1996, Conacher and Conacher 2000). Resolving issues is made difficult by the absence of a long-term formal monitoring plan (Hobbs 1996). Having no long-term monitoring is not at all unusual in Australia but it does mean that making definitive statements about the efficacy of fire management for biodiversity as a whole in any particular area is impossible. What is particularly valuable about the fire management of south-western forests is that records of fires have been kept for decades; this is rare. Also, recently, there has been the recognition of char-rings on grass-trees and their value for assessing fire histories. These types of data enable estimates to be made of average fire intervals for certain eras.

### **Landscapes of the south-western region (after Gill and Moore 1997)**

The study region has a relatively high rainfall but markedly dry summer and autumn (Gentilli 1989). In general, mean annual rainfall declines to the north and east. In the far south, rainfall is high and the summers not as dry as in the north (Milewski 1979, Gentilli 1989). A detailed climatic classification was devised by Milewski (1979).

Along the west and south coasts are dune systems (Mulcahy 1973). Paralleling the west coast there is a major north-south fault line expressed as an escarpment delimiting a narrow, often lime-rich, coastal zone to the west and lateritic uplands to the east (Mulcahy 1973). Laterites become dissected (Mulcahy 1973), and rainfall declines, from the escarpment to the east (Gentilli 1989).

In the region, a total of 312 vegetation complexes has been defined by Matiske and Havel (1997, cited in Western Australian Regional Forest Agreement 1998). Woodlands and forests are common on the coastal plain, forests dominate the wetter parts of the lateritic terrain while woodlands, mallee and heath are common on the more eastern, dissected, lateritic terrain and in the lower rainfall areas. "Lithic complexes" (Abbott 1999) are common on the uplands while swamps are common in the lowlands.

Two eucalypt species dominate the forests of south-western Australia. They are the jarrah (*Eucalyptus marginata*) and the karri (*E. diversicolor*) although about 60 eucalypt taxa occur there ("Group L" of Gill *et al.* 1985). While jarrah and karri were, and are, the mainstay of the hardwood timber industry in the south-west, the Forests Department (and its successor CALM - formed by the amalgamation of a number of Departments, including the Forests Department, in 1985) was (and is) responsible for large areas of mallet plantation and sandalwood (*Santalum spicatum*) areas together with smaller areas of tuart (*E. gomphocephala*) forest. The main types of forest are sketched below.

Jarrah is the most common tree species of the southwestern forests. It is often associated with another species of eucalypt, marri (*E. calophylla*, syn. *Corymbia*

*calophylla*). The forest has a mature tree height of 20 to 35m with high quality jarrah exceeding 27m in height (Abbott and Loneragan 1986). Jarrah can be found outside the forests as a mallee a few metres tall (personal observation). In 1982 *ca.* 72% of the forest estate was typified as jarrah (Beggs 1982). If areas classified as “wandoo” - which includes some jarrah - are added, the figure becomes 77% (Beggs 1982).

Jarrah covers most of the forest belt. Detailed descriptions of the northern jarrah forest and its environment are provided by Havel (1975*a,b*) and Dell *et al.* (1989). Detailed information on the southern jarrah forest is to be found in Strelin (1988). Already mentioned is the work of Mattiske and Havel (1997, cited in Western Australian Regional Forest Agreement 1998).

Karri forms magnificent forests in the wetter areas of the far southwest. Tree height can reach 85m but mature heights of 70m are more common (Bradshaw and Lush 1981). Karri stands are interspersed with stands of jarrah and jarrah-marri (Bradshaw and Lush *loc. cit.*). Further information is provided by the cited authors and also Breidahl and Hewett (1992) and Hopper *et al.* (1992). In 1982 *ca.* 7% of the forest estate was typified as karri (Beggs 1982). Included in the “karri” category in 1982 were the tingle forests (Beggs 1982). Floristic associations of the ‘tingles’ - a group of endemic south-western eucalypt species (*E. brevistylis*, *E. jacksonii* and *E. guilfoylia*) forming tall forests in the southern forest belt - have been described recently by Wardell-Johnson and Williams (1996).

Tuart (*E. gomphocephala*) is found on the “yellow sand over limestone” on the coastal plain (see Havel 1968) but formed only small areas of prime forest (Kessell 1929, p.6). Early observations revealed “little or no regeneration for several decades” possibly due to “frequent burning and overgrazing” (Kessell 1929, p.6). A detailed history of the tuart region has been given by Ward (2000).

## Fire regimes

The effects of fires on biodiversity are a function of the fire regime. The fire regime consists of the components of intensity, seasonality, interval between fires and the type of fire (Gill 1975). In the last category are the relatively rare, but often biologically important, peat fires (Gill 1975). Spatial aspects of fire occurrence can be important and these are mentioned in the Section on biodiversity and fire regimes (below).

Sources of information for determining fire regimes are:

- observation (information as in Hallam 1975 for the activities of Aboriginal people);
- trees with annual rings and/or fire scars (point based) - Burrows *et al.* (1995), Bradshaw and Rayner (1997*a,b*);
- grasstree (*Xanthorrhoea* spp.) markings (point based) - Ward and van Didden (1997), Ward (2000);
- statistical records of government departments (area based, but locations not mapped) – Gill and Moore (1997);
- maps from departmental records (area based) - Lang (1997);
- point monitoring;

- use of life history and other attributes of selected plant and animal species (Christensen and Annels 1985);
- quantitative modelling of life history and other attributes of selected plant and animal species (e.g. McCarthy *et al.* 1999);
- palynological and charcoal data (point measured, but area influenced, e.g. Churchill 1968).

*Between fire intervals using inferences from Departmental statistics*

If the areas burned per year are known from a region with a known area then the proportion burned per year can be calculated. In the south-western forests records have been kept for part of the region since 1936. Since 1951-52, when prescribed burning became an integral and systematically recorded part of forest management, statistics are available from annual reports. Gill and Moore (1997) investigated these data and found that the areas, or proportions, burned per year by prescription increased linearly from 1951-52 to 1968-69 in a statistically significant way. During this time the proportion burned increased to triple the early amount (from about 0.08 to 0.25). In the years that followed, the proportion then declined in a linear way to values similar to those at the start of the prescribed burning program. If the simplest model for converting proportions burned to average interval between fires is used (*i.e.* the reciprocal), then the average intervals between prescribed fires at the beginning and end of the study period were *c.*12 years, while in the middle of the period the interval briefly touched 4 years. Both these limits would expand proportionately given that an average of 70% of areas recorded as burning (from records of areas of blocks burned) actually do burn, to 18 and 6 years respectively. Note that these figures take no account of the fact that the ridges are likely to burn every time but that the valleys, especially in the south, may not. There is no account taken of burning in different vegetation types in these data. Unplanned fires add a little to the areas burned, but a relatively small amount.

