

## C H A P T E R F I V E

# Sustaining Key Old-Growth Characteristics in Native Forests Used for Wood Production: Retention of Trees with Hollows

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### **ABSTRACT**

Many old-growth forests are characterised by an abundance of hollow-bearing trees. Hollows represent an important habitat resource to a wealth of fauna. In Australia alone, the number of vertebrate species that utilise hollows has been suggested to be near 400. Taxa known to occupy southern temperate eucalypt forests are listed.

Timber harvesting in natural forests represents a potential threat to hollow-dependent fauna. A number of studies have demonstrated that populations of hollow-dependent species are reduced or eliminated upon the removal of all or a portion of the hollow resource. Logging cycles or rotation lengths are also typically shorter than the period required for trees to form hollows suitable for occupation by wildlife.

Since the management of Australia's native forests is guided by the principles of multiple-use and ecologically sustainable development, prescriptions have been introduced in an attempt to minimise the impacts of timber harvesting on hollow-dependent fauna. Current prescriptions are suggested in this chapter to be sub-optimal for a number of reasons. It is recommended that 'prescriptions' be formulated for a specific site and its suite of fauna rather than across broad areas. Goals need to be formulated against which the performance of habitat tree prescriptions can be routinely monitored. In the absence of sufficient resource data, a more precautionary approach to management is warranted. Greater emphasis needs to be given in management strategies to the optimal distribution and perpetual recruitment of hollow-bearing trees. A number of changes to current silvicultural practice are suggested.

### **INTRODUCTION**

An abundance of hollow-bearing trees is cited as a characteristic of many old-growth forests. That tree-hollows have long been a prominent attribute of the Australian landscape is reflected in the degree to which they have acted as a selection pressure. In this country alone it has been suggested that nearly 400 vertebrate species utilise the resource (Ambrose 1982), many on an obligate basis.

Timber harvesting in Australia's southern temperate eucalypt forests represents a potential threat to hollow-dependent species. Cutting cycles or rotation lengths are typically shorter (30–150 years) (Victorian Government 1986; Richards *et al.* 1990; Van Saane and Gordon 1991) than the period required for trees to form cavities suitable for occupation by wildlife. As part of moves towards multiple-use management, measures have been introduced to retain hollow-bearing trees for wildlife purposes. The issue is most critical in clearfelled areas where a small proportion of the stand's total basal area is retained. Prescriptions for these forests specify the retention of hollow-bearing trees (sometimes including potential hollow-bearing trees) at a rate of between less than one tree to ten trees per hectare, depending on location (Taylor 1990; Department of Conservation and Environment 1992; Department of Conservation and Land Management 1992; Forestry Commission of New South Wales 1992). However, the ecological sustainability of these practices in many forests has been questioned (see for example, Recher *et al.* 1980; Australian Biological Research Group 1984; Davey and Norton 1990; Lindenmayer *et al.* 1990b; 1993; Scotts 1991; Norton and May, Chapter 2).

Management of hollow-bearing trees in timber production forest is a complex issue. Dependent fauna have different requirements with respect to habitat therefore complicating the optimum number and configuration of trees that should be retained after harvesting. Maintaining target numbers and spatial arrangements of hollow-bearing trees is further complicated both by their fickle longevity and variable recruitment rates, particularly under conditions created following logging. This chapter discusses the factors bearing on the management of hollow-dependent fauna in southern temperate eucalypt forests available for timber production. Some reference will also be made to those taxa that rely on other features provided by such trees: decaying timber, loose bark and dead limbs. A critique is given on some aspects of current practice and a number of measures are identified which have the potential to improve the sustainability of current practice for hollow-dependent fauna.

### THE HOLLOW-FORMATION PROCESS

Hollow formation is almost invariably facilitated by the process of decay (Shigo 1979; Wilkes 1982a). Wounds or stress essentially predispose a tree to attack by decay-causing organisms (Greaves and Florence 1966; Shigo 1979). Shigo (1979) defined a wound as 'a break in the protective bark that exposes the xylem', although Manion and Zabel (1979) suggested that sapwood wounds are the most common point of entry for decay-causing organisms. The process of decay begins with discolouration of the affected timber, initially as a result of a chemical response by cells (production of phenols), but later as a reaction to invading pioneer organisms such as bacteria and nonhymenomycetous (non-decay) fungi (Shigo 1979; Wilkes 1982a). The invasion of the wood by hymenomycetous (decay) fungi will usually follow and if the wood is not effectively occluded a succession of other decay and non-decay organisms, facilitated by a changing substrate, are likely to follow, including fungi, bacteria, protozoa, nematodes, insects and vertebrates (Shigo 1979; Wilkes 1982a). The decay process varies with many factors. Shigo (1979) noted that the rate of decay is affected by the severity of the wound, its position, size, time of the year, wounding agent and the agents of infection; the severity of the wound is usually positively correlated with the degree of decay. The decay process has also been suggested to be affected by species and wood durability (Jacobs 1955; Ruyooka and Griffin 1980; Marks 1982; Wilkes 1982a, 1982b) and health of the tree (Greaves and Florence 1966; Wilkes 1982a). The suggestion

that hollow formation ceases in dead eucalypts (Calder *et al.* 1983) is not consistent with observations of the effects of fungi, termites, fire, wind and vertebrates on hollow-formation and development (Saunders 1979; Saunders *et al.* 1982; Calder *et al.* 1983; Inions 1985; Smith and Lindenmayer 1988; Inions *et al.* 1989).

In the absence of primary excavating vertebrates such as woodpeckers (Piciformes), the process of hollow formation in Australian forests is largely dependent upon the action of invertebrates. Hollow formation in Blackbutt (*Eucalyptus pilularis*) as described by Mackowski (1984, 1987) is a good illustration of this process. It begins with wood discolouration in the branches which is consistent with the early stages of decay described by Shigo (1979). In the vanguard of the hollow-forming process are termites (*Coptotermes* spp.), galleries of which are first detected within the stems of trees of about 75 cm diameter at breast height (dbh). At 100 cm dbh trees may harbour termite nests at 1–2 metres above ground level, with workings extending to branches within the crown. Noting the presence of wood fungi, Mackowski (1984) suggested their contribution to hollow-formation in Blackbutt is negligible other than predisposing the tree to attack by termites. Termites transport soil (mudguts) into most parts of the tree. Branches broken-off and drained of mudguts represent hollows suitable for arboreal marsupials. The age of the tree at this stage is estimated to be between 144 and 194 years. Mudguts subsequently provide habitat to a range of wood-eating beetles of the families Lucanidae, Passalidae and Dynastidae, which play a role in continued hollow formation.

Fire has been noted as an agent of both hollow formation and destruction (Cowley 1971; Ashton 1975; Ambrose 1982; Inions 1985; Inions *et al.* 1989). Intense burning may cause a shortage of natural hollows (Calder *et al.* 1983; Inions *et al.* 1989), but has also been observed to facilitate hollow formation in small trees (Inions *et al.* 1989). Fire has also been observed to increase the number of dead, hollow-bearing stags in a stand, which may persist for a considerable time after regeneration has become established (Franklin *et al.* 1981; Smith and Lindenmayer 1988). Inions *et al.* (1989) noted an increase in the occupancy of hollows by the Common Brushtail Possum after fire, although they were unable to determine whether this was in response to an alteration of hollow configuration or attraction to epicormic growth. Notably the total number of hollows observed were considerably reduced compared with those present prior to the fire. Fire damage to the butt has been noted to kill parts of the crown of a tree and thus facilitate the decay process (Rundel 1962; Ambrose 1982). A significantly greater proportion of fire-scarred eucalypts have been shown to bear hollows compared with undamaged trees (Taylor and Haseler 1993; Gibbons 1994).

