Management of tree hollows in the jarrah *Eucalyptus marginata* forest of Western Australia

Kim Whitford[1] and Geoff Stoneman[2]

[1]Science Division, Department of Conservation and Land Management, Dwellingup, WA 6213, Australia

email: kimwh@calm.wa.gov.au

[2]Sustainable Forest Management Division, Department of Conservation and Land Management, Kensington, WA, 6983, Australia

ABSTRACT

This paper examines the management of hollows in trees in the jarrah forest. In the first part of this paper we present a framework for development of strategies for the retention of hollow-bearing trees at the stand scale, and the information available for application of this process to the jarrah forest is reviewed. Whilst significant information in relation to many elements of the framework is available, there is currently insufficient information to conclusively determine the effectiveness of the strategies adopted for management of hollows in the jarrah forest. In the second part of the paper, we discuss the suite of strategies that are employed across spatial scales to conserve hollows in the jarrah forest. The strategies include: the establishment of formal conservation reserves and informal reserves; the introduction of fauna habitat zones dispersed throughout the forest available for harvest; maintenance of connectivity, landscape heterogeneity, and stand structural complexity within the harvest area matrix, through silvicultural constraints such as limits on the size of gaps, prescribing minimum basal area to retain, and the retention of marked habitat trees, large marri trees and unmerchantable trees on harvested stands. These strategies reduce the risk to hollow-dependent fauna from timber harvesting.

Introduction

The loss of hollows that form in the branches and boles of trees is a consequence of timber harvesting and a challenge for ecologically sustainable forest management. Hollows are slow to develop in trees and are generally assumed to be a characteristic of old trees. Because of the long time periods required for recruitment of hollow-bearing trees, managing this attribute in forests that are used for timber production requires careful planning across spatial scales.

This paper deals with the jarrah *Eucalyptus marginata* forest of south-west Western Australia. Jarrah forest typically occurs as a mixture of jarrah and marri *Corymbia calophylla* trees with the marri component increasing from a low proportion in the north to a moderate proportion in the south. Wandoo trees *E. wandoo* occur in the eastern part of jarrah forest and wandoo woodland previously dominated the vegetation of the Wheatbelt bioregion, now largely cleared for agriculture, to the east of the jarrah forest.

The climate in the region is mediterranean, with cool wet winters and hot dry summers (rainfall 635 mm to 1300 mm) (Gentilli 1989). Soils are mostly sandy gravels overlying clays (Churchward and Dimmock 1989), are nutrient deficient (Grove *et al*. 1986), and the soil profile is deep and highly weathered (Churchward and Dimmock 1989).

Though comparable in many ways with the forests of eastern Australia, there are important differences between these forests which impact on the management of hollows in the jarrah forest. The jarrah forest lacks the rich assemblage of arboreal mammals of the forests of eastern Australia, having only three large hollow-dependent arboreal mammals (Abbott and Whitford 2002). The jarrah forest consists of two major overstorey tree species (jarrah and marri) and four minor overstorey tree species (wandoo, yarri *E. patens*, bullich *E. megacarpa* and flooded gum *E. rudis*) that bear hollows. The forest is in a largely contiguous form with relatively limited fragmentation by clearing for agriculture and thus provides a spatially extensive forest mosaic through which species can disperse (Commonwealth and W.A. RFA Steering Committee 1998, map 1; Conservation Commission 2004, Maps 2 and 3). Mammal distributions are restricted, and vertebrate fauna populations over large areas of the forest are low (de Tores 1999) as are rates of occupancy of hollows (Whitford and Williams 2002, Whitford 2002). Broadscale fox-baiting is practised in the jarrah forest and the response of fauna populations to fox baiting (Kinnear *et al*. 1998, de Tores 1999, Burrows and Christensen 2002, Friend and Thomas 2003, Morris *et al*. 2003) has highlighted the impact of fox predation on fauna in the jarrah forest.

While native mammals are absent or their numbers are low, studying hollow use by each arboreal mammal is expensive and slow in providing the knowledge required to manage hollows. Instead a body of research unique to the jarrah forest focuses on the availability, recruitment, loss, and management of hollows in trees and logs (Faunt 1992; McComb *et al*. 1994, Williams and Faunt 1997; Whitford and Williams 2001; Whitford 2002; Whitford and Williams 2002). A key element of the approach used by these researchers has been to thoroughly measure the relevant dimensions of hollows, use these dimensions to determine which hollows are potentially suited to various species, and examine the relationships between hollows of these various size classes and the attributes of the

Pp 807 - 829 in the *Conservation of Australia's Forest Fauna* (second edition) 2004, edited by Daniel Lunney. Royal Zoological Society of New South Wales, Mosman, NSW, Australia.