*Between fire intervals in jarrah forest from Departmental maps*

Lang (1997) began the systematic process of examining the abundant historical data held by the Department of Conservation and Land Management in the form of maps of fire areas held on microfilm (approximately 2000 frames). In the course of Lang's study only a small amount of the total data holdings was used, *viz.* the data for the Collie Forest District for the period from 1937-38 to 1987-88. In the period from the start of the data to 1958-59, the average between-fire interval calculated by Lang (1997) averaged 8.8 years while in the period after that it averaged 6.1 years. From the first of these periods to the second there appeared to be a shift from more random to less random fire intervals, and an increasing prominence of low intensity spring burning. The accuracy of maps is likely to have improved over the period of study and could have had some effect on the output from the analysis.

*Between-fire intervals from tree rings and fire scars on jarrah*

A classical technique from the Northern Hemisphere is the use of annual tree rings to provide a calendar date in years and to relate these to fire scars on the tree. This technique has been tried in a number of places but most Australian trees produce equivocal results either for the tree rings (not necessarily annual), or for the frequency

of scarring (not necessarily equal to the frequency of fires). In open-forests of the south-western region, tree rings and fire scars have been examined in order to determine changes in fire intervals (Burrows *et al.* 1995). Unfortunately, “the resilience of jarrah to injury by fire and the limitations of ring counting as an ageing technique, prevented an accurate reconstruction of the fire frequency”. Even so, Burrows *et al.* (1995) were able to estimate that the average interval between tree-scarring fires was about 80 years. This probably provides an upper limit for between-fire interval in the places where these trees were found.

*Between-fire intervals from tree rings alone* (Bradshaw and Rayner 1997a,b)

From a comprehensive investigation into the ring counts of karri trees and the distribution of cohorts of the trees, the authors found a peak in the age-class distribution at 150-250 years. They considered a number of possible reasons why the distribution was bell-shaped rather than, say, a negative exponential relationship. Their conclusion was that: “the primary reason for the relatively large proportion of middle-aged stands is the existence and continued recruitment of younger cohorts in multi-aged stands”. As karri is believed to recruit mainly after fire (Christensen and Annels (1985), understorey age distributions, and the sensitivity of cohorts of different age to fires of varying intensity, may provide the key to overstorey history. However, age distributions of current understoreys in karri forests have been strongly influenced by management practice (Christensen and Annels 1985). After considering life-cycle evidence, these authors suggest prescribed burning should take place at 15-20 year intervals but the mean interval during late Aboriginal occupation is unknown.

*Between-fire intervals: methods and results using grasstrees of the jarrah forest* (Ward and van Didden 1997)

The grasstree data represent a significant new development for the understanding of the fire history of south-western Australian forests. Developed by Ward and van Didden (1997), the data show negative effects of rainfall, plant height and time on the average number of fires per decade. In the period from about 1750 until 1859 there appear to have been just under 3 fires per decade on average. The detailed analyses have not yet been reported but their regression equation suggests that higher rainfall plants would have experienced fewer fires and lower rainfall ones more fires. An exception to this, as they reported, would be on the rocky outcrops in the drier areas, in *Eucalyptus wandoo* country where the fire interval made apparent on the grasstrees may have been nearer 10-20 years while that in the grassy valleys was about 2-3 years. Fires at intervals less than two years apart seem unlikely in the absence of grass.

Based on *a priori* reasoning it would appear that there could be curvilinear relationships, rather than linear ones (as indicated by the regression model), between scars per decade, rainfall and plant height. We might expect that, as the environment becomes wetter, numbers of fires per decade would decrease due to decreasing opportunity for fire spread – just as the multiple-regression model indicates (Ward and van Didden 1997, p.13). However, the same effect may be expected as rainfalls become very low and fuels become discontinuous - if the data domain extended far enough. Thus a non-linear model may be appropriate but apparently remains unexplored (or was dismissed as irrelevant).

For the influence of plant height, one might expect that plants with long leaves reaching the ground would be easily ignited. As plants increased in height and experienced more fires, dead leaves could become out of reach of flames from low intensity fires so decreasing chances of their ignition. Vines (1968) illustrated a lack of ignition in taller grasstrees in jarrah forest during a prescribed, low intensity fire: “the fires have been so mild that even inflammable blackboys [grasstrees] remain unburnt”. Thus, the probability of registering a fire will depend on the relationship of plant height, leaf length and flame height. Ward and van Didden (1997) recorded an effect of plant height in their multiple regression in concert with this argument. Again, however, we might expect the relationship between plant height and the registration of fires on the stem to be non-linear because of the need for a particular height to be reached before plants escape even the lowest intensity fire.

Ward and van Didden’s (1997, p.15) graph shows a curvilinear relationship between fires per decade as a function of time from 1750 although, in the period from 1860 to 1990, the relationship approaches linearity with a decline from near 3 fires per decade to much less than one per decade. It is not known whether this graphical ‘summary’ compensated in any way for the effects of plant height and rainfall.

Ideally, the “crude preliminary test of the hypotheses” mediated through multiple regression (Ward and van Didden 1997) will be followed by an analysis of variance incorporating within-individual variance, between-individual variance, and between-site variance. The effects of height of the plant, rainfall and time will all be important factors (Ward and van Didden 1997) but, as noted above, curvilinearity of relationships may need to be accounted for.

Note that the expression of results on a decade-to-decade basis will smooth the results where changes have been rapid – as from the time of the introduction of the prescribed burning program (see above). Thus, the results for the presumed interval between fires based on the expansion and subsequent decline of prescribed burning would be conservative compared with the annual data.

Ward (2000) has recently interpreted the marks on the grasstrees of the tuart forest. He found a similar fire interval in the period of Aboriginal activity to that in jarrah forest (median interval of about 3 years) but rising rapidly in recent decades to 25 in Yalgorup National Park.