Hollows most commonly occur in older trees. Jacobs (1955) noted that as trees mature, the ratio of crown size to the surface area of the bole decreases and thus growth increment or the sheath of new wood also decreases or becomes progressively thinner. It follows that the rate at which a tree is able to occlude a wound to the stem is reduced. Jacobs also suggested that the rate of decay overtakes the rate of growth as the tree gets older. One may additionally suggest that the increased mass of heartwood with age provides a resource that can support larger populations of decay-causing organisms. Hollow formation occurs when trees develop large branches, which represent too great an area to occlude when they eventually break. Hollow formation is therefore exacerbated as branch diameter and length increases, which tends to be greatest in wide-spaced stands. This observation suggests potential for some manipulation of hollow formation. Ambrose (1982) noted that trees with weak apical dominance, and therefore a heavy branching habit, such as River Red Gum (*Eucalyptus camaldulensis*), produce the most hollows, with up to 30 observed in a single tree.

TABLE 1: Age at which various species of eucalypt begin to form hollows.

Species	Location	Age (years)	dbh (cm)	Source
<i>E. camaldulensis</i>	Victoria (various locations)	35*	48	1
<i>E. cypellocarpa</i>	Central Highlands, Vic	135	63	1
<i>E. obliqua</i>	Central Highlands, Vic	110	80	1
<i>E. regnans</i>	Central Highlands, Vic	120	127	1
<i>E. pilularis</i>	North East NSW Coast	144–194	120	2
<i>E. fastigata</i>	Bathurst, NSW	50*	50*	3
<i>E. viminalis</i>	Bathurst, NSW	50*	50*	3
<i>E. dalrympleana</i>	Bathurst, NSW	50*	50*	3
<i>E. macrorhyncha</i>	Bathurst, NSW	100*	50*	3
<i>E. dives</i>	Bathurst, NSW	100*	50*	3
<i>E. pauciflora</i>	Bathurst, NSW	100*	50*	3
<i>E. salmonophloia</i>	South West WA	130	70	4
<i>E. marginata</i>	Manjimup, WA	50*	20*	5
<i>E. sieberi</i>	East Gippsland, Vic	110*	64	6

\* See text for qualification.

#### Sources:

1. Ambrose 1982

4. Saunders *et al.* 1982

2. Mackowski 1984

5. Inions *et al.* 1989

3. Disney and Stokes 1976

6. Gibbons 1994

Estimates of the age at which eucalypts begin to develop hollows vary greatly with species (Table 1), although some of these estimates must be interpreted with caution. Disney and Stokes (1976), for instance, noted that most hollow-bearing trees they observed were at least 500 mm dbh and suggested that these species can increase the diameter of their trunks by 10 mm per annum, therefore concluding that trees of this size would be of at least 50 years of age. They add, 'In dry sclerophyll forest of *E. macrorhyncha*, *E. dives* and *E. pauciflora* it would probably take 100 years to achieve this diameter.' No data are offered to support these assumptions. Ambrose (1982) provided more rigorous estimates of the age at which hollow-formation commences in selected Victorian eucalypts. Mountain Ash (*E. regnans*) was estimated to begin producing hollows at approximately 120 years of age, increasing in number until about 570 years, after which hollow availability decreases as the tree declines. Most hollow-bearing branches, he estimated, are lost by 700 years of age, although cavities and fissures in the bole are suitable habitat to a number of arboreal marsupials in this forest type (Smith and Lindenmayer 1988; Lindenmayer *et al.* 1990a; 1991). Hollow formation was estimated to commence in Messmate (*E. obliqua*) at 110 years, peaking at 430 years; in Mountain Grey Gum (*E. cypellocarpa*) at 135 years, peaking at 730 years; and in River Red Gum (*E. camaldulensis*) commencing at about 35 years, but not increasing markedly until about 120 years and peaking at 185 years (Ambrose 1982). Calder *et al.* (1983), in studies conducted within dry sclerophyll forest of Central Victoria, found very few hollows in trees below 50 cm dbh and of the five hollows found in smaller trees, three were in dead individuals. The authors noted that Messmate developed hollows after a considerably long period relative to Candlebark (*E. rubida*), Manna Gum (*E. viminalis*),

Peppermint (*E. radiata* and *E. dives*), Long-leaved Box (*E. goniocalyx*) and Red Box (*E. polyanthemos*). In studies conducted in Western Australia, Inions *et al.* (1989) estimated the youngest den tree being studied — a Jarrah (*E. marginata*) — to be 50 years of age, although it was noted that the mean age of Jarrah den trees was at least 230 years. The data presented indicated that the mean age of trees with hollows varied considerably with species, site and fire history. The smallest Silvertop Ash recorded to contain hollows in a study conducted in lowland sclerophyll forest of East Gippsland, Victoria (Gibbons 1994) had a dbh of 64 cm. Applying a size-age relationship developed for Silvertop Ash in south-east NSW (Banks 1990; Richards *et al.* 1990), a tree of this size would be approximately 110 years old, although this relationship was developed at a dry sclerophyll forest site at a much higher elevation. The age at which a tree produces hollows must not be mistaken for the age at which a tree begins to produce hollows suitable for occupation by wildlife. The two may be very different. The great diversity of hollow-dependent taxa is at least paralleled by the number of factors that determine the suitability of hollows as a habitat resource.

## HOLLOW-DEPENDENT FAUNA

If overseas literature can be taken as a guide, hollows can be expected to be important habitat to a range of vertebrate and invertebrate taxa in Australian forests. In a study of natural and artificial cavity use in mid-south forest habitats of the United States, 12% of the total species recorded were mammals, 16% birds, 9% reptiles, 5% amphibians and 58% invertebrates (McComb and Noble 1981, 1982).

### Mammals

Arboreal marsupials are the most prominent hollow-dependent mammal of Australia's southern temperate forests. Thirteen species are known to occupy hollows (Table 2), and most of these are obligate hollow-nesters. The larger species such as the Yellow-bellied Glider (*Petaurus australis*), Sugar Glider (*P. breviceps*), Greater Glider (*Petaurodes volans*), Leadbeater's Possum (*Gymnobelideus leadbeateri*) and Common and Mountain Brushtail Possums (*Trichosurus vulpecula* and *T. caninus*) have been subject to considerable research. Less information has been assembled for the inherently more difficult species to study such as the Pygmy Possums (*Cercartetus* spp.) and Feathertail Glider (*Acrobates pygmaeus*).

Scansorial mammalian fauna that utilise tree-hollows in Australian forests are little-studied by comparison. In North America this group includes conspicuous species such as the Raccoon (*Procyon lotor*), Spotted Skunk (*Mephitis mephitis*) and Black Bear (*Ursus americanus*) (Thomas *et al.* 1979; Rogers *et al.* 1988). In Australia the group is somewhat more cryptic. Known species are listed in Table 2. Dickman (1991) presented evidence that hollow use by some scansorial mammals is not simply opportunistic. He suggested that the influence of hollows as a selection pressure on Yellow-footed Antechinus (*Antechinus flavipes*), Brown Antechinus (*A. stuartii*) and Red-tailed Phascogale (*Phascogale calura*) can be evidenced in their breeding biology. These species deposit newborn in the nest at a much younger and hence more vulnerable age than log- or ground-nesting species such as Dusky Antechinus (*A. swainsonii*), Swamp Antechinus (*A. minimus*) and Dibbler (*Parantechinus apicalis*).