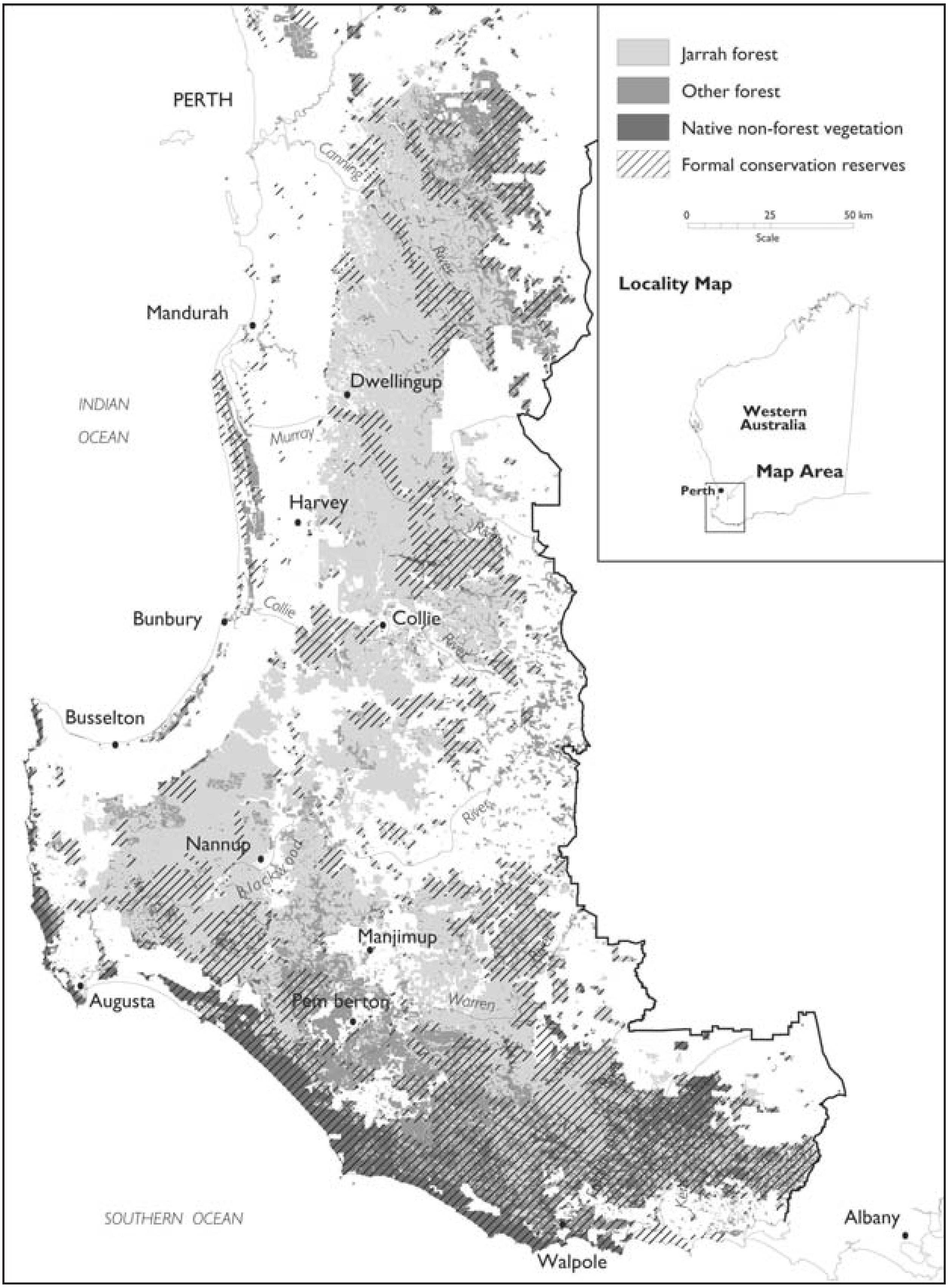

Figure 1. Forests on land managed by the Department of Conservation and Land Management (CALM) in the south-west of Western Australia. The map shows the 1.57 million hectares of CALM managed jarrah *Eucalyptus marginata* forest, and other forests (karri, wandoo, tuart and the introduced pine forests) and the area of lands in existing and proposed formal reserves (Conservation Commission 2004). Formal reserves include nature reserves, national parks and conservation parks, and CLM Act 5(1)(g) and 5(1)(h) reserves. Forest conservation areas are also shown as formal reserves for the purpose of this figure. Informal reserves and fauna habitat zones are not shown on the map.

trees bearing these hollows. This approach has enabled research on hollows in stands where hollow-dependent fauna are absent or their numbers are low. Combined with the ecological studies of hollow-using species, these studies of hollows and trees provide information that is widely applicable in developing management strategies for hollows in the jarrah forest.

Scale as a context for the development of management strategies for hollow-bearing trees is a focus of this paper. In the first part of this paper we present a framework for development of strategies for the retention of hollow-bearing trees at the stand scale, and the information available for application of this process to the jarrah forest is reviewed. The number of hollows provided by the application of current silvicultural guidelines (CALM 1995) is estimated, as is the number of hollows existing on jarrah stands. In the second part of the paper we discuss the strategies adopted in the jarrah forest to manage habitat and tree hollows in particular, at a range of scales from the stand, to the landscape, and the whole forest. The strategies used to provide connectivity, landscape heterogeneity and stand structural complexity are discussed (Lindenmayer and Franklin 2002). Research priorities related to the management of fauna habitat in the jarrah forest are outlined. The paper is focused on the jarrah forest managed by the Department of Conservation and Land Management (CALM), in particular the portion available for timber harvesting. The impacts of activities in the jarrah forest such as bauxite mining, coal mining, inundation for water supply and beekeeping are not directly addressed in this paper.

Management of Jarrah Forest

Jarrah forest originally covered some 2.78 million hectares of which 1.81 million hectares remain and 1.57 million hectares is managed by the Department of Conservation and Land Management (Conservation Commission 2004)(Fig.1). Land use in the jarrah forest depends on land category. State forest and timber reserves are multiple-use land categories that allow for water and timber production, mining, conservation, recreation and other uses. Informal reserves within State forest are not available for timber harvesting. Formal conservation reserves are managed for conservation of natural and cultural values. Management of private property and other Crown reserves with jarrah forest is variable and depends on the gazetted purpose of the Crown reserve or the purpose determined by the land-owner.

Under the Forest Management Plan 2004-2013 (Conservation Commission 2004), the jarrah forest managed by the Department of Conservation and Land Management is allocated to the following categories:

- 51.0 percent is State forest and timber reserves and is available for timber harvesting;
- 8.5 percent is informal reserves in State forest and not available for timber harvesting; and
- 40.5 percent is formal conservation reserves (Fig. 1).

Management of the jarrah forest has changed substantially in recent decades with an increased emphasis on management for biodiversity and a reduced emphasis on production of timber (Forests Department 1982, CALM 1987a,b,c, CALM 1994, Commonwealth of Australia and State of Western Australia 1999). At the time this chapter was finalized (December 2003) the current Forest Management Plan 1994-2003 was about to expire. The Forest Management Plan 2004-2013, proposes further major increases in forest reservation and substantial changes to forest management practices to favour biodiversity conservation. The guidelines for silvicultural practice in the jarrah forest are also under review and the changes provide for increased emphasis on diversity and habitat in areas subject to timber harvesting (Conservation Commission 2004). This paper has been prepared based on the policies and practices of the Forest Management Plan 2004-2013 and the draft guidelines for silvicultural practices in jarrah forest.

Spatial scale

Although scale is continuous, forest management can be considered at particular scales. The scale for managing hollows ranges from the individual tree, a group of trees, the harvest coupe (~ 1000 ha), a forest block (~ 5000 ha), a group of forest blocks, and ultimately the whole-of-forest scale (1.8 M ha). Depending on the species of animal, these scales loosely correspond to an individual hollow, a home range, a local population, several populations, metapopulations, and the distribution of a species. For simplicity and clarity we use three terms to refer to scale: stand; landscape; and whole-of-forest. We use the term stand to refer to scales from groups of trees through to a forest block, the term landscape to refer to scales from a forest block through to approximately 200,000 ha, and the term whole-of-forest to refer to the full extent of the jarrah forest. This choice of scales is somewhat arbitrary and the scale indicated by a particular term will vary depending on the management activity, or situation, and the animal species discussed.

There is no single perfect scale for applying strategies to manage hollows. Implementing management strategies for hollow-bearing trees across spatial scales caters for the array of fauna species that occur at various scales, their home ranges, populations, and population dispersal. When management strategies are applied at various scales the strategies support one another and enhance the value of all retained habitat. Lindenmayer and Franklin (2002) identify four reasons why the multi-scaled approach is desirable. Firstly, the range of hollow-dependent species that use the forest occur at a range of spatial scales, and have differing spatial requirements. Secondly, these species may respond to a range of environmental factors that impact at different spatial scales. Thirdly, there is interdependence between different scales. For example, the response of fauna to the harvesting of a patch of mature jarrah forest in a landscape consisting of uncut areas in informal reserves, fauna habitat zones and formal reserves may be quite different to the response of fauna to harvesting of a patch of mature forest surrounded only by 50,000 hectares of regrowth forest. Fourthly, a multi-scaled approach is more likely to provide a heterogeneous landscape that is important for some species. Lindenmayer and Franklin (2002) describe the implementation of a range of strategies across spatial scales as risk-spreading. This is a particular form of risk management for forests (Ferguson *et al*. 1997) that specifically addresses risk at a range of scales but also recognizes the interdependence of how these risks are managed at each of the scales.