#### *Between-fire intervals: methods and results using palynological techniques*

Churchill (1968) studied pollen sequences in the peats of southwestern Australia. In these sequences he found charcoal to be common for at least the past 5,000 years. For current purposes the most detail comes from Weld Swamp, a swamp surrounded by a dense woodland of jarrah, marri and karri.

*“A sandy horizon between 120 and 150cm contains abundant charcoal which provides evidence of at least one fire of sufficient magnitude to burn the peat and severely truncate the bog. This fire was certainly much more intense than the periodic fires which spread charcoal across the surface of the peat at other horizons.”* (Churchill, 1968, p.140).

Churchill's (1968) analysis of the peat deposits tells us that there have been periodic fires for millennia but it is unable to provide us with quantitative data as to the nature of fire regimes. The analysis does reveal that there have been peat fires in the southwest but, contrary to Churchill's conclusion quoted above, the charcoal deposit "truncating the bog" reveals that the peat was dry enough to combust – possibly during a severe drought – but does not reveal to us anything about fire intensity. It is possible of course that fires during this period were "much more intense" than usual if the daily weather was appropriate and the fuel loadings were sufficiently high. If "intensity" for these peat fires was just a synonym for the 'depth of burning' this seems appropriate but the implication of the text is that "intensity" applies to the fires in the "dense woodland" of the area. The application of an "intensity" concept – in the sense of Byram (1959) - to peat fires presently would be considered misplaced.

### *Silvicultural systems and fire regimes*

Forestry is a major industry in the eucalypt forests of south-western Australia and "About 33 per cent of jarrah forest and 46% of karri forest have been excised as national parks, nature reserves, and road, river and stream reserves and will not be logged in the future" (Abbott 1999). In forestry areas two main silvicultural systems may be distinguished (after Gill and Moore 1997). In the jarrah system fires are designed to be of low intensity even after logging. The main prescribed-fire circumstances in jarrah are: litter fires for fuel reduction; pre-logging burning; and top burning after selective or group logging. In the karri system there are litter fires for fuel reduction and high intensity slash-disposal fires. Fire is excluded from regenerating karri for 15 years (Abbott 1999, p. 69). There are many variations on silvicultural themes such as logging of jarrah for thinning, gap creation and shelterwood with the retention of "habitat trees" while, in karri forest, thinning or clearcutting with retained seed trees may take place (Abbott 1999, p. 69). Silvicultural practices are a function of time.

## **Biodiversity and fire regimes**

Biodiversity is the variety of life. This is often categorized in terms of genes, species and communities but in this report only populations of organisms, species and communities will be discussed, albeit briefly.

Sources of information for the depiction and understanding of fire-regime effects are: anecdotal observation; comparison of biota in areas burned at different times in the past; experiments; life-history knowledge; regular monitoring; and, palynology.

Commentary on the effects of fire regimes on biodiversity will be contemporary, only, and relate to:

- the abundances of noisy scrub birds in the absence of fire and how this has changed over about 20 years;
- the interaction between woylies, shrub cover, foxes, fungi and fires in the jarrah forest;
- hollows as habitats;
- plant species in a jarrah forest; and,
- orchids and frogs.

These examples have been chosen because of the substantial amounts of research conducted on them in Western Australia, with the exception of orchids and frogs. The better-studied examples are instructive in terms of fire management.

Fire regimes are not static. They are influenced by the changing ideas of management practice, by the changing rates of ignition of unplanned fires and by the efficacy of suppression. Management practices such as the control of exotic plants and animals (including exotic diseases such as the soil-borne *Phytophthora*), silvicultural practices (including logging), prescribed burning, fragmentation of the vegetation and fire suppression, affect the fire regime. All these practices have changed to varying degrees over the last 50 years. The extent of the sensitivity of much of the biota to these changes is largely unknown.

### *Fire regimes and noisy scrub birds*

After fears that the secretive, but noisy, noisy scrub bird (*Atrichornis clamosus*) had become extinct after sightings in the 1800s, a population of the bird was discovered at Two Peoples Bay, 40km east of Albany (Smith 1985a). There, on a rocky headland with deeply incised drainage lines, the vegetation consists of heath, thicket and low forest (Smith 1985b). This was the last remnant of a wider range of habitats of the bird that had included karri forest (Abbott 1999, p.23). While the following account concerns one bird species, albeit a rare one, it may be noted that a similar narrative may apply to other bird species at Two Peoples Bay (Abbott 1999) and to the long-nosed potoroo (*Potorus tridactylus*) recently discovered in the same area (Start *et al.*, 1995).

The best habitat for the noisy scrub bird at Two Peoples Bay is “low forest” (Smith 1985c) while that of the first reports of the species were in forests marginal to swamps (Smith 1985a). “The dense layer of rushes and shrubs at the edge of the swamp provides cover as well as the sites and material for nests” while sites uphill develop a thick layer of litter – the feeding habitat (Smith 1985a). At Two Peoples Bay, the breeding habitat is only 4-10m wide and centred on creeks (Smith 1985a).

Aspects of the habitat for these birds in relation to fire have been noted by Smith (1985c). Perhaps the most significant observation has been the increase in the number of birds in Two Peoples Bay reserve due to the exclusion of fire. If burnt, habitat becomes available after 4 (in good habitat) to 10 (in poor habitat) years. It is unknown how long the habitat remains suitable but the period appears to be more than 50 years (Smith 1994).

Fires every 4 years over the whole breeding area would lead to the elimination of the species. In coastal areas, apparently, settlers adopted “the practice of burning areas of heath and thicket every two to three years to provide new growth for stock” (Smith 1985b) but what the fire interval may have been in the wetter gullies, the best habitat, is unknown. Even so, the areas away from the gullies are the prime feeding areas so too-frequent burning there would eliminate the food source even if the gullies remained unburned.

If the area was all low forest (the best habitat) and fires had an equal chance of fire occurrence at all times (a random model), then a mean between-fire interval of two years would make 94% of the habitat unsuitable (first suitability at 4 years); at a 4 year mean interval this figure would be 68% and at a mean of 10 years 34%. Smith (1985*b*) argues that a fire interval of the order of 50 years seems appropriate. Given the previous assumptions, the proportion of unsuitable habitat would then be about 8% but the area having had no fire for at least 50 years would be about 33%. It is not known how the birds respond to such older habitats (Smith 1985*b*) but they may persist to 50 years at least (Smith 1994). If the fires were at fixed intervals, rather than random, these figures would change, the chances of extinction being greater when intervals were short.