TABLE 2: Mammalian fauna known to utilise tree-hollows in southern temperate eucalypt forests of Australia  
(Source: Ambrose 1982; Strahan 1983; Australian Biological Research Group 1984; Tidemann and Flavel 1987; Dickman 1991).

#### Arboreal marsupial fauna

Brush-tailed Phascogale	<i>Phascogale tapoatafa</i>
Common Ringtail Possum	<i>Pseudochirus peregrinus</i>
Greater Glider	<i>Petauroides volans</i>
Yellow-bellied Glider	<i>Petaurus australis</i>
Sugar Glider	<i>P. breviceps</i>
Squirrel Glider	<i>P. norfolkensis</i>
Leadbeater's Possum	<i>Gymnobelideus leadbeateri</i>
Common Brushtail Possum	<i>Trichosurus vulpecula</i>
Mountain Brushtail Possum	<i>T. caninus</i>
Eastern Pygmy-possum	<i>Cercartetus nanus</i>
Western Pygmy-possum	<i>C. concinnus</i>
Little Pygmy-possum	<i>C. lepidus</i>
Feathertail Glider	<i>Acrobates pygmaeus</i>

#### Scansorial mammalian fauna

Spotted-tailed Quoll	<i>Dasyurus maculatus</i>
Western Quoll	<i>D. geoffroii</i>
Yellow-footed Antechinus	<i>Antechinus flavipes</i>
Brown Antechinus	<i>A. stuartii</i>
Dusky Antechinus	<i>A. swainsonii</i>
White-footed Dunnart	<i>Sminthopsis leucopus</i>
Grey-bellied Dunnart	<i>S. griseoventer</i>
Little Long-tailed Dunnart	<i>S. dolichura</i>
Bush Rat	<i>Rattus fuscipes</i>

#### Microchiropteran bats

Yellow-bellied Sheathtail-bat	<i>Taphozous flaviventris</i>
White-striped Mastiff-bat	<i>Tadarida australis</i>
Little Northern Mastiff-bat	<i>Mormopterus loriae</i>
Eastern Little Mastiff-bat	<i>M. norfolkensis</i>
Greater Long-eared Bat	<i>Nyctophilus timoriensis</i>
Gould's Long-eared Bat	<i>N. gouldi</i>
Lesser Long-eared Bat	<i>N. geoffroyi</i>
Gould's Wattled Bat	<i>Chalinolobus gouldii</i>
Chocolate Wattled Bat	<i>C. morio</i>
Greater Broad-nosed Bat	<i>Nycticeius rueppellii</i>
Little Broad-nosed Bat	<i>N. greyii</i>
Great Pipistrelle	<i>Pipistrellus tasmaniensis</i>
Little Forest Eptesicus	<i>Eptesicus vulturinus</i>
Large Forest Eptesicus	<i>E. sagittula</i>
Little Brown Bat	<i>E. pumilus</i>
King River Eptesicus	<i>E. regulus</i>

Approximately half of Australia's Microchiropteran bat fauna are known to roost and/or breed in tree-hollows (Strahan 1983; see Table 2). In southern Australian temperate forests, these species use hollows during periods of torpor (Tidemann and Flavel 1987). A number of Microchiropteran bats are also known to roost behind peeling bark, large resources of which are associated with older trees of certain species.

#### Birds

A rich avian fauna also utilise hollows throughout the world. The best-known taxon is probably the woodpeckers (Order Piciformes), which number about 200 species (Hunter 1990). There are no representatives of this order in Australia, but there is a diverse avian fauna that utilise hollows nevertheless. Saunders *et al.* (1982) listed 94 Australian hollow-nesting species, compared with 55 in Southern Africa and 58 in Northern America. Avian fauna that nest in southern temperate forests are listed in Table 3 and are dominated by members of the Order Psittaciformes (parrots). A number of other bird species also utilise the dead or decaying timber of standing trees as foraging or nesting substrate. Sittellas (*Daphoenositta* spp.) are obligate dead-wood nesters, whilst flycatchers (*Myiagra* spp.) and cuckoo shrikes (*Coracina* spp.) nest on horizontal, dead branches and the Grey Shrike-thrush (*Colluricinclla harmonica*) and Buff-rumped Thornbill (*Acanthiza reguloides*) behind loose bark of large, old trees (Recher 1991). Recher (1991) also cited a number of species that use dead wood as a major foraging substrate, viz. Orange-winged Sitella (*D. chrysoptera*), Black-capped Sitella (*D. pileata*) and White-throated Treecreeper (*Climacteris leucophaea*). Dead timber has also been noted as an important hawking platform to a number of predatory birds such as robins (*Petroica* spp.), cuckoos (*Cuculus* spp.) and raptors (Slater *et al.* 1986; Recher 1991; Hollands 1993).

TABLE 3: Avian fauna known to utilise tree-hollows in southern temperate eucalypt forests of Australia  
(Source: Ambrose 1982; Saunders *et al.* 1982; Calder *et al.* 1983; Australian Biological Research Group 1984; Slater *et al.* 1986; Recher 1991; Hollands 1993; Taylor and Haseler 1993; S. Pell pers. comm.).

Australian Shelduck	<i>Tadorna tadornoides</i>
Pacific Black Duck	<i>Anas superciliosa</i>
Australian Grey Teal	<i>Anas gracilis</i>
Wood Duck	<i>Chenonetta jubata</i>
Nankeen Kestrel	<i>Falco cenchroides</i>
Peregrine Falcon	<i>F. peregrinus</i>
Gang Gang Cockatoo	<i>Callocephalon fimbriatum</i>
Galah	<i>Cacatua roseicapilla</i>
Sulphur-crested Cockatoo	<i>C. galerita</i>
Yellow-tailed Black Cockatoo	<i>Calyptorhynchus funereus</i>
Long-billed Black Cockatoo	<i>C. baudinii</i>

TABLE 2 *continued*

Carnaby's Black Cockatoo	<i>C. latirostris</i>
Glossy Black Cockatoo	<i>C. lathami</i>
Rainbow Lorikeet	<i>Trichoglossus haematodus</i>
Scaly-breasted Lorikeet	<i>T. chlorolepidotus</i>
Swift Parrot	<i>Lathamus discolor</i>
Musk Lorikeet	<i>Glossopsitta concinna</i>
Little Lorikeet	<i>G. pusilla</i>
Australian King Parrot	<i>Alisterus scapularis</i>
Ringneck	<i>Platycercus zonarius</i>
Red-capped Parrot	<i>Purpureicephalus spurius</i>
Regent Parrot	<i>Polytelis anthopeplus</i>
Eastern Rosella	<i>Platycercus eximius</i>
Western Rosella	<i>P. icterotis</i>
Green Rosella	<i>P. caledonicus</i>
Blue-cheeked Rosella	<i>P. elegans</i>
Blue-winged Parrot	<i>Neophema chrysostoma</i>
Laughing Kookaburra	<i>Dacelo novaeguineae</i>
Sacred Kingfisher	<i>Halcyon sancta</i>
Dollarbird	<i>Eurystomus orientalis</i>
Masked Owl	<i>Tyto novaehollandiae</i>
Sooty Owl	<i>T. tenebricosa</i>
Southern Boobook	<i>Ninox novaeseelandiae</i>
Powerful Owl	<i>N. strenua</i>
Australian Owlet Nightjar	<i>Aegotheles cristatus</i>
Tree Martin	<i>Cecropis nigricans</i>
Australian Ground Thrush	<i>Zoothera lunulata</i>
Scarlet Robin	<i>Petroica multicolor</i>
Grey Shrike-thrush	<i>Colluricincla harmonica</i>
Buff-rumped Thornbill	<i>Acanthiza reguloides</i>
White-throated Treecreeper	<i>Cormobates leucophaea</i>
Red-browed Treecreeper	<i>Climacteris erythrops</i>
Rufous Treecreeper	<i>C. rufa</i>
Brown Treecreeper	<i>C. picumnus</i>
Striated Pardalote	<i>Pardalotus striatus</i>
Dusky Woodswallow	<i>Artamus cyanopterus</i>
*Common Myna	<i>Acridotheres tristis</i>
*Tree Sparrow	<i>Passer montanus</i>
*House Sparrow	<i>P. domesticus</i>
*Common Starling	<i>Sturnus vulgaris</i>

\* Denotes introduced species

### Herpetofauna

Very little information exists on the use of cavities and other features of habitat trees by herpetofauna. It has been suggested that 185 species of frog (order Aneura), lizard (sub order Lacertilia) and snake (sub order Sepentes) utilise tree-hollows in Australia, with 67 species occurring in forested environments (Ambrose 1982). Mackowski (1987) observed reptiles in a number of hollow-bearing trees which were felled and dissected. Dead, standing trees have been suggested as important basking sites for heliothermic reptiles, especially in cool temperate forests (Brown and Nelson 1992).