Spatial scale also underlies all strategies for hollow management, affecting how the management of hollows is perceived and implemented. The extremes of scale illustrate how some scales enable particular management solutions and hinder others. A small stand-scale perspective tends to focus attention on retained habitat trees. This scale deals poorly with stands that lack hollow-bearing trees and does not facilitate perspectives for landscape and whole-of-forest issues, such as species dispersal and metapopulation dynamics, particularly in forests managed for timber production that include recently harvested stands. The other spatial extreme is the whole-of-forest. At this scale, planning and management has historically tended to focus conservation management on formal conservation reserves that provide relatively secure habitat. Focusing on the provision of hollows across spatial scales enables the inclusion and management of the matrix of forest (Lindenmayer and Franklin 2002) that is subject to timber harvesting. Thinking and planning can then focus attention on the stand or the whole forest, and include all trees remaining in the stand, not just trees prescribed for retention. This approach facilitates conservation planning across the forest and provides an effective context for managing stands that have relatively few or no hollow-bearing trees, including those that have been harvested for timber.

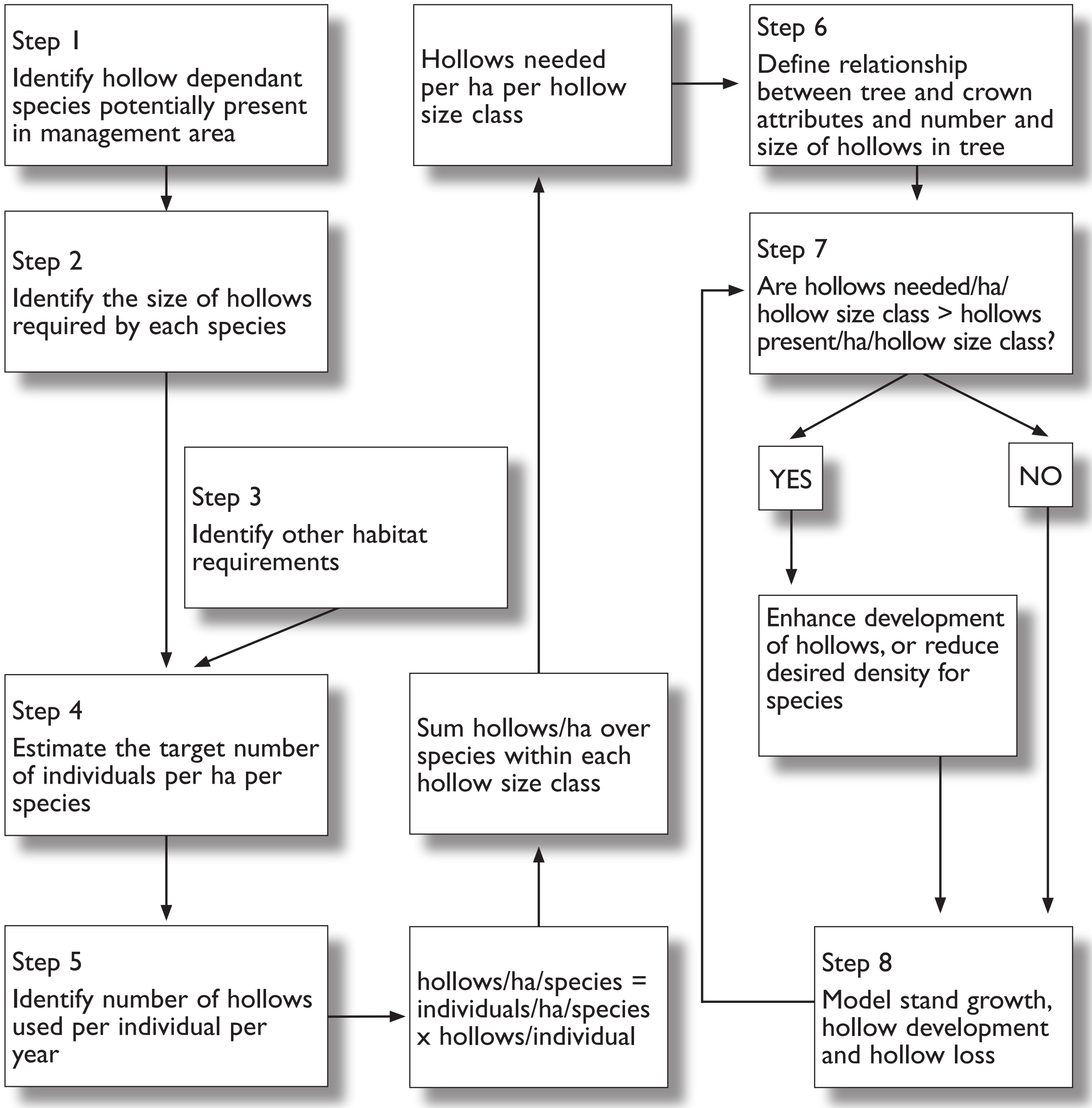

Figure 2. A framework for determining how many hollow-bearing trees need to be retained at the stand scale and assessing the adequacy of the retained hollow bearing trees (based on McComb 1992, after Nietro *et al* 1985).

Part 1: A stand-scale numerical approach

Fig. 2 is an example of a stand-scale approach to hollow management. The process outlined in this figure provides an estimate of the number of hollows and hollow-bearing trees required on a forest stand to support the hollow-dependent fauna. Fig. 2 also includes an assessment of the adequacy of these hollows at present, and at some time in the future. Completing the process outlined by Fig. 2 in a rigorous manner is not yet possible in the jarrah forest. However, the framework provides a useful structure in which to review the information that is available, identify deficiencies in knowledge, and assess and interpret the habitat value of stands.

The process outlined in Fig. 2 consists of eight key information inputs, a calculation process, and an adaptive loop on the right hand side of the figure. The key steps are:

Step 1. Identify the hollow-using species potentially present in the management area;

Step 2. Identify the size of hollows required by each species;

Step 3. Determine other habitat requirements;

Step 4. Estimate the target population densities of males and females per ha per species;

Step 5. Estimate the number of hollows used per individual per year;

Step 6. Determine the relationship between tree and crown attributes and the number and size of hollows in the tree crown;

Step 7. Determine if sufficient hollows exist in the stand;

Step 8. Model stand growth, hollow development, and hollow loss.

From these inputs, the number of hollows required, and the sizes of these hollows, could be estimated per hectare and individual stands managed on the basis of the number, size and condition of the trees retained on the stand. We do not propose that such intensive, stand-focused management of hollows is required across the jarrah forest. However the process of Fig. 2 provides a basis for assessing the hollow-bearing capacity of individual stands and interpreting how these stands might work together to affect conservation across the landscape. The process of Fig. 2 also indicates the knowledge base needed to provide detailed support for stand-scale guidelines for retention of trees to provide habitat for hollow-dependent fauna. In Western Australia, primary habitat trees and secondary habitat trees are marked with a large white 'H' and retained in the timber harvesting process. The selection and marking of these trees is the first priority in the tree marking process (CALM 1995).