The noisy scrub bird has a territory the order of 6 ha in area but most of its activity may be confined to an inner core of just over one hectare (Smith 1985*a*). Given that the birds are poor dispersers (Smith 1985*b*), the sizes of areas burnt may be important to the success of the species in the area. Burning small areas may delay colonization because of increased grazing pressure from kangaroos (Smith 1985*b*). The maximum size of this “small area” for a significant grazing effect would depend on the density of kangaroos.

The pattern of burning before the arrival of white settlers is not known. However, it can be inferred that survival of the scrub birds at Two Peoples Bay was due to the rocky terrain and the presence of wet gullies; these fine-grained landscape features would tend to protect pockets of vegetation from fire for considerable periods and allow small populations of the birds to persist.

The case of the noisy scrub bird is interesting in that it suggests that the mean fire interval for maximum population size is decades long. It is possible that the mean fire interval previous to white settlement was shorter than this (Ward and van Didden 1997, Ward 2000) and that population sizes were very low. A common misconception is that a species surviving to the time of white settlement necessarily did so at maximum population sizes and that the fire regime at the time was therefore optimal. Management today can strive for survival of species, only, or it can aim for maximum ‘performance’ (numbers, breeding pairs, biomass *etc.*). Assessing the state of the environment, therefore, depends on the value systems implicit in the assessment.

Any prescribed burning that does take place for the survival or maximum performance of noisy scrub birds should follow some random pattern (McCarthy *et al.* in press) through time at the appropriate mean interval as this will ensure a diversity of habitat. Having said this, it should be noted that the discussion has centred around fire intervals whereas it is likely that there are effects of season of burning, fire intensities and even types of fires (burning peat in swamps for example *vs* surface fires). The observations about kangaroo grazing pressures are also pertinent. Keeping kangaroos at maximum population sizes would possibly conflict with the aim of maintaining maximum populations of noisy scrub birds.

For a detailed study of all the bird species of the “primordial forest” and their subsequent decline, stasis or ascendancy, see Abbott (1999).

A classic example of the interactions of fires and ecosystems began with the studies of Christensen on woylies (Christensen 1980). Woylies are one of a group of species of small kangaroo-like marsupials – sometimes called rat kangaroos - scientifically known as *Bettongia* spp. and *Potorus* spp. The bettongs were widespread in Australia but the potoroos were confined to the wetter areas of south-eastern mainland Australia, Tasmania and south-western Australia (Ride 1970). Woylies (Christensen 1980), and potoroos (Claridge 1992) are consumers of hypogeous (underground) fungi like *Mesophellia* spp. and their ingestion of the fungus may be essential for the germination of its spores (Lamont 1995). Claridge and May (1994), however, cast doubt on how essential the passage of spores through the animal was for germination.

*Mesophellia* spp. are truffle-like fungi whose hyphae form mycorrhizal associations with vascular plants (Ashton 1976, Claridge *et al.* 1992). Their occurrence as hyphae may be influenced by fire regimes (Warcup 1981) but their fruiting seems to be stimulated by the occurrence of fire (Christensen 1980, Taylor 1991, Johnson 1997). Among macrofungi in general, Robinson (1999) found large numbers of species exclusively fruiting in burnt karri forest. What the mechanism of the fruiting stimulus may be remains unknown but is possibly due to a rise in pH (Warcup 1981). However, Claridge (1992) doubted that there was a fire stimulus because sporocarps ‘appeared’ within a few days of fire; he suggested that *Mesophellia* sporocarps, found deeper in the soil than those of other truffle-like fungi survived fire better and thus dominated the post-fire fungal resource. In Johnson’s (1997) study the biomass of sporocarps was highest at 4 years after fire. For the animals surviving a fire, these fungi – stimulated by fire or not - are an important and convenient source of food.

Christensen (1980) noted the importance of scrub-with-gaps to the survival of woylies. The thickets provide dense cover which allows them to escape from predators like the fox and native cat. A fire may destroy the thicket and expose the animals to predators but these do not necessarily eliminate the population; some animals survive and breed (Christensen 1980, Christensen and Maisey 1987). Dense thickets are required by the tammar (*Macropus eugenii*) which co-exists with the woylie (Christensen 1980). Thickets are favoured in their regeneration by relatively intense fires which stimulate the hard seed of thicket species to germinate (Christensen 1980). Low intensity fires over moist soils may provide only weak regeneration and thereby allow the depletion of the thicket to occur.

Predation from foxes is possibly the major reason for the importance of cover for woylies and tammars (Christensen 1980). Christensen noted that the eagle and the native cat were also predators. What the role of the dingo was after its introduction about 5000 years or so ago (Corbett 1985) is unknown. With the removal of foxes, woylie populations have increased dramatically (Start *et al.* 1998).

For woylies alone the burning of forest at 8 to 12 years in spring seemed appropriate but for the tammar, occasional autumn fires were also deemed necessary in order to regenerate the valley thickets (Christensen and Maisey 1987). Thus, for one or two rotations of 6-7 years, low intensity fires were scheduled in spring to burn the ridges then, in the next rotation, an autumn fire was prescribed (Christensen and Maisey 1987). Because intensities in normal autumn burning were too low for suitable regeneration, higher intensities were achieved by hand burning. Christensen and Maisey (1987) pointed out that burning only small areas led to overgrazing by

kangaroos which destroyed the seedling regeneration on which thicket formation depended; the size of patch burned was important. Patches up to 500ha may be needed.

The whole fire-cover-woylie-fungus-predator interaction appears to be simplified in the absence of predators, especially foxes (Christensen 1980) to a fire-woylie-fungus system or even to a woylie-fungus system. In the absence of foxes the cover requirements may be reduced just as they are for potoroos in fox-free Tasmania (Johnson 1997). It is possible that, in the absence of fire, the woylie could persist – like the tammar does on offshore islands without foxes or dingoes (Christensen 1980) – and presumably as the long nosed potoroo has at Two Peoples Bay (see Calver and Dell 1998).