### Invertebrates

As with herpetofauna, there is a dearth of information on the use of cavities by invertebrates. The number of species that rely, at least in part, on decaying trees is unknown, but undoubtedly high. In Britain, Elton (1966) noted that 974 vertebrate and invertebrate species are known to live in wood or under bark where decay has commenced. Such diversity of dependent organisms is not surprising when one considers the quantity of decaying timber available in many forested ecosystems. Maser *et al.* (1979) noted that decaying logs in western Douglas-fir forests represented more volume than the total above-ground woody biomass of a typical deciduous forest of the east coast. In Swiss National Forest approximately one third of the estimated standing timber is dying or dead (Elton 1966). McComb and Noble (1982) noted the use of natural and artificial cavities by 39 invertebrate species. The use of hollows by Cave Crickets (*Ceuthophilus* spp.) in the same study raises interesting questions with regard to potential hollow-dependent fauna. As previously discussed, invertebrates are also important in facilitating hollow-formation.

### FACTORS AFFECTING THE OCCUPANCY OF HOLLOW BY WILDLIFE

A range of factors affect the suitability of hollows as habitat for wildlife. Some species appear to occupy only those hollows that meet rather specific criteria relating to factors such as size, orientation and surrounding vegetation. For some species their hollow requirements may be linked to physiology — perhaps for thermoregulatory reasons. The behavioural ecology of other species largely determines the availability of hollows. The next two sections explore the habitat requirements of hollow-dependent taxa as they relate to hollows. The latter section refers to habitat requirements as they relate specifically to the effects of timber harvesting.

Many species show a preference for hollow-bearing trees at a certain stage of decay. In forests of Oregon and Washington State, for example, certain species of Microchiropteran bats favour dead trees with loose bark for roosting. Standing dead trees at later stages of decay were most frequently used by excavating woodpeckers, presumably because of softer wood. Horton and Mannan (1988), in studies conducted within Ponderosa Pine (*Pinus ponderosa*) forest, found woodpeckers to favour trees in a different decay class to the Pygmy Nuthatch (*Sitta pygmaea*), Western Bluebird (*Sialia mexicana*) and Violet-green Tree Swallow (*Tachycineta thalassina*). In Australia, Smith and Lindenmayer (1988) observed preference by Leadbeater's Possum for living hollow-bearing trees with dead tops. Russell (1984) observed that Yellow-bellied Gliders line their dens with twigs and leaves broken from live branches, which are carried back to the den with the tail, thus disabling the individual from gliding. He suggests that this behaviour prompts the species to select live — as opposed to dead — den trees.

Species						
PAAT	14	black-capped chickadee	●			10.2 (4)
DEWI	13	hairy woodpecker		●		25.4 (10)
CEFA	14	brown creeper		●		25.4 (10)
EUAM	15	yellow pine chipmunk	●			25.4 (10)
FASP	14	American kestrel		●		30.5 (12)
OTFL	14	flammulated owl	●			30.5 (12)
SICA	13	white breasted nuthatch		●		30.5 (12)
LANO	14	silver-haired bat		●		30.5 (12)
MUFR	15	long-tailed weasel	●			30.5 (12)
PRLO	14	raccoon	●			50.8 (20)
MAPE	14	fisher			●	50.8 (20)
SPPU	15	spotted skunk	●			50.8 (20)
Letter code	Life form	$\geq 1.8 (\geq 6)$ $\geq 4.6 (\geq 15)$ $\geq 9.1 (\geq 30)$ $\geq 9.5 (\geq 31)$ Minimum nesting height, in metres (feet)				
		Minimum d.b.h. required for nesting in cm. (in.)				
	KP					

FIGURE 1: Minimum diameter at breast height (dbh) and nesting heights of trees used by a sample of cavity-nesting species in the Blue Mountains of Washington and Oregon (modified from Taylor *et al.* 1979).

The overall dimensions of hollow-bearing trees has also been demonstrated as important for some species. Taylor *et al.* (1979) listed the minimum dbh of snags utilised by 39 bird and 23 mammal species in the Blue Mountains of the Pacific Northwest (Figure 1). In a study conducted in California, cavity-nesting birds showed preference for trees with an average dbh of 62 cm, compared with an average dbh of 32 cm for the entire stand (Raphael and White 1984).

The size of the tree may be important because it could feasibly correlate with the size and insulation properties of the hollows produced. A number of mammal and bird species show preference for hollows with entrance dimensions of minimal size (McComb and Noble 1981; Tidemann and Flavel 1987). Cavities with entrances of minimal dimensions are likely to be more effective at excluding intra- and inter-specific competitors and predators and may possess thermoregulatory benefits (Ambrose 1982). Saunders *et al.* (1982) found

that, in addition to the size of the hollow entrance, internal dimensions were also important in determining occupation by a number of species of cockatoo (Order Psittaciformes). The depth of the hollow, rather than entrance dimensions, was the most important feature determining occupation by the Common Brushtail and Ringtail Possums (Inions *et al.* 1989). A number of species, such as the Sooty Owl (*Tyto tenebricosa*) (Schodde and Mason 1980), roosting and/or maternity colonies of some forest bats and family groups of the Yellow-bellied Glider (Henry and Craig 1984), will occupy only large, deep hollows that form in older eucalypts.

The location of the hollow in the tree is another consideration. Taylor *et al.* (1979) provided the minimum observed nesting height for hollow-nesting species in the Blue Mountains of Washington and Oregon (Figure 1). Similar differences were recorded for various species inhabiting hardwood forests of Louisiana and Mississippi (McComb and Noble 1981). Equivalent data has been reported for relatively fewer Australian species. In studies of hollow requirements for Common Brushtail Possum, Winter (1976 cited in Green 1984) found that the species selected hollows at least 6 metres above the ground with an opening of at least 10 cm diameter. Smith and Lindenmayer (1988) plotted nest entrance height and hollow type against the number of recorded Leadbeater's Possum colonies. Preferences have been shown to vary even between closely related species. The White-throated Treecreeper (*Climacteris leucophaea*) has been observed to select cavities mainly in tree trunks whilst the Red-browed Treecreeper (*C. erythrops*) appears to prefer a sloping, dead hollow spout with an upward facing entrance (Noske 1985; Recher 1991). Owen and Thompson (1965) reported that Common Brushtail Possum selected hollow spouts and trunks of live or dead eucalypts whereas Mountain Brushtail Possum preferred hollow fallen logs or stumps close to the ground.