The following sections review research relevant to the jarrah forest in relation to the eight steps in the framework of Fig. 2.

Species potentially present in the management area, step 1

Hollow using species in the forest

Step 1 of Fig. 2 requires knowledge of the hollow-dependent species potentially present in the forest stand or management area. There are 42 vertebrate species[1] known to use hollows in the jarrah and karri forests (Abbott and Whitford 2002). These comprise 18 landbird, 16 mammal, three waterbird and five reptile species. The 16 mammals consist of three large mammals (chuditch *Dasyurus geoffroii,* common brushtail possum *Trichosurus vulpecula* and western ringtail possum *Pseudocheirus occidentalis*), two medium-sized mammals (the numbat *Myrmecobius fasciatus* and the Wambenger, an undescribed *Phascogale species*, previously *Phascogale tapoatafa*), and 11 small mammals (nine bat species, and the mardo *Antechinus flavipes* and western pygmy possum *Cercatetus concinnus)*. Nine mammal and 17 bird species are totally dependent on hollows in standing trees for successful breeding (Abbott and Whitford 2002).

Species using hollows in logs

Of the hollow-using vertebrates, the numbat, chuditch and mardo are ground-dwelling species and are currently well catered for by the substantial number of tree limbs and log ends present on the ground from historical timber harvest activities and natural tree fall. Faunt conducted a detailed survey of ground logs and large limbs on seven one-hectare plots in harvested forest and three plots in forest that had never been harvested for timber (Faunt 1992; Williams and Faunt 1997). Analysis of these data revealed there were twice as many logs and limbs (≥70 cm in diameter) on the ground in the harvested plots (15.9 ± 2.0 per ha, n=7) than in the unharvested plots (8.0 ± 2.1 per ha, n=3). There were also five times more log hollows on harvested plots than on unharvested plots (4.7 ± 1.1 per ha, n=7 vs 0.7 ± 0.3 per ha, n=3). Jarrah logs are relatively decay resistant (Bootle 1983, Brown *et al.* 1996) and commonly persist for periods of 80 years or more on the forest floor (Whitford and Williams 2001). Thus, although sustainable management of the forest requires consideration of the long-term recruitment, decay, and removal of ground logs, and these longer-term issues should be considered at some stage, the current availability of hollows in ground logs is not a concern in the jarrah forest. Thus the focus of discussion of this paper is on hollows in standing trees.

Hollow-dependent species present on individual stands

Biological surveys of priority areas throughout the forest are proposed for completion over the next 10 years under the Forest Management Plan 2004-2013 (Conservation Commission 2004). While native mammals are absent or their numbers are low (de Tores 1999), fauna surveys of jarrah stands may provide little guidance for determining which species could be expected to occur, or their population density. If available, knowledge of the species that may occur on forests stands would inform step 1 of Fig. 2. The Forest Management Plan 2004-2013 commits

[1] Abbott and Whitford (2002) excluded from their list five Western Australian forest species that Gibbons and Lindenmayer (2002) described as hollow users: *Petroica multicolor, Melanodryas cucullata, Colluricincla harmonica, Acanthiza inornata,* and *Tarsipes rostratus.*

to the development of a system of pre-logging fauna assessment to predict fauna occurrence in each forest block (Conservation Commission 2004). Christensen *et al.* (2001) developed a system (Fauna Distribution Information System) for determining the likely presence of vertebrate fauna species based on the association of fauna species with vegetation complexes (Mattiske and Havel 2000). Formal monitoring activities such as FORESTCHECK (Abbott and Burrows 2004), the biological surveys of priority areas anticipated in the Forest Management Plan 2004-2013 (Conservation Commission 2004), and informal monitoring and survey activities, provide more locally focused information that could add to and refine information from the Fauna Distribution Information System. Information from all of these sources should be considered when determining which fauna species may occur in a particular forest area.

Hollow-dependent species most impacted

For the 42 vertebrate species that use hollows in standing trees, the degree of reliance on hollows varies widely, with 29 being highly-dependent on hollows for breeding (11 mammals, 17 birds and 1 reptile) (Abbott and Whitford 2002). Abbott and Whitford (2002) used body dimensions to estimate minimum dimensions of hollows that could be used by these 42 species. The frequency with which hollows larger than this size occurred in a sample distribution of hollow sizes was used to calculate a Relative Frequency of Hollow Occurrence. On this basis, 17 of the 42 species (2 mammals and 15 birds) were identified as likely to use hollows of sizes that are relatively rare (Abbott and Whitford 2002). The 17 species listed in Table 1 are the logical focus for any strategy for the retention of hollows at the stand scale. This is because smaller hollows are more numerous than large hollows in trees of any size, smaller hollows can occur frequently in smaller trees whereas large hollows are rare in smaller trees, smaller hollows regularly occur in trees with large crowns that also have large hollows, and smaller hollows are more likely than larger hollows to occur in the trees that are not specifically retained for their hollow-bearing value, but remain after harvesting (Whitford 2002, Whitford and Williams 2002). From the 17 species likely to use hollows of sizes that are relatively rare, Abbott and Whitford (2002) identified two mammal and six bird species that had relatively small home ranges. These eight species were proposed by Abbott and Whitford (2002) as those species most likely to reveal the impact of any local or regional shortage of hollows.

The size of hollows required by each species, step 2

The size of hollows is of primary importance in identifying and managing hollows that can potentially be used by hollow-dependent animals. There are clear physical limits to how small a hollow an animal will use as the animal must first fit through the entrance, and then must fit inside the hollow. There are also sound reasons for expecting an upper limit to hollow size as this influences factors such as predation, competition (Van Balen *et al.* 1982, Newton 1994) and thermal regulation (Schmidt 1979, Driscoll 2000). Knowledge of hollow size can be used to manage hollows. The size of hollows is related to branch and tree size, and to crown attributes, and the tree is the unit of management around which the planning, growth and tending of forests is logically based.

Table 1. From the 42 species that use hollows in the jarrah and karri forest, Abbott and Whitford (2002) identified two mammals and 15 birds that are highly-dependent on hollows for breeding and use hollows that are relatively rare in the jarrah forest. Of these, both mammal species and six bird species were identified as most likely to reveal the impact of any shortage of hollows, and thus are prime candidates for monitoring.