### *Jarrah, termites, fungi, fires and hollow-dependent vertebrates*

There are large numbers of native vertebrate animals, at least, that are largely, or completely, dependent on hollows in trees for nesting, denning or shelter (Ambrose 1982, in Gibbons and Lindenmayer 1997). Hollows are used by 18 species of birds in the forests of south-western Western Australia (Abbott 1999) and 16 mammals and 5 reptiles (Abbott quoted in Western Australian Regional Forest Agreement 1998). Fires have a place in affecting the demography of hollows but the nature of this has not been elaborated. Given the importance of hollows, a scenario of hollow development is sketched out here in relation to the role of fires (see also Gill and Catling in press).

The general sequence of hollow development is: breach of the bark of the tree to expose the sapwood; attack of sapwood and heartwood by fungi; and, attack of decomposing wood by termites (Perry *et al.* 1984). The most common bark-breaching agents observed in jarrah by Perry *et al.* (1984) were fire and machines but other mechanisms are known such as bark splitting (associated with drought) and beetle attack. Bark breaching occurs following branch breakage. This raises the question of how important are branch breakage, branch shedding and wound healing.

Jacobs (1955) was perhaps the first to note the branch-shedding mechanism of *Eucalyptus* in which there is in-built protection of the trunk from decay; this system is effective for branches up to about 2cm diameter in most species but perhaps up to 4cm in some cases (p.39). Clean breaks represent intrinsic shedding mechanisms; splintery breaks are the result of external factors. It seems likely that splintery breaks are ports of entry for fungi and termites, thereby leading to the formation of hollows.

When trees reach maturity, thick branches may develop which predispose the tree to develop hollows (Gibbons and Lindenmayer 1997). Natural breakage of large branches may be the first stage in the formation of substantial hollows. The more large branches the greater the chance. The lower size limit for trees to develop hollows (Gibbons and Lindenmayer 1997) could be associated with the size at which larger branches are present, break and allow hollows to develop. In the absence of fire, then, we can envisage a tree developing for many years before a hollow is possible. At tree maturity, hollows can form and persist even after the tree has died.

In the presence of fires, a number of outcomes in relation to hollows are possible and it is this variety that may be the reason why the effects of fires on hollow formation

have not been spelled out. In short, there will be a fire regime that maximizes hollow formation while there will be others that preclude it altogether.

Jarrah, like many other eucalypts, is not easily killed by a single fire even during the developing phase. However, if fires are intense and common, the shoot is killed but the plant survives as a coppice; as such it is precluded from forming trunk or branch hollows. Absence of fires or very low intensity fires would give a result similar to the unburned condition in trees up to the size of first branch breakage. If, however, a tree was exposed to fires which breached the bark by causing fire scars on the lower trunk (Gill 1974) and created patchy damage to the upper trunk then we could expect maximum chances of hollow formation to occur. To create the patchy damage on the upper trunk a fire would need to have an intensity sufficient to near kill the shoot. This would reduce subsequent growth so, for optimum hollow formation, it would occur when the tree exceeded the minimum diameter necessary for effective hollow formation.

Having been butt and trunk scarred the chances of damage to the tree from fire would be increased. Low intensity fires may allow further hollowing of the scars at the base from fire in adjacent fuel. If self-sustaining fires become possible in the exposed dead wood inside the tree then the whole tree may be destroyed by fire, the trunk collapse, and all the hollows lost. Under some circumstances only a low intensity fire is necessary to cause this ignition and collapse in trees with hollows (personal observation). In the Western Australian Regional Forest Agreement (1998, p.162) it was reported that fire was the major cause of tree fall after the hollowing out of tree butts. An alternative scenario is that the tree does not ignite yet the shoot is killed and the hollows remain until tree collapse occurs.

For the maximum development of hollows, the presence of fires appears to be necessary. Gibbons and Lindenmayer (1997) quote studies by Haseler and Taylor (1993) and Gibbons (1994) indicating that a greater proportion of fire-scarred eucalypts contained hollows than those trees that were undamaged by fires. There is no simple relationship between fire occurrences and hollow formation despite the importance of both in the functioning of the jarrah forest ecosystem. The optimum fire regime for hollows is one that causes maximum degrade in timber so is inimical to the timber industry. Forestry authorities, however, seek to retain sufficient trees with hollows during their silvicultural operations so that arboreal species persist (Abbott and Christensen 1994). Abbott (1999) refutes suggestions by other authors that hollows are limiting to bird species in south-western forests. Monitoring could resolve the issue.

#### *A floristically-rich Jarrah forest (Burrows and Friend 1998)*

Burrows and Friend (1998) describe the fire-response characteristics of plant species in three jarrah forests found in different rainfall zones. The intermediate rainfall site had a record of 222 species, 30% of which required seed for their recovery if killed by fire. It is assumed that all plants of these species would normally be killed even by low intensity fires. These species are called 'seeders'. The other 70% of species were 'resprouters' which indicates a greater resilience of mature individuals to fire intensity. Attention is focussed on the seeders as they are considered more vulnerable to frequent fires than the resprouters (Gill and Bradstock 1995).

All the species, including the seeders, were found to have flowered within two years of fire. As a rule of thumb we can consider that seed production for all these species would be adequate to re-establish a population after 4 years even if no seed is stored in the soil (Gill and Nicholls 1989). However, this estimate is open to question because it has not been verified and is outside the domain of the isolated example considered by Gill and Nicholls (1989). It is possible, though not demonstrated, that shrubby plant species with short juvenile periods (one to three years) may take longer than double the juvenile period to re-establish a suitable seed pool while those with long juvenile periods (10 to 20 years) may take less than double the juvenile period. Whatever the case may be, the juvenile periods of the jarrah species reported were, importantly, all short.

Given that no fire is possible the year following a fire, and that the chances of ignition are lower in the following few years, the resilience of the system to fires at short-intervals becomes apparent. If the species all re-establish their seed populations in 4 years, then burning at four years maintains the *status quo*. Burrows and Friend (1998) suggested that a repeated cycle of three fires at two intervals of 6 years (the first and second in spring and the third in autumn) followed by a period of 12 years without fire would be an appropriate regime for the achievement of conservation and the fire-protection of other forest values. This regime has similarities to that suggested by Christensen and Maisey (1987) for the conservation of woylies and tammars but differs in having the 12-year fire-free period and no specified high intensity fire in valleys.