The number of entrances to the hollow have also been found to be important for some species. Saunders *et al.* (1982) found that the Red-tailed Black Cockatoo showed preference for hollows with more than one entrance. The number of holes was found to be significant in describing the occupancy of a tree by arboreal marsupials in the Montane Ash forests of Victoria (Lindenmayer *et al.* 1990a).

It has been suggested that some species show preference for occupying hollows in certain tree species over others. Sanderson (1975) noted various species as optimum habitat for Grey Squirrels (*Sciurus carolinensis*) on the basis of being good den or mast producers. Taylor *et al.* (1979) suggested that woodpeckers favour certain tree species for excavating nests, but added that this relationship was non-obligate. In Western Australia, Saunders *et al.* (1982) found that Salmon Gums (*Eucalyptus salmonophloia*), which represented 74% of the total number of trees, provided 95% of the available hollows of a suitable size for cockatoos to nest. Kehl and Borsboom (1984) found that 49% of den trees selected by the Greater Glider were in Broad-leaved White Mahogany (*E. umbra*) despite the species representing only 22% of the total area, although 44% of total hollows found in the study area occurred in the same species. Forest Red Gum (*E. tereticornis*), as noted by the same authors, was used more frequently as a den tree (16%) than the total availability of hollows for the species (7%) would suggest. Taylor and Haseler (1993) similarly noted that Manna Gum (*E. viminalis*) was used as a nest tree by four hollow-nesting bird species in Tasmania to a greater extent than expected from its relative abundance. One cannot determine from the above data whether hollow-dependent fauna show a preference for certain species simply because of their propensity to develop suitable hollows or whether the preference is a function of other factors. The readiness with which a number of vertebrate species utilise artificial hollows (McComb and Noble 1981; Suckling and Macfarlane 1983; Menkhorst 1984) suggests the

former. Indeed, Inions *et al.* (1989) found that selection of tree hollows by the Common Brushtail and Ringtail Possums was related to internal hollow dimensions rather than tree species.

### SOME EFFECTS OF TIMBER HARVESTING ON HOLLOW-DEPENDENT FAUNA

Results from a number of studies suggest *a priori* that the hollow resource in undisturbed environments may be in excess. Only 9.5% of tree-hollows were found to be utilised by vertebrates in natural forests and plantations in mid-south forests of the United States (McComb and Noble 1981). In certain Netherlands forests 54–93% of all cavities were occupied by hole-nesting birds (Van Balen *et al.* 1982). Studies of Southern Flying Squirrels in South Carolina found that this species occupied 36% of total cavities (Carmichael and Guynn 1983). Tyndale-Biscoe and Calaby (1975) and Smith and Lindenmayer (1988) found that approximately one third of potential nest trees were occupied by arboreal marsupials. Saunders *et al.* (1982) recorded 47% of examined hollows to be occupied by nesting parrots in spring. It is dangerous, however, to equate this apparent largesse with the number of hollows that can be safely removed from a forest as part of a logging operation without adversely affecting dependent fauna.

Many studies have found a positive relationship between the number of hollows and the diversity and abundance of hollow-dependent fauna. A stark illustration of this relationship was demonstrated in a study conducted by Scott (1979) who censused an area of Ponderosa Pine (*Pinus ponderosa*) in Arizona for avian fauna then re-visited the area after all snags had been removed. Up to a 52% decline in cavity-nesting birds was recorded after removal of the snag population. Thomas *et al.* (1979) predicted that an increasing number of woodpeckers will be supported by higher snag densities with 100 per cent of the population supported at a snag density of between 32 and 741 per 100 ha, depending on the species. The population density of hole-nesting birds in Douglas-fir forest was found to be positively correlated with increasing stand age, which was in turn positively correlated with density of snags with cavities (Horton and Mannan 1988). By providing additional artificial cavities to an area of Scandinavian forest, Lennerstedt (1983 cited in Hunter 1990) detected a 16-fold increase in Pied Flycatchers (*Ficedula hypoleuca*).

Similar results have been obtained in Australia. In a study comparing bird populations in native eucalypt forest and a Monterey Pine (*Pinus radiata*) plantation, Disney and Stokes (1976), concluded that the significantly fewer breeding species in the latter was largely attributed to the dearth of hollows. In a study comparing bird populations in logged and unlogged native forests of south-east NSW, Kavanagh *et al.* (1985) found avian species most immediately affected by logging to include those that forage extensively among bark substrates and require hollows for nesting. Although 78% of the original complement of avian fauna had returned to harvested areas four years after logging, many were in low numbers. Loyn (1980) found that species relying on habitats provided by mature forest were still unable to occupy 40 year logging regeneration. In studies of arboreal marsupials in the Central Highlands of Victoria, a strong positive relationship was found between the abundance of hollow-bearing trees and possum density (Smith and Lindenmayer 1988), with no asymptote even when the number of 'potential nest trees' was 30 per 3 ha (Lindenmayer *et al.* 1990c). Notably, the models predicted negative densities of Leadbeater's Possum for sites with less than 4 potential nest trees per 3 ha or less than 1.6, 0.25-hectare blocks with potential nest

trees (Smith and Lindenmayer 1988). From data collected in East Gippsland, Victoria, Loyn (1993) concluded that bird abundance and species richness were 'approximately equal' in 25–40 year-old regrowth to mature forest. Notably, however, birds grouped as medium or large hole-nesters and small hole-nesters were significantly more abundant in mature forest. It was also noted that emergent, old trees were an essential part of the habitat for a number of species inhabiting logging regrowth.

An unoccupied hollow does not necessarily equate with an unused hollow. McComb and Noble (1981), in research conducted in mid-south forest habitats of Mississippi and Louisiana, found amphibian use of cavities was highest in the summer and early autumn, mammal use in late autumn and winter and avian use in the spring and early summer, indicating that use by different taxa may overlap at certain times of the year. This would not be detected unless observations were made in this period. Davey (1989) suggested that hollows with a northern aspect may be utilised more in winter, and those with a southern aspect in summer. In a study conducted in south-east Queensland, most Greater Gliders were recorded using three or four different dens per month, with from 4 to 18 separate dens being utilised by individuals over a 10 month period (Kehl and Borsboom 1984). Henry and Craig (1984) and Russell (1984) similarly reported Yellow-bellied Gliders to change den trees frequently. One family group was observed to use 13 different den trees; 5 in one 7 day period.

The occupation of multiple hollows may, in part, be a strategy to help defend the home range. Territoriality is common amongst hollow-dependent taxa and this behaviour exacerbates the effects of timber harvesting for a number of reasons. Mean home ranges for arboreal marsupials occurring in temperate forests of Australia range from 1 to 50 ha, with most species occupying an area of between 1 and 5 ha (Lacey *et al.* 1990). Home ranges show minimal overlap between individuals of a number of these species, including: Yellow-bellied Glider (Henry and Craig 1984; Russell 1984); Sugar Glider (Henry and Suckling 1984); Greater Glider (Henry 1984; Kehl and Borsboom 1984); Leadbeater's Possum (Smith 1984); and Common Brushtail Possum (Green 1984). Data collected by Lindenmayer *et al.* (1990a) also suggested interspecific defence of the home range by a number of arboreal marsupials. Less than 1% of 1125 trees observed to contain hollows were found to be occupied by more than one species. Territorial species will not generally recolonise new areas because of a high site-attachment and/or active defence of adjacent home ranges by other individuals (Tyndale-Biscoe and Smith 1969; How 1972 cited in Tyndale-Biscoe and Calaby 1975; Goldingay and Kavanagh 1991). Territorial arboreal mammals are consequently eliminated from clearfelled areas (Recher *et al.* 1980). Individuals will only survive if sufficient unlogged forest (including hollows) are retained in their home range (Tyndale-Biscoe and Smith 1969; Recher *et al.* 1980; Goldingay and Kavanagh 1991). Territorial behaviour is also exhibited by a number of hollow-nesting birds. Large forest owls in particular have highlighted conflicts between timber harvesting and wildlife conservation (SEIS 1993). Forest owls are reliant upon trees with hollows not only as nesting sites, but also because, for some species, their major food source is arboreal marsupials. It has been suggested that the Powerful Owl (*Ninox strenua*) may consume 250–300 Greater Gliders annually (Tilley 1982). Forest owls also occupy large territories; Davey (1989) suggested between 400–1450 ha. As with arboreal marsupials, forest owls will only survive logging if sufficient habitat is retained within their home range.