Common name	Species	Species most likely to be impacted (Abbott and Whitford 2002)
Koomal or Brush-tailed Possum	*Trichosurus vulpecula*	X
Wambenger	*Phascogale sp.*	X
Rufous Treecreeper	*Climacteris rufa*	X
Sacred Kingfisher	*Todiramphus sanctus*	X
Forest Red-tailed Black Cockatoo	*Calyptorhynchus banksii naso*	X
Baudin's Cockatoo	*Calyptorhynchus baudinii*	X
Carnaby's Cockatoo	*Calyptorhynchus latirostris*	
Western Long-billed Cockatoo	*Cacatua pastinator*	
Western Rosella	*Platycercus icterotis*	X
Red-capped Parrot	*Platycercus spurius*	X
Ring-necked Parrot	*Platycercus zonarius*	
Purple-crowned Lorikeet	*Glossopsitta porphyrocephala*	
Masked Owl	*Tyto novaehollandiae*	
Boobook Owl	*Ninox novaeseelandiae*	
Barking Owl	*Ninox connivens*	
Barn Owl	*Tyto alba*	
Mountain Duck	*Tadorna tadornoides*	

Correctly describing the size and shape of hollows used by animal species is essential if size and shape are used to identify hollows that are potentially suited to individual fauna species, and thus to predict how many of these hollows occur in the forest (Step 2, Fig. 2). Studies of hollow size have been completed for six bird and four mammal species of the jarrah forest: common brushtail possum, western ringtail possum, wambenger, mardo, red-tailed black cockatoo *Calyptorhynchus banksii naso* and *Calyptorhynchus banksii samueli*, Australian ringneck *Platycercus zonarius*, western rosella *Platycercus icterotis*, red-capped parrot *Platycercus spurius*, rufous treecreeper *Climacteris rufa*, and striated pardalote *Pardalotus striatus*. As these 10 species cover the range in size of hollows used by all hollow-dependent species that occur in the forest, they can be used to reach general conclusions about availability of hollows for all birds and mammals. Whitford (2001) examined the dimensions of 197 hollows used by these birds and mammals and developed a system for consistently defining the range of the hollow sizes and shapes used by each species. This work built on data collected by Saunders *et al.* (1982), Wardell-Johnson (1986), Long (1990), Dickman[2] (1991 and unpublished data), Soderquist[3] (1993 and unpublished data), M. Craig[4] (unpublished data), R. Johnstone[5] (unpublished data), and Rhind (1998). The Whitford (2001) system defined the range of suitable hollow entrance sizes and shapes as those lying within a two-dimensional space. Similarly the depth and width dimensions of the hollow were considered a related pair of dimensions that defined the range of internal hollow spaces. Combined with a specified minimum height of the hollow above the ground, these attributes were used to identify hollows potentially suited to each species (Whitford 2001, Whitford and Williams 2002).

Linear regressions of body dimensions have also been used to estimate the minimum dimensions of hollows used by the 42 species of birds and mammals that use hollows in standing trees in the jarrah and karri forests (Abbott and Whitford 2002). This simple characterisation of the size of hollows is useful only in making general interpretations about hollows such as the relative frequency of hollow occurrence and the minimum size and age of trees bearing hollows potentially suited a species.

Other habitat requirements, step 3

The size of the hollows required by a species determines the number of hollows potentially available for that species to use. There may be many other attributes of the hollow and factors associated with the location of the hollow in the tree, stand, or landscape, that influence use of the hollow. These include the spacing of hollows; the density, structure, and composition of surrounding vegetation; and the location of hollows relative to food and water. Presumably, animals will select and use hollows from the population of all hollows with suitable dimensions, based on the attributes of the hollow and of the surrounding vegetation.

Evidence of the importance of factors other than hollow entrance dimensions, internal dimensions, and entrance height in determining the suitability and use of hollows is inconclusive. A review of 38 papers that discuss the attributes of hollows used by hollow-dependent fauna (excluding papers that deal only with the attributes of trees used by hollow-dependent fauna) identified 33 papers that report the dimensions of hollows. Of these 33 papers, 26 (79 percent) present data showing particular species used hollows with particular dimensions (Table 2). Three of the papers identified species for which no significant preference for size of hollows could be identified. The remaining four papers expressed no clear conclusion regarding dimensions of hollows.

Of the habitat requirements other than hollow size, the aspect of hollow entrances has received the most attention. Of 20 papers that deal with this issue, 10 state that aspect is not important or not significant in determining hollow use, four papers identify aspect as an important or significant factor affecting hollow use, and seven papers are inconclusive regarding the importance of entrance aspect (Table 2). Similarly there is not strong evidence for the importance of other hollow attributes that may influence hollow selection and use, such as branch versus trunk location, and living or dead wood.

In Western Australia, Saunders *et al.* (1982) found the Wheatbelt subspecies of red-tailed black cockatoo *C. banksii samueli*, that occurs in the dry agricultural zone east of the jarrah forest, did not randomly select live and dead trees, but preferentially used hollows in dead trees at approximately twice the frequency expected based on random selection. Rhind (1996) found that 27 percent of trees used by phascogales were 'dead' (defined as ≥ 95 'dead' in contrast to the more usual definition of 100 percent dead). Whitford and Williams (2002) found that dead trees are more likely to bear hollows suited to phascogale (and rufous treecreeper), and the finding of Rhind (1996) may reflect this likelihood rather than a preference or a greater value of hollows in dead trees. van der Ree *et al.* (2001), working in box woodland in the Northern Plains of Victoria, found that 94 percent of phascogale nests were located in living trees. These studies highlight the value of work such as Saunders *et al.* (1982), which established the attributes of all hollows on the stand and thus enabled the researchers to distinguish preferential use from availability.

Luck (2002) found that 82 percent of 48 hollows used by rufous treecreepers in the Wheatbelt had a spout angle ≥ 50° (angle of major axis of limb from the horizontal). Observations in the jarrah forest may indicate a similar preference as Craig[6] and Whitford (unpublished data) found that of 20 hollows used by rufous treecreepers, 90 percent had a spout angle > 60°.

In unharvested stands, Gibbons *et al.* (2002) found that internal dimension of hollows were the best predictors of hollow occupation, even when tree and site factors were considered. However, the selection and use of hollows by some species may be influenced by attributes of the surrounding habitat. For example, Lindenmayer *et al.* (1991a) found that hollows in trees with dense surrounding vegetation were more likely to be used by Leadbeater's

[1]Abbott and Whitford (2002) excluded from their list five Western Australian forest species that Gibbons and Lindenmayer (2002) described as hollow users: *Petroica multicolor, Melanodryas cucullata, Colluricincla harmonica, Acanthiza inornata,* and *Tarsipes rostratus*. [2]Dr C. Dickman, School of Biological Sciences University of Sydney, NSW. [3]Dr T. Soderquist, National Parks and Wildlife Service, Dubbo, NSW. [4]Dr M. Craig, Mosman Park, WA. [5]R. Johnstone, Western Australian Museum, Perth, WA. [6]Dr. M. Craig, Mosman Park, Western Australia.

Table 2. Hollow size and aspect were examined in 38 papers as important factors influencing hollow use. This table lists those papers that examined these attributes and summarizes their conclusions regarding the importance of these hollow attributes.