#### *Fire regimes, orchids, frogs*

As far as the author is aware there is no published treatise on the effects of fire regimes on the orchids of the south-western region. However, the observations on orchids by Mr R. Heberle (personal communication, 30/2/97) over a period of 40 years within 300kms of Albany suggest that “fuel reduction burning in late spring and autumn could not be worse” [for orchids]. He notes that “most terrestrial orchids in this region appear to respond to previous summer burns in January, February and March” [my emphasis]. Such observations of an experienced and proficient observer should not be ignored but, on the other hand, should not be taken as read. They point to the need for experimental studies on the effects of fire regimes on orchids and the need to monitor orchid species as indicators of the seasonal effects of fires. Autecological studies would also be useful. Our knowledge of the effects of season of fire in general is poor and Haberle’s observations provide us with a key target for further investigation.

A similar conclusion may be reached after consideration of a small group of frogs in the southwestern area near Walpole studied by Driscoll and Roberts (1997). These frogs form the *Geocrinia rosea* complex. The public land within the entire range of two rare species of this complex – *G. alba* and *G. vitellina* – is subject to prescribed burning. They occur in “broad swamps with distinctive riparian vegetation”. In a broad scale experiment, the numbers of calling males of the representative taxon for the group, *G. lutea*, were reduced by two thirds after spring burning. This decline was considered to represent the decline in the general population at the time, not just that of the males. Unknown is what the long-term trend in the population would be. While high mortality immediately after fire has been shown, the population could still recover as breeding adults remained. If any one population was killed the loss of the

species at that site is possible because the species' have very poor dispersal. Much more data is needed if we are to know what the effects of different fire regimes may be on these frogs. Like the observations on orchids, this example highlights a potential target for further experiment and monitoring.

## Discussion and conclusions

Evaluations of the effects of fire regimes on biodiversity are often given either for the persistence of the species or for the performance of a population. A performance criterion would involve the number of individuals, the ground cover of the vegetation or the volume or biomass of a population rather than just presence or absence. Performance is often the criterion for rare species and for animals (where counts are usual as the result of trapping). For other than rare species, a persistence criterion for biodiversity management is the more appropriate for management, the persistence being seen in the light of maintained sequences of time-since-fires of various characteristics.

The woylie example strongly suggests that the ecological context of the understanding gained is important to any conclusion reached regarding fire management. In the presence of the exotic fox, the regime for woylie persistence may be different than if it was absent. What the effects on optimal fire regimes may be due to the current silvicultural context or that created by the exotic root-rotting fungus *Phytophthora* spp. are impossible to say at present. Certainly, logging affects the presence of hollows in tree populations, a major animal habitat, and this has caused an adjustment in management practice.

Our knowledge of biodiversity is relatively strong for vertebrates and vascular plants but weaker for other animals and plants. Because of this it is tempting to resort to the use of a process-based criterion for effective management rather than use one based on what is there – the system state. However, using processes as the criterion for effective management is undermined by repeated changes in management processes during even short histories of forest use (for wood products or conservation of biodiversity etc.). Fire history data reveal only glimpses of pre-historic fire regimes because they relate to fire intervals only and are not themselves unequivocal. Furthermore, we do not know what changes have occurred in biodiversity before and during white settlement so we are unable to say that changes in the fire regime have, or have not, been beneficial or detrimental to particular outcomes. Abbott (1999) illustrated the complex changes in the diversity of bird species that have taken place in the south-west region due, in part, to the interdigitation of farm and forest.

Any serious program of biodiversity management will include monitoring of both system states and significant processes so changes in one may be, or may not be, related to changes in the other. System states can be monitored following inventory, survey and the development of a comprehensive monitoring program for scientifically selected species. Present indications are that the systematic monitoring of certain orchids and frogs may be worthwhile in southwestern forests. In the monitoring of processes it is important that all the components of fire regimes be monitored in each management zone (*e.g.* Parks, Forests, Recreation areas *etc.*) and in each significant subset of these management zones (*e.g.* clear-felled areas, group-selection areas, buffer strips *etc.*).

Management is influenced and constrained by many factors. This contribution is not meant as a critique of any particular management practice in any particular regional jurisdiction. It has sought to highlight a number of instructive examples of the effects of fire regimes on elements of biodiversity irrespective of any current management activity in any agency operating in the forests of southwestern Australia. With rapidly increasing knowledge, more-intensive land management, increasing influence of public opinion on forest management, changing climate, the prevalence of exotic species and changing fire regimes, formal monitoring systems are essential if we are to know the state of our environments and how they are changing (see also Hobbs 1996).

## Acknowledgement

I would like to thank Peter H.R. Moore for his technical support during this project. Dr N. Burrows and Mr S. Lang formally reviewed the draft manuscript.

## Literature Cited

- Abbott, I. (1999). The avifauna of the forests of south-west Western Australia: changes in species composition, distribution and abundance following anthropogenic disturbance. *CALMScience Supplement 5*, 1-176.
- Abbott, I. and Christensen, P. (1994). Application of ecological and evolutionary principles to forest management in Western Australia. *Aust. For.* **57**, 109-122.
- Abbott, I. and Loneragan, O. (1986). Ecology of Jarrah (*Eucalyptus marginata*) in the Northern Jarrah Forest of Western Australia. W.A. Department of CALM Bulletin No. 1, 137p.
- Ambrose, , G.J. (1982). An Ecological and Behavioural Study of Vertebrates using Hollows in Eucalypt Branches. PhD thesis, La Trobe University, Melbourne.
- Ashton, D.H. (1976). Studies on the mycorrhizae of *Eucalyptus regnans* F.Muell. *Aust. J. Bot.* **24**, 723-741.
- Beggs, B.J. (1982). *Working Plan No. 87, Part I*. W.A. Forests Department, Perth.
- Bradshaw, F. J. and Lush, A.R. (1981). *Conservation of the Karri Forest*. Forests Department of Western Australia, Perth. 60p.
- Bradshaw, F.J. and Rayner, M.E. (1997a). Age structure of the karri forests: 1. Defining and mapping structural development stages. *Aust. For.* **60**, 178-187.
- Bradshaw, F.J. and Rayner, M.E. (1997b). Age structure of the karri forests: 2. Projections of future forest structure and implications for management. *Aust. For.* **60**, 188-195.
- Breidahl and Hewett, P.J. (1992). The Silviculture of Karri. W.A. Department of CALM Research Bulletin No. 5.
- Burrows, N.D. and Friend, G. (1998). Biological indicators of appropriate fire regimes in southwest Australian ecosystems. *Tall Timbers Fire Ecology Conference Proceedings 20*, 413-421.
- Burrows, N.D., Ward, B.G. and Robinson, A.D. (1995). Jarrah forest fire history from stem analysis and anthropological evidence. *Aust. For.* **58**, 7-16.
- Byram, G.M. (1959). Combustion of forest fuels. In: K.P. Davis (ed) *Forest Fire: Control and Use*. Pp. 61-89. McGraw Hill, New York.