It is true that some species will utilise alternative resources in the absence, or following removal, of tree hollows. On Barrow Island, Western Australia, the Common Brushtail Possum has been observed to nest in limestone solution pipes and termite mounds (Allen and Bradshaw 1978). In forested habitats the same species has been observed to nest in

logs, rabbit warrens and rock crevices (Kean 1967; Green 1973). In some areas of New Zealand, where the Common Brushtail Possum is introduced, 60–70% of nest sites have been observed at ground level (Green 1973). It has been suggested, however, that this behaviour leaves the individual more vulnerable to predation (Kerle 1984). A number of taxa have been observed to nest or roost in buildings, including various Microchiropteran bats (Tidemann and Flavel 1987), the Common Brushtail Possum, Starlings and the Common Mynah (*Acridotheres tristis*). Artificial nest boxes are also readily occupied. McComb and Noble (1981; 1982) recorded their use by 12 species of mammal, bird, reptile and amphibian and 39 invertebrate taxa. The Sugar Glider was successfully reintroduced to planted eucalypt forest in western Victoria with the use of nest boxes (Suckling and MacFarlane 1983). Nest-box use has also been recorded for a number of other Australian bird and mammal species (see for example Ambrose 1982; Calder *et al.* 1983; Menkhorst 1984).

One would be guilty of *reductio ad absurdum* attempting to explain the impacts of timber harvesting in terms of the loss of hollow-bearing trees alone. A number of hollow-dependent taxa have other rather specific habitat requirements that inhibit them from surviving in or recolonising intensively logged areas. Their omission here is not a reflection of their significance.

## IMPLICATIONS FOR MANAGEMENT

It has been established that hollows are an important habitat resource to a wealth of forest fauna. The conservation of such taxa within native forests available for timber production is facilitated in part by prescriptions that call for the retention of hollow-bearing trees (in part, because protected areas also contribute to the conservation of hollow-dependent fauna in timber production areas). Current prescriptions frequently detail little more than the number of trees to be retained. The following section examines this issue, and a number of others, that must be addressed in order to arrive at an ecologically sustainable approach to the conservation of hollow-dependent taxa.

### Aims and principles of habitat tree management

The aims of tree-hollow prescriptions are rarely reported by forest management agencies. Australia's National Forest Policy (Commonwealth of Australia 1992) states that public forests must be managed to retain biological diversity, this reflecting a principle tenet of wildlife conservation (see for example, Ehrlich and Ehrlich 1981). Biological diversity is about managing for all species and communities, not simply those which are either prominent or for which there are substantial data. Since there is no forest in Australia for which such information exists, effective management demands some exercise of precaution, one of the principles of ecologically sustainable development (World Commission on Environment and Development 1987).

Given the diversity of hollow-dependent fauna in Australia and their range of habitat requirements, effective application of the precautionary principle would logically see diversity in the retained hollow and potential hollow resource. Subsequent discussion will elaborate on the diversity required, but it is clear from the data assembled in this chapter that, from a wildlife conservation perspective, maintenance of the maximum number of hollow-bearing trees and recruits in the logged area will be of most benefit.

Biological diversity can be managed within different spatial and temporal scales. For instance, habitat tree prescriptions can be formulated in an attempt to maintain biological diversity of dependent fauna on the logged area. Current management of some of the production forests of north-east New South Wales, in which light selection logging is practised, may be such an example. Alternatively, a temporary loss of dependent species over the harvested area might be accepted, but conditions created such that the time taken to recolonise the area is minimised. Habitat tree prescriptions applied in clearfelled environments should be hoping to achieve at least the latter.

Without explicit objectives it is difficult to formulate prescriptions and impossible to set measures against which the success or otherwise of these can be judged. Indeed, monitoring is standard practice with regard to other aspects of silviculture, e.g. regeneration, and should also apply to wildlife.

### Which trees to retain?

As discussed previously, hollow-dependent fauna are diverse and it follows that so are their habitat requirements. For this reason a generic definition of 'habitat tree' is singularly inappropriate. Diversity could be reflected on any given site by retained trees that vary in terms of factors such as species, size, age, health and location, particularly where information is lacking on the composition and specific habitat requirements of resident fauna in the area.

For reasons of both immediate and future habitat potential, the retained habitat tree resource must consist of trees representing a range of age/health classes. A number of arboreal species show preference for older, more senescent trees, possibly due to a correlation between tree age and hollow characteristics. Older trees tend to bear the largest and multiple hollows, which have been demonstrated as important determinants for occupation by some arboreal mammal and bird species (see for example, Saunders *et al.* 1982; Kehl and Borsboom 1984; Mackowski 1984; Lindenmayer *et al.* 1990a). Dead branches, which are a characteristic of old or over-mature trees, are also important nesting and hawking sites for a number of bird species (Recher 1991; Hollands 1993). Despite many species showing a preference for more senescent individuals, live trees are often preferred to dead individuals (Russell 1984; Lindenmayer *et al.* 1990a). The retention of live trees is also consistent with maximising their longevity (Lindenmayer *et al.* 1990b). Nevertheless, dead trees have been shown to be utilised by a range of forest fauna (both when standing and fallen) and, therefore, should be considered a cohort in the habitat tree resource. They also do not represent competition to regeneration.

Healthy, mature trees (with or without hollows) also represent important habitat to forest fauna that cannot be substituted by logging regrowth. The observation that a number of species choose hollows with an entrance only slightly larger than themselves (see for example, Tidemann and Flavel 1987) suggests that 'young' hollows provided by such trees may be an important resource to a whole suite of species. Mature trees, unlike regrowth, shed large strips of bark, and represent nesting and/or roosting sites for some Microchiropteran bat species and birds, habitat to an unknown range of insects and foraging substrate to a number of amphibians, reptiles, birds and mammals. Further, a large number of species will only nest and/or feed in larger trees. Thomas *et al.* (1979) and Hunter (1990) suggested that large trees should be favoured because they can generally provide a range of habitat resources that smaller trees cannot. Nevertheless younger trees still represent suitable feeding substrate to a number of species and also represent future hollow-bearing trees.

It is often suggested that many animals show a preference for particular tree species. This is certainly true in a dietary sense. Folivorous species are selective, as are species that feed on exuded gums. Some trees are prone to heavy flowering and seeding, providing important food resources for certain taxa. Trees that shed strips of bark also represent habitat and feeding substrate to a range of fauna. The relationship between occupancy of hollows by forest fauna and tree species is somewhat more tenuous. Nevertheless, it appears that some tree species more readily produce hollows than others and some are more inclined to produce hollows of certain dimensions. Larger hollows, for instance, are often found in trees with a heavy branching habit, e.g. River Red Gum (Ambrose 1982). Conversely, some species, eg Red Stringybark (*Eucalyptus macrorhyncha*), are rarely observed to bear hollows (Australian Biological Research Group 1984). A number of authors list species that should be favoured over others when selecting habitat trees (see for example, Australian Biological Research Group 1984; Department of Conservation, Forests and Lands undated). Indeed, where the existence of hollows can be combined with identified feed trees, this is desirable, but it must be acknowledged that the diets of many species change throughout the year. Generally a diversity of species should therefore be represented in the retained habitat tree resource.