Papers discussing the attributes of hollows used by hollow-dependent fauna	Examined hollow dimensions	Identified hollow size preference	Found no hollow size preference	Examined hollow aspect	Found aspect is important	Found aspect is not important	Inconclusive regarding aspect
Ambrose, 1982	I	I		I		I	
Buchanan *et al.,* 1993							
Calder *et al.,* 1983	I	I		I		I	
Dickman, 1991	I						
Gehlbach, 1994	I	I					
Gellman and Zielinski, 1996	I	I					
Gysel, 1961	I	I	I				
Haseler and Taylor, 1993	I	I		I	I		I
Inions, 1985	I	I		I			I
Inions *et al.,* 1989	I	I		I			I
Jackson, 1978	I						
Joseph *et al.,* 1991				I			I
Kehl and Borsboom, 1984	I		I	I			I
Long, 1990	I	I					
Luck, 2002	I	I		I			I
McComb and Noble, 1981	I	I		I	I		
Medway and Marshall, 1970	I	I					
Menkhorst, 1984a	I	I		I		I	
Menkhorst, 1984b	I	I		I		I	
Newton, 1994 (a review)	I	I					
Nilsson, 1975	I	I					
Oldroyd *et al.,* 1994	I	I					
Pinkowski, 1976	I	I		I		I	
Rhind, 1996	I	I					
Rose, 1996	I						
Saunders *et al.,* 1982	I	I		I		I	
Saunders, 1979	I	I		I		I	
Schmidt, 1979				I		I	
Soderquist, 1993	I						
Taylor and Savva, 1988	I						
Tidemann and Flavel, 1987	I	I		I			I
Traill and Coates, 1993	I	I					
Traill, 1995	I	I		I		I	
Van Balen *et al.*, 1982	I	I		I	I		
Vonhof and Barclay, 1996	I			I		I	
Wardell-Johnson, 1986	I	I					
Woinarski and Bulman, 1985							
Woinarski, 1973.				I	I		
Totals for all 38 papers	33	26	2	20	4	10	7

possum *Gymnobelideus leadbeateri*, and Jones and Hilcox (1992) found that abundance of western ringtail possum was positively related to both the abundance of hollow-bearing trees and the extent of the myrtaceous canopy in a tuart *Eucalyptus gomphocephala* forest. Similarly, for some species in the jarrah forest, proximity to food and water sources may influence the selection of hollow-bearing trees by some fauna.

Apart from the above studies, little is known of the importance of other physical, social, and environmental factors that determine whether a hollow that has suitable dimensions is used by a bird or mammal in the jarrah forest. Results from current and future research can be expected to improve knowledge in this area. However, as indicated by Fig. 2, these factors are only relevant once a hollow is present, and the provision of hollows of appropriate size and number is a fundamental goal. Substantial progress in managing the resource of hollows can be made on the basis of the relationship between hollow size, and tree size and form. This can be achieved by managing the number, size, types and forms of trees that are retained - as the tree, not the hollow, is the basis of forest management. Hollow-dependent fauna will select suitable hollows from the retained trees.

Target population density, step 4

Step 4 of Fig. 2 makes use of an estimate of the target population density. Such an estimate would be consistent with the goal of conserving biodiversity in the jarrah forest (Conservation Commission 2004) and thus maintaining a viable population of the species. Such target population densities for individual species would vary between forest areas as not all fauna species can be expected to occur on all sites at all times, and a range of habitat attributes influence the size of fauna populations (Lindenmayer *et al.* 1991b).

For some fauna species, populations may be restricted to discrete habitats or areas that have experienced relatively little disturbance (national parks, reserves, and forest blocks or patches with particular disturbance histories or vegetation structure or type). These areas may be separated by, or interspersed with, areas that have less complete habitat value to particular fauna species (i.e. they lack some particular habitat attributes of value to the species). This situation is embodied in the concept of fauna habitat zones and in the status of the various land categories that make up the publicly owned forest (Conservation Commission 2004). These land categories encompass the notion that particular forest areas will have higher conservation value or emphasis than other areas either through intrinsic value or by specified management purpose. For example, nature reserves are managed for the express purpose of the protection, care, and study of indigenous flora and fauna. State forest and timber reserves are managed for a combined purpose of conservation, recreation, sustainable timber production and water catchment protection, while timber reserves and State forests that are planted with exotic timber species are managed to maximise timber production. These latter two land categories, State forests and timber reserves, are referred to by Lindenmayer and Franklin (2002) as the matrix. Lindenmayer and Franklin (2002) propose critical biodiversity conservation roles for the matrix of supporting populations of species, regulating the movement of organisms, and buffering sensitive areas and reserves, and maintaining the integrity of aquatic systems. Figs 3 and 4 provide examples of the matrix of forest available for timber harvesting at the landscape and stand scales. Within the area subject to timber harvesting, a variety of restrictions and activities, including reservation of stands and trees, aim to ensure the harvested matrix contributes to biodiversity conservation (see the sections on *Landscape heterogeneity* and *Stand structural complexity*). It follows that the target population densities for individual species will vary across the forest, and between stands.

At this stage there is little published information available on the population density of the hollow-dependent birds, mammals, and reptiles that inhabit jarrah forest stands. The population density of the common brushtail possum has been studied in the jarrah forest and a variety of other habitats throughout Australia (How 1981; Kerle 1984; How and Kerle 1995; How and Hillcox 2000). Estimates of population density range from 0.2 to 4 possums per ha (How and Kerle 1995). Maxwell *et al.* (1996) cite densities of up to 0.4-0.5 per ha in the jarrah forest. Wayne (pers. comm.) found that population densities of common brushtail possum at Perup in the jarrah forest were highly variable over relatively small spatial scales (<1000 m) and ranged from approximately 0.5 to 2.6 per ha. Population densities may significantly exceed this in more optimal jarrah habitat (Wayne pers. comm.). How and Hillcox (2000) found a mean of 78.4 individuals on a 44.3 ha site of tuart/peppermint forest, a density of 1.7 to 2.84 possums per ha. How and Hillcox (2000) discuss the considerable variation in home range sizes for male and female common brushtail possum in various habitats and note that this is probably a reflection of density, social organisation, and resource availability.

By comparison the population density of phascogale is typically low. Soderquist (1995a) estimated a density of one individual per 50 ha in Victorian box-ironbark forest. Female phascogales occupy intra-sexually exclusive territories (Soderquist 1995a, Rhind 1998). van der Ree *et al.* (2001) found that the size of female home ranges in box woodland in Victoria was one-eighth the size of female home ranges at three other Victorian forest sites (Soderquist 1995a). van der Ree *et al.* (2001) attribute this difference in size of home range to differences in habitat quality, the box-ironbark forest sites being less fragmented but more degraded, and having a lesser number of large trees than the box woodland. Rhind (1998) similarly observed variations in size of home range related to habitat quality. Presumably these differences in home range size that relate to habitat quality indicate corresponding differences in population density that relate to habitat quality. Rhind (1996) estimated territory size of the phascogale to be between 20 and 30 hectares in jarrah forest.

Wayne *et al.* (2000) noted a dramatic reduction in the population of phascogale in the Kingston block (in the southern jarrah forest) at the same time that Scarff *et al.* (1998) observed a reduction in weight and reproductive success associated with a reduction in abundance of invertebrates. Along with the habitat quality of the study area, such episodic variations in populations need to be considered when interpreting reported population densities for all species.

There is little information available on the population density of the many hollow-using birds that inhabit the jarrah forest. Abbott (1998) undertook surveys of red-tailed black cockatoo in the jarrah and karri forests and examined 387 earlier surveys. He noted that the species, which has a large home range and occurs in breeding groups with an average of six birds, is often not observed in bird surveys. Consequently, observations of population density for the red-tailed black cockatoo are heavily skewed. Abbott (1998) estimated the mean population density to be 0.0107 birds/ha or one nesting pair to 116-187 ha, and a total population of around 16,000 to 26,000 birds.