- Calver, M.C. and Dell, J. (1998). Conservation status of mammals and birds in southwestern Australian forests. II. Are there unstudied, indirect or long-term links between forestry practices and species decline and extinction? *Pacific Conservation Biology* **4**, 315-325.
- Calver, M.C., Hobbs, R.J., Horwitz, P. and Main, A.R. (1996). Science, principles and forest management: a response to Abbott and Christensen. *Aust. For.* **59**, 1-6.
- Christensen, P.E.S. (1980). The Biology of *Bettongia penicillata* Gray, 1837, and *Macropus eugenii* (Desmarest, 1817) in Relation to Fire. Forests Department of Western Australia Bulletin 91.
- Christensen, P. and Annels, A. (1985). Fire in southern tall forests. In: J.R. Ford (ed.) *Fire Ecology and Management of Western Australian Ecosystems*. Pp. 67-82. Western Australian Institute of Technology, Perth.
- Christensen, P. and Maisey, K. (1987). The use of fire as a management tool in fauna conservation areas. In: D.A. Saunders, G.W. Arnold, A.A. Burbidge and A.J.M. Hopkins (eds) *Nature Conservation: the Role of Remnants of Native Vegetation*. Pp. 323-329. Surrey Beatty and Sons, Chipping Norton.
- Churchill, D.M. (1968). The distribution and prehistory of *Eucalyptus diversicolor* F.Muell., *E. marginata* Donn. ex Sm., and *E. calophylla* R.Br. in relation to rainfall. *Aust. J. Bot.* **16**, 125-151.
- Claridge, A.W. (1992). Is the relationship among mycophagous marsupials, mycorrhizal fungi and plants dependent on fire? *Aust. J. Ecol.* **17**, 223-225.
- Claridge, A.W. and May, T.W. (1994). Mycophagy among Australian mammals. *Aust. J. Ecol.* **19**, 251-275.
- Claridge, A.W., Tanton, M.T., Seebeck, J.H., Cork, S.J. and Cunningham, R.B. (1992). Establishment of ectomycorrhizae on the roots of two species of eucalypts from fungal spores contained in the faeces of the long-nosed potoroo (*Potorus tridactylus*). *Aust. J. Ecol.* **17**, 207- 217.
- Conacher, A. and Conacher, J. (2000). *Environmental Planning and Management in Australia*. Oxford University Press, South Melbourne.
- Corbett, L.K. (1985). Morphological comparisons of Australian and Thai dingoes: a reappraisal of dingo status, distribution and ancestry. *Proc. Ecol. Soc. Aust.* **13**, 277-291.
- Dell, B., Havel, J.J. and Malajczuk, N.(eds) (1989). *The Jarrah Forest*. Kluwer Academic.
- Driscoll, D.A. and Roberts, J.D. (1997). Impact of fuel-reduction burning on the frog *Geocrinia lutea* in southwest Western Australia. *Aust. J. Ecol.* **22**, 334-339.
- Gentilli, J. (1989). Climate of the jarrah forest. In: B. Dell, J.J. Havel and N. Malajczuk (eds) *The Jarrah Forest*. Pp. 23-40. Kluwer Academic.
- Gibbons, P. (1994). Management of Wildlife Habitat Trees in Forests also Managed for Wood Production. Master of Forestry Thesis, Australian National University, Canberra.
- Gibbons, P. and Lindenmayer, D.B. (1997). Conserving Hollow-dependent Fauna in Timber-production Forests. NSW National Parks and Wildlife Service, Sydney, Forest Issues 2, 110p.
- Gill, A.M. (1974). Toward an understanding of fire-scar formation: field observation and laboratory simulation. *For. Sci.* **20**, 198-205.
- Gill, A.M. (1975). Fire and the Australian flora: a review. *Aust. For.* **38**, 4-25.
- Gill, A.M. and Catling, P.C. (in press). Fire regimes and biodiversity of forested landscapes of southern Australia. In: R.A. Bradstock, J.E. Williams and A.M. Gill (eds) *Flammable Australia: the Fire Regimes and Biodiversity of a Continent*. Cambridge University Press, Cambridge, U.K.