#### **How many trees?**

Existing prescriptions for the numbers of habitat trees that should be retained in Australian timber production forest appear to either lack foundation or are loosely based on the requirements of one or a number of arboreal mammal species only. For instance, the prescription of 3 hollow-bearing trees per ha adopted in the Kempsey/Wauchope management area (Truyard Pty. Ltd. 1993) seem to draw only from data obtained by Mackowski (1984) for Common Brushtail Possum, Yellow-bellied Glider and Greater Glider populations. Mackowski's figures are not a panacea. Certainly, they do not allow for observations that a number of arboreal marsupials occupy more than one hollow (Henry and Craig 1984; Kehl and Borsboom 1984; Russell 1984) and may defend vacant hollows from occupancy by other species (Lindenmayer *et al.* 1990a). Similarly, the rate of habitat tree retention adopted in the Otway Forest Management Plan (Department of Conservation and Environment 1992) draws only from data collected for arboreal marsupial fauna by Smith and Lindenmayer (1988) and Lindenmayer *et al.* (1990a; 1990b; 1991). Many other prescriptions seem to lack any scientific base to their formulation.

It is true that data in Australia for the habitat requirements of taxa other than arboreal marsupials are insufficient. Nevertheless, in the knowledge that arboreal marsupials account for around 20% of the total vertebrate fauna that utilise hollows (Ambrose 1982), a number of taxa only occupy hollows for a certain period and a number of species occupy multiple hollows within their home range, one might expect that a figure of tree retention formulated on the basis of this taxon alone would be far from adequate for any area of forest. The considerable data on arboreal marsupial species (e.g. diet, maximum gliding distances, territories and number of hollows occupied within these) should, therefore, be utilised only as the starting point for formulating habitat tree prescriptions for a given area of forest.

#### **Distribution**

The distribution of habitat trees is also an important consideration. For species that rely on gliding to cover large distances (e.g. Yellow-bellied Glider) it would seem logical that an even distribution of tall trees would be necessary for them to recolonise logging regrowth.

Should the periodic occupation of hollows throughout the individual's territory be an aid to successful defence of the home range, as suggested by Kehl and Borsboom (1984), then a fairly uniform distribution of retained trees would be optimal. Inter- and intra-specific defence of a territory by a species (see for example, Lindenmayer *et al.* 1990a) may also result in utilisation of an unevenly distributed hollow resource by a low diversity of species. Leaving large areas free of hollow-bearing trees additionally has the potential of fragmenting populations, thereby disrupting metapopulation dynamics. Indeed, Lindenmayer *et al.* (1992) found that the more evenly hollow-bearing trees were distributed throughout the ash-type forests of the Central Highlands of Victoria, the greater the probability of occurrence of arboreal marsupials. The effectiveness of prescriptions may be reduced where a preference is given to clumping habitat trees in a sparse arrangement. The Forest Commission of Tasmania, for instance, prescribed the retention of clumps of trees every 4–6 ha (Taylor 1991), which could see a distance of 500 m between habitat trees, a distance that is far too great for a number of arboreal marsupials. Similarly, the prescription of 5 trees per 15 ha, preferably clumped, applied in parts of Victoria and NSW (Department of Conservation, Forests and Lands 1986; Forestry Commission of NSW 1992) could theoretically be applied in the field as one clump, containing 5 habitat trees, every 15 ha.

Habitat trees retained as part of a group of trees of varying ages have been found to support a greater diversity of bird species than solitary trees (Loyn 1980). Furthermore, hollow-nesting birds do not usually nest in solitary trees retained after logging (Loyn *et al.* 1980). The quantity of vegetation surrounding the den tree was an important factor determining occupancy by Leadbeater's Possum (Lindenmayer *et al.* 1991). Retaining hollow-bearing trees as part of a clump is also consistent with maximising their longevity (see below). Retaining habitat trees as part of a clump of trees is therefore desirable, in order to either retain hollow-dependent fauna on the logged area, or minimise the period before recolonisation. However, clumps must be evenly distributed at a distance consistent with the requirements of resident fauna.

#### **Recruitment**

Ensuring continuity in the hollow resource over time is critical to effective forest wildlife management. The future of the endangered Leadbeater's Possum is believed to be uncertain because of an impending temporary dearth of hollow-bearing trees (Lindenmayer *et al.* 1990b). Indeed, this could be the case in many forests with a long history of even-aged silviculture. Perpetual recruitment of hollow-bearing trees is difficult to guarantee in the light of current knowledge. It has yet to be determined how to identify a tree at an early age that is likely to develop hollows, although certain species are known to more readily form cavities than others, as are trees with a heavy branching habit (Jacobs 1955; Ambrose 1982). Ensuring recruitment is also confounded in the knowledge that the hollow-formation process differs with species — and possibly environments — and is considerably influenced by episodic events such as fire.

Prescriptions aimed at encouraging continuity in the hollow resource place little emphasis on ensuring that retained trees persist for the period required for them to either be occupied by wildlife or develop to become the next cohort of hollow-bearing trees. Indeed, there is evidence to suggest that the environment to which habitat trees are exposed following logging, particularly clearfelling, is antagonistic to their persistence (Australian Biological Research Group 1984; Lindenmayer *et al.* 1990b; Morrison and Raphael 1993).

Trees retained in clearfelled coupes are usually exposed to a medium or high intensity regeneration burn, may have suffered a degree of crown or butt damage during the logging operation, are no longer afforded protection from wind by surrounding vegetation, and represent a target for parasites. Indeed, these factors have been shown to predispose a tree to collapse. Fire is capable of affecting wood properties (McArthur 1968; Nicholls 1974; Nicholls and Cheney 1974) and is also a vector for invasion by decay-causing organisms (Shigo 1979; Manion and Zabel 1979). Both factors are believed to reduce a tree's standing life (Fraser 1962; Putz *et al.* 1983). Hot burns, as often conducted in clearfelling operations, may kill retained trees, dead trees being more susceptible to collapse than live ones (Lindenmayer *et al.* 1990b). Wind velocity is also increased when the density of a forest canopy is reduced or as the cleared area increases (Alexander 1964; Miller *et al.* 1991; Department of Conservation and Natural Resources, unpublished data). This translates to an increased rate of windthrow relative to uncut forest (Raphael and Morrison 1987; Chen *et al.* 1992). Exposure has also been associated with increased parasite attack which can weaken retained trees (Raphael and Morrison 1987).

Studies conducted in East Gippsland, Victoria, suggest that those trees that are healthy and sound, have not suffered butt damage as a result of fire, are not located on the steepest slopes, are closer to uncut bush or are immediately surrounded by other living trees, are those most likely to persist (Gibbons 1994). Similar results have been found in other forest types. Lindenmayer *et al.* (1990b) reported that trees at a later stage of senescence were more likely to collapse. Morrison and Raphael (1993), in studies conducted in California, found that trees burnt at their base, regardless of species, were particularly susceptible to windthrow. North American literature also suggests that the incidence of windthrow on the margins of coupes is reduced where there are fewer abrupt changes to coupe boundaries and boundaries are of shorter length (Gordon 1973; DeWalle 1983). Smaller coupes also tend to experience comparatively reduced wind velocities than larger coupes (Alexander 1964). Savill (1983) cited a number of European studies that suggested that windthrow to trees retained on a coupe edge may be reduced by creating a gradually rising canopy zone and retaining an understorey. These principles may also be applied to trees retained in clumps.