The shortage of information on population densities for jarrah forest fauna prevents the completion of the numerical process outlined in Fig. 2 in a rigorous manner. However, informed estimates of population densities, consistent with the goal of maintaining viable populations, could be applied to the process of Fig. 2 and used to assess the value of stands for fauna conservation and determine the contribution that particular stands can make toward fauna conservation.

Number of hollows used per individual and home range, step 5

Step 5 of Fig. 2 requires the estimation of the number of hollows used per individual per year. Inions *et al.* (1989) found between 2 and 7 hollow-bearing trees per hectare were used by the common brushtail possum and the western ringtail possum (mean, 3 trees per ha), and identified a mean home range of 3 ha for both species (Inions 1985). Vellios (1981) (cited in Inions 1985) found common brushtail possums used between 4 and 11 hollow-bearing trees. How and Hillcox (2000) found the maximum number of hollows used by an individual brushtail possum in a tuart/peppermint forest was 17.

Wayne *et al.* (2000) studied the refuge use and home range of western ringtail possums before and after harvesting at Kingston block in the jarrah forest. From a study of 2626 occupancies of 413 refuges by 29 animals (two to 35 refuges per individual), they estimated that the western ringtail possum would use 20 different refuges each year in both harvested and control areas. These refuges included hollows in standing trees, nests in grass trees *Xanthorrhoea preissi*, nests above ground, nests in forest debris, burrows, and hollow in tree stumps. Although the proportional occupancy of each type of refuge varied greatly between and within experimental treatments, Wayne (pers. comm.) estimated, that in a typical jarrah forest stand, approximately 65 percent (13) of the 20 refuges used each year by a western ringtail possum would be hollows in standing trees. Driscoll (2000) compared the thermal properties of refuges used by the western ringtail possum in grass trees with those in tree hollows and concluded that grass trees provided insulation qualities equivalent to tree hollows. These studies highlight the flexibility in refuge use by western ringtail possums, and the habitat contribution made by refuges other than hollows in trees, e.g. grass trees, above ground nests, and forest debris.

Wayne *et al.* (2000) estimated home range size for western ringtail possums from refuge locations. The mean home range was 2.7 ha for individuals with more than 100 refuge observations. This is consistent with Inions (1995) estimate of 2.6 ha in similar forest, and the estimate of Jones (1995) of 2.5 ha. Jones *et al.* (1994) observed a mean home range of 0.28 ha for males and 0.44 ha for females in a tuart forest, and about 2.4 ha for females in a jarrah forest. The females used between four and seven hollow trees (Jones *et al.* 1994). Wayne *et al.* (2000) concluded that a mean of 7.7 refuges per ha were used by western ringtail possums. These included all refuge types. Considering only the tree refuges, this is approximately five tree hollows per ha.

Rhind (1996) estimated territory size of the phascogale to be between 20 and 30 hectares in the jarrah forest and that females would use an average of 38 trees, while males would use an average of 27 trees. To estimate the required density of hollow-bearing trees, Rhind (1998) assumed male and female territories overlapped (i.e. a mean territory size of 20 ha), that male and female densities were equal, and that tree use was exclusive. This provides a total requirement of 3.25 trees per ha with 7.3 trees per ha required when males congregate around females at mating. In contrast, van der Ree *et al.* (2001) found that female phascogales, tracked over and average of 28 days, used an average of only 11.4 refuges in high-quality woodland where home ranges were much smaller than in the jarrah forest (mean 5.58 ha). This corresponds to a mean of approximately 2 trees per ha. In contrast to phascogales in forest in Victoria (Soderquist 1993, 1995b), phascogales in the jarrah forest have rarely been observed using refuges other than hollows in standing trees (Rhind 1996).

Rose (1996) and Luck (2001) studied rufous treecreeper in wandoo woodland south-east of the jarrah forest in Western Australia. Rose (1996) found the mean territory size was 7.8 ha. In woodland with a high density of hollows (91 ha^{-1}), Luck (2001) determined a mean territory size of 2.6 ha. Each territory supported a cooperative breeding group of between two to seven individuals (mean = 3).

Lindstedt *et al.* (1986) discussed the relationships between size of home range and body size for mammals in north America and developed regressions for predicting the size of home ranges. McComb and Lindenmayer (1999) have noted the relationship between body mass and territory size for birds. Abbott (1998) developed a regression for predicting the size of the home ranges of bird species in the Western Australian forest based on the body weight of the bird species. Such relationships provide a basis for estimating territory or home range size where sufficient information is unavailable.

Size of home range varies with population density (How and Kerle 1995), and home range size and number of hollows used may vary with habitat quality. As with population density, there is insufficient information on home ranges and the numbers of hollows used to rigorously complete the process outlined by Fig. 2. This shortage of information is acknowledged in the recent Draft Forest Management Plan for these forests (Conservation Commission, 2002).

Tree and crown attributes and the number and size of hollows in the tree, step 6

Faunt (1992), McComb *et al.* (1994), Whitford (2001, 2002) Whitford and Williams (2002) have studied the relationship between tree and crown attributes, and occurrence of hollows in the jarrah forest and these studies provide information for step 6 of Fig. 2. The most recent of the studies examined 665 hollows found in 154 jarrah and 85 marri trees (Whitford and Williams 2002). Of these 665 hollows, 204 hollows in 84 trees, were of a size and shape consistent with dimensions of hollows used by at least one of 10 species of hollow-using birds and mammals (Whitford 2001). For each of the 10 hollow-using birds and mammals, logistic and Poisson regressions were used to determine the relationship between hollow occurrence and tree and crown attributes. Although these relationships varied for each animal species, common themes emerged:

- The general relationship between tree age and diameter is similar for jarrah and marri trees;
- For most birds and mammals, usable hollows are not uncommon in trees of 40 - 50 cm diameter over bark (DOB) measured at 1.3 m above ground. The mean age of trees of this size is 100 - 124 years (Whitford and Williams 2002);
- The probability that a tree will bear at least one hollow increases with tree diameter and age;
- The number of hollows borne by jarrah and marri also increases as the size and age of the tree increases;
- Trees with large crowns (wide and/or deep) bear more hollows than trees with small crowns;
- Trees with intact and undamaged crowns bear few or no hollows;
- Small hollows are far more numerous in the forest than large hollows;
- Large hollows are more likely to occur in large trees than in small trees;
- Large hollows are more common in highly senescent crowns;
- Highly senescent trees with no crown or few branches bear few or no hollows. However, these trees can bear very large hollows used by large cockatoos and common brushtail possums;
- Jarrah and marri trees are equally likely to be hollow-bearing. However;
- Hollow-bearing marri trees have more hollows than hollow-bearing jarrah trees; and
- Marri trees are more likely than jarrah trees to bear some types of large deep hollows;
- 65 percent of all potentially usable hollows occur in the dead wood and 35 percent occur in live wood;
- For most hollow-dependent species, dead trees are no more likely than live trees to bear potentially usable hollows;
- However, dead trees are more likely to bear hollows potentially suited to rufous treecreeper and phascogale, and to bear more of these hollows than live trees.
- Large hollows are more common in tree crowns (live and dead) that bear dead branches with large diameter terminal ends;
- The size of the terminal end of the largest dead branch in the crown is related to the size and number of hollows in the crown; and
- The number of hollows borne by a tree increases as the size of the terminal end of the largest dead branch in the crown increases, and then decreases as the associated branch loss reduces the size of the crown.