- Gill, A.M. and Bradstock, R.A. (1995). Extinctions of biota by fires. In: R.A. Bradstock, T.D. Auld, D.A. Keith, R. Kingsford, D. Lunney and D. Sivertsen (eds) *Conserving Biodiversity: Threats and Solutions*. Pp. 309-322. Surrey Beatty and Sons, Sydney.
- Gill, A.M. and Moore, P.H.R. (1997). Contemporary Fire Regimes in the Forests of Southwestern Australia. Report for Environment Australia. CSIRO Plant Industry, Canberra.
- Gill, A.M. and Nicholls, A.O. (1989). Monitoring fire-prone flora in reserves for nature conservation. In: N. Burrows, L. McCaw and G. Friend (eds) *Fire Management on Nature Conservation Lands*. W.A. Department of CALM Occ.Pap.1/89, pp. 137-151.
- Gill, A.M., Belbin, L. and Chippendale, G.M. (1985). Phytogeography of *Eucalyptus* in Australia. Bureau of Flora and Fauna, Australian Flora and Fauna Series No. 3., 53p.
- Hallam, S.J. (1975). Fire and Hearth. A Study of Aboriginal Usage and European Usurpation in South-western Australia. Australian Aboriginal Studies No. 80. Australian Institute of Aboriginal Studies, Canberra.
- Haseler, M. and Taylor, R. (1993). Use of tree hollows by birds in sclerophyll forest in north-eastern Tasmania. *Tasforests*, October, pp. 51-56.
- Havel, J.J. (1968). The potential of the northern Swan coastal plain for *Pinus pinaster* Ait. plantations. W.A. Forests Department Bulletin No. 76.
- Havel, J.J. (1975a). Site-vegetation mapping in the northern jarrah forest (Darling Range) 1. Definition of the site-vegetation types. W.A. Forests Department Bulletin No. 86.
- Havel, J.J. (1975b). Site-vegetation mapping in the northern jarrah forest (Darling Range) 2. Location and mapping of the site-vegetation types. W.A. Forests Department Bulletin No. 87.
- Hobbs, R.J. (1996). Ecosystem dynamics and management in relation to conservation in forest systems. *J. Roy. Soc. W.A.* **79**, 293-300.
- Hopper, S.D., Keighery, G.J. and Wardell-Johnson, G. (1992). Flora of the karri forest and other communities in the Warren Botanical subdistrict of Western Australia. W.A. Department of CALM Occ. Pap. 2/92, 1-32.
- Jacobs, M.R. (1955). *Growth Habits of the Eucalypts*. Commonwealth of Australia, Forestry and Timber Bureau, Canberra.
- Johnson, C.N. (1997). Fire and habitat management for the mycophagous marsupial, the Tasmanian bettong *Bettongia gaimardi*. *Aust. J. Ecol.* **22**, 101-105.
- Kessell, S.L. (1929). First Decennial Review of the Operations of the Forests Department together with the Annual Report of the Conservator of Forests for the year ended June 30th, 1929. W.A. Forests Department, Perth.
- Lamont, B.B. (1995). Interdependence of woody plants, higher fungi and small marsupials in the context of fire. *CALMScience Supplement 4*, 151-158.
- Lang, S. (1997). Burning the Bush. A Spatio-temporal Analysis of Jarrah Forest Fire Regimes. BSc (Hons) Thesis, Australian National University.
- McCarthy, M.A., Gill, A.M. and Lindenmayer, D.B. (1999). Fire regimes in mountain ash forest: evidence from forest age structure, extinction models and wildlife habitat. *For. Ecol. and Manag.* **124**, 193-203.
- McCarthy, M.A., Gill, A.M. and Bradstock, R.A. (in press) Theoretical fire-interval distributions. *Int. J. Wildl. Fire*
- Milewski, A.V. (1979). A climatic basis for the study of the convergence of vegetation structure in mediterranean Australia and southern Africa. *J. Biogeog.* **6**, 293-299.

- Mulcahy, M.J. (1973). Landforms and soils of southwestern Australia. *J. Roy. Soc. W.A.* **56**, 16-22.
- Perry, D.H., Lenz, M. and Watson, J.A.L. (1984). Relationships between fire, fungal rots and termite damage in Australian forest trees. *Aust. For.* **48**, 46-53.
- Ride, W.D.L. (1970). *A Guide to the Native Mammals of Australia*. Oxford University Press, Melbourne.
- Robinson, R. (1999). The effect of wildfire on the fruiting of macrofungi in karri regrowth forests in the south-west of Western Australia. [A note.] *Australasian Mycologist* **18** (2), 39.
- Smith, G.T. (1985a). The Noisy Scrub Bird *Atrichornis clamosus*. Does it's past suggest a future? In: A. Keast, H.F. Recher, H. Ford and D. Saunders (eds) *Birds in Eucalypt Forests and Woodlands: Ecology, Conservation and Management*. Pp. 301-308. Royal Australian Ornithologists Union and Surrey Beatty and Sons, Chipping Norton.
- Smith, G.T. (1985b). Fire effects on populations of the Noisy Scrub Bird (*Atrichornis clamosus*), Western Bristlebird (*Dasyornis longirostris*) and Western Whipbird (*Psophodes nigrogularis*). In : J.R. Ford (ed) *Symposium on 'Fire Ecology and Management in Western Australian Ecosystems'*. Pp. 95-102. Western Australian Institute of Technology, Perth.
- Smith, G.T. (1985c). Population and habitat selection of the Noisy Scrub Bird, *Atrichornis clamosus*, 1962-83. *Aust. Wildl. Res.* **12**, 479-485.
- Smith, G.T. (1994). Fire and rare species conservation in coastal southwest Australia. *J. fur Ornithologie* **135**, 492.
- Start, A.N., Burbidge, A.A. and Armstrong, D. (1998). A review of the conservation status of the woylie, *Bettongia penicillata ogilbyi* (Marsupialia: Potoroidae) using IUCN criteria. *CALMScience* **2**(4), 277-289.
- Start, T., Burbidge, A., Sinclair, E. and Wayne, A. (1995). Lost and found: Gilbert's Potoroo. *Landscape* **10**(3), 28-33.
- Strelin, G.J. (1988). Site classification in the southern jarrah forest of Western Australia. W.A. Department of CALM Research Bulletin No. 2, 71 p.
- Taylor, R.J. (1991). Plants, fungi and bettongs: a fire-dependent co-evolutionary relationship. *Aust. J. Ecol.* **16**, 409-411.
- Warcup, J. H. (1981). Effect of fire on the soil micro flora and other non-vascular plants. In: A.M. Gill, R.H. Groves and I.R. Noble (eds) *Fire and the Australian Biota*. Pp. 203-214. Australian Academy of Science, Canberra.
- Vines, R.G. (1968). The forest fire problem in Australia – a survey of past attitudes and modern practice. *Aust. Sci. Teachers J.* **14**, 5-16.
- Ward, D. (2000). *Trouble in the Tuart. A Brief Fire History*. Western Australian Department of Conservation and Land Management, Perth.
- Ward, D. and van Didden, G. (1998). *Reconstructing the Fire History of the Jarrah Forest of South-western Australia*. Report to Environment Australia, Canberra.
- Wardell-Johnson, G. and Williams, M. (1996). A floristic survey of the tingle mosaic, south-western Australia: applications in land use planning and management. *J. Roy. Soc. W.A.* **79**, 249-276.
- Western Australian Regional Forest Agreement, South-west Forest Region (1998). Western Australian Comprehensive Regional Assessment Report - Volume 1.