The chances of maintaining a suitable hollow resource in perpetuity are therefore likely to be maximised with the retention of a large number of trees that are of a range of ages and form classes grading from young and healthy to old and senescent. As discussed previously, dead trees should also be retained. However, the fact that dead trees do not represent habitat to all hollow-dependent species and their longevity is less than for live trees (Lindenmayer *et al.* 1990b), suggests that they should not be relied upon as a long-term resource. Successful recruitment of hollow-bearing trees also necessitates a silvicultural system that maintains an uneven-aged forest structure, such as selection or modified clearfelling with a high rate of basal area retention. Silvicultural systems that employ an intense slash burn for regeneration purposes should be abandoned. Trees should also be retained as clumps or adjacent to coupe boundaries. An attempt should be made to design clumps to reduce windthrow. The fact that the complete habitat tree resource (including recruits) must be constantly represented by a stagger of trees of all ages (including dead trees), and afforded a fair degree of protection, indicates that they must be managed largely independently of the timber resource.

#### **Should wildlife management on logging coupes be pursued?**

Forests available for timber production (State Forests) are recognised as necessary partners to the conservation reserve system (Davey and Norton 1990; Commonwealth of Australia

1992). As a consequence, we see the multiple-use concept underpinning the management of Australia's state forests. A multiple-use paradigm can be translated on the ground in a number of ways.

One extreme would see areas of forest zoned separately, with one or more objectives pursued within each zone. In any single area of forest this approach may result in one area managed specifically for water production, another for flora and fauna conservation, one for wood production and one for recreation. The area zoned specifically for timber production would be managed to maximise timber yield. Silviculture may therefore be intensive and standardised for operational efficiency. Minimal regard would be given to other forest values and therefore prescriptions to retain habitat trees would not be implemented. The area(s) zoned for wildlife conservation may be strategically sited and maintained in a condition that is optimal for biodiversity conservation.

The other extreme demands a more integrated pursuit of objectives. With respect to wildlife conservation and timber production, intensive silviculture might only be implemented on sites deemed to be unimportant for wildlife conservation, perhaps well represented elsewhere and containing species not considered to be under any threat. In areas identified as important for wildlife, silvicultural systems might be formulated to retain structural features of the forest that will either continue to support certain populations or enable rapid recolonisation following harvesting. A system of retained habitat will aim to ensure that a stronghold remains for species affected adversely by disturbance, will provide a pool from which recolonisation can be initiated and avert fragmentation of populations.

The first approach to multiple-use management is criticised by Recher *et al.* (1987) on the basis that it is unlikely to sample the full range of genetic diversity of forest fauna within the planning unit, and it would fragment populations and hinder recolonisation of logged or burnt forest. Davey (1989) suggested that systems based on zoning tend to be inflexible and unable to accommodate temporal change in biological systems. Moreover, if an area managed to maximise wood productivity can no longer revert to suitable wildlife habitat then the opportunity to amend land-use decisions is permanently lost. However, the concessional approach could potentially result in less area required for equivalent timber output and therefore potentially more area available for wildlife conservation; the trade-off being that most wildlife would be totally excluded from the former. Within the bounds of current knowledge one can only speculate as to whether this represents a better scenario for the conservation of biodiversity. The concessional approach, whilst still demanding a high degree of strategic planning, would be potentially easier to implement on the ground, negating the need to formulate complex measures such as habitat tree prescriptions. The *ex situ* effects of single-use timber production zones, however, may be potentially deleterious should they involve, for instance, applications of herbicides and fertilisers and/or use of exotic species.

Indeed it is the second approach — that which demands a more integrated pursuit of objectives — which seems to more truly embrace the multiple-use concept. It has the potential, in theory at least, of maintaining a high degree of structural and biological diversity over the greater forest area. However, success is contingent upon effective planning at two levels: at the regional level, where strategic areas for wildlife management need to be identified, as do areas suitable for harvesting, and issues such as connectivity investigated; and at the harvesting unit or coupe level, where silviculture, including measures such as habitat tree retention, need to be formulated for any given site on the basis of resident fauna, rather than prescribed over a greater area. Successful implementation of such a

strategy relies upon a substantive resource base and considerable commitment of field staff. The rewards, however, will include a better understanding of forest processes. If this approach (which is currently being pursued in production forests of Australia) continues to fall short of reaching its potential in terms of wildlife conservation, then the concessional approach to multiple-use management — and possibly a move away from retaining hollow-bearing trees on logging coupes — may be an option worth considering.

### Research directions

Any attempt to ameliorate the impacts of timber harvesting on Australia's hollow-dependent fauna is thwarted by a dearth of resource information. This is partly reflected in the rather loose prescriptions laid-down for habitat tree management in Australia's production forests. An inventory of hollow-dependent fauna must be given some priority. Considerable data already exist, but need collation. The specific requirements of hollow-dependent fauna, along the lines of work conducted by Inions (1985), Mackowski (1984), Saunders *et al.* (1982) and Lindenmayer *et al.* (1990a), needs to be pursued for a range of taxa. North America is well ahead of Australia in this regard and work reported by authors such as Thomas *et al.* (1979) are useful guides. Relationships between different taxa also need to be examined. The number and configuration of hollows required to conserve faunal communities have yet to be ascertained (Norton and May Chapter 2).

With respect to the habitat resource, considerable research is required on habitat tree dynamics. Decay transition rates and hollow ontogeny need to be modelled for a range of species and forest types in order to plan for the future hollow resource. Further work on the longevity of retained trees in logged environments within Australian forests, and appropriate silviculture, also need to be pursued.

A significant improvement in the resource base would come about if forest management agencies began monitoring their activities. Simple data, such as: which species return and within what time frame for different configurations of retained trees, would be information that would be relatively easy to collect. Trials of different habitat tree prescriptions are warranted and scope exists for data from different forest types and silvicultural systems from around the country to be assembled and compared.

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CHAPTER SIX

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## Sustaining Sensitive Wildlife Within Temperate Forest Landscapes: Regional Systems of Retained Habitat as a Planning Framework

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### ABSTRACT

Certain forest fauna are vulnerable to habitat modification, fragmentation and loss brought about by logging, burning and grazing activities. Current forest reserve systems appear inadequate to provide for the long-term conservation of these species. Forests logged, burnt and grazed outside the formal reserve system are done so under generalised and inadequate mitigation procedures. There is an urgent need for forest wildlife habitat management guidelines and prescriptions based on ecological principles and existing knowledge of forest systems. Regional Systems of Retained Habitat are recommended as a sound framework for rational long-term management of forest wildlife habitats.

### FOREST WILDLIFE CONSERVATION — A NEW EMPHASIS FOR FOREST PLANNERS AND MANAGERS

Forest management and planning in the 1990s and onwards is destined to become a benchmark in the assessment of our generation's ability to pass on an ecologically sustainable future to our descendants. In Australia, perhaps more than anywhere else, we have the opportunity to set precedents and trends in this sphere of land management that will act as standards for rational and responsible forest management in other countries.

With ever increasing environmental education and concern the public, timber industry and governments will continue to expect improved standards from forest planners and managers. In order to satisfy these demands, forest managers and planners must be, and be seen to be, capable of a new, holistic approach to land management.

This approach is encapsulated in one word, biodiversity. The term, biodiversity, has gained popularity within the media, government circles, academic institutions and consequently within public forums. But what is 'biodiversity'?

The concept of biodiversity is one that recognises the role and intrinsic right to existence for all living things and their environments as part of viable natural ecosystems. Conservation of biodiversity involves a regional approach to land management and strives to maintain