The number of hollows available, step 7

The data of the Jarrah Forest Inventory (JFI) (Spencer 1992) provides tree diameter (measured in 1991) and species for all trees on 2,836 plots of 0.125 ha which are distributed throughout the jarrah forest that was available for harvesting at that time. The Poisson regression models of Whitford and Williams (2002) can be applied to data from the JFI to estimate the number of hollows available for forest fauna. To validate these models, the number and sizes of hollows predicted by the regression models was compared with the number and sizes of hollows actually found (n = 160) in the 145 trees searched by McComb *et al.* (1994) on nine harvest coupes across the jarrah forest. The searches by McComb *et al.* (1994) did not include trees smaller than 50 cm diameter, did not include highly senescent and decayed trees that were left standing in the harvesting operations, and examined no trees greater than 130 cm in diameter. Consequently, the validation and the subsequent use of the regressions models only considered trees larger than 50 cm in diameter, and trees with diameters larger than 130 cm were assigned a diameter of 130 cm. In the jarrah forest, some hollows are found in trees smaller than 50 cm (Inions *et al.* 1989; Rhind 1996; Whitford 2002). As the assessments of Crown Senescence required by the regression models were not made by McComb *et al.* (1994), or collected in the JFI, a set of distributions of Crown Senescence for each 10 cm diameter class were applied to both the data of McComb *et al.* (1994) and to the JFI. These distributions of Crown Senescence were determined from assessments of 2328 jarrah and marri trees on 79 sites across the jarrah forest (Whitford and Williams 2001). Less than 6 percent of trees in the JFI are species other than jarrah and marri for which the regression models of Whitford and Williams (2002) were developed. For this exercise, the small number of minor tree species was assumed to be either jarrah or marri.

The number of hollows predicted to occur in the trees searched by McComb *et al.* (1994) and the number of hollows actually found in these trees is compared in Table 3. The regression models under-estimated, by a small to moderate extent, the number of hollows found in the trees for all species except for forest red-tailed black cockatoos, common brushtail possums and western rosellas. For these three species the regression models over-estimated the number of hollows. For the western rosella, this over-estimate was minor (2.5 percent). For the two largest species, forest red-tailed black cockatoo and common brushtail possum, this over-estimate of the number of hollows was substantial.

In this validation exercise the distribution of Crown Senescence for each 10 cm diameter class was modelled on that found in the wider forest. The validation data set of McComb *et al.* (1994) was a random sample, but it excluded large decayed and highly senescent trees that are generally not felled, as they produce low quality sawlogs. These highly senescent trees have a high probability of bearing large hollows that are potentially suited to red-tailed black cockatoo, and common brushtail possum. The exclusion of these large trees from the validation data set probably accounts for the apparent over-estimate of the number of large hollows, and a random sample

Table 3. Validation and application of the regressions of Whitford and Williams (2002). McComb ***et al.*** (1994) found 160 hollows in 145 trees. The table lists 68 of these that were potentially suited to one or more species of bird or mammal (Whitford 2001) and the number of hollows predicted by the Poisson regression models of Whitford and Williams (2002) for these 145 trees. Estimates of the mean number of hollows occurring in the jarrah forest are based on the Jarrah Forest Inventory (JFI) of tree diameters and species for all trees on 2836 plots of 0.125 ha (Spencer 1992). The mean number of potentially usable hollows present per hectare in a sample of five trees randomly selected under the habitat tree specification (excludes any hollows that may be provided by the 6-8 potential habitat trees). The habitat trees are jarrah and marri ≥ 70 cm diameter over bark measured at 1.3 metres, with a Crown Senescence assessment (Whitford 2002) in the range of 4 – 7. The trees and hollows may be used by more than one species. Consequently, the predicted total of 4.5 hollows occurring in 2.6 of the 5 trees is not the sum across all species.

Common name	No. of hollows found in search of 145 trees (McComb et al. 1994),	No. of hollows predicted for these 145 trees (Whitford and Williams 2002).	Mean number of hollows/ ha in jarrah forest. predicted from JFI mean ± s.e.	Mean number of trees bearing hollows predicted from five primary habitat trees	Mean number of hollows potentially suited to species predicted in five primary habitat trees
Forest Red-tailed Black Cockatoo	1	3.7	1.0 ± 0.02	0.3	0.3
Koomal or Brush-tailed Possum	3	8.7	3.4 ± 0.06	0.7	0.9
Ngwayir or Western ringtail possum	22	20.4	8.1 ± 0.15	1.4	2.0
Mardo	16	9.5	3.7 ± 0.07	0.7	0.9
Wambenger or Phascogale	35	22.3	8.9 ± 0.16	1.3	2.2
Australian Ring-necked Parrot	9	8.3	3.2 ± 0.06	0.7	0.9
Red-capped Parrot	18	11.1	4.3 ± 0.08	0.8	1.2
Western Rosella	4	4.1	1.5 ± 0.03	0.4	0.5
Rufous treecreeper	25	20.8	6.4 ± 0.09	1.3	1.7
Striated Pardalote	29	27.0	8.3 ± 0.12	1.5	2.1
Total of all potentially usable hollows	68	51.8	18.0 ± 0.28	2.6	4.5

of trees should provide a better comparison. However, predictions for red-tailed black cockatoo and common brushtail possum must be regarded with caution.

Table 3 provides predictions of the mean number of hollows per ha in the jarrah forest for ten species including six species identified as most likely to be impacted by a shortage of hollows (Abbott and Whitford 2002). The predictions will under-estimate hollow availability for phascogales and rufous treecreepers as these species have been observed using trees smaller than our minimum diameter of 50 cm (Rhind 1996; M. Craig pers. comm.) and possibly over-estimate the number of hollows suited to red-tailed black cockatoo and common brushtail possums. The latter possibility re-emphasises the importance of monitoring the red-tailed black cockatoo and common brushtail possum and the similar-sized Baudin's Cockatoo *Calyptorhynchus baudinii*, and the value of validating all of these regressions against other data sets of hollow measurements or observations of hollow use. The mean values for the whole jarrah forest (Table 3) are of limited practical use as the forest structure is variable, the data are becoming dated (JFI data collected in 1991) and the predictions are for hollows potentially suited to the 10 fauna species rather than hollows actually in use. Notwithstanding these limitations, the figures in Table 3 provide a valuable numerical perspective on the availability of hollows for the 10 fauna species, and of hollow occurrence in general.

More importantly, this exercise demonstrates how stand scale solutions such as tree retention strategies can be assessed to determine their contribution to the suite of hollow conservation strategies, and how the contribution of all of the strategies, that operate across spatial scales, can be assessed on individual stands. This is examined further in the discussion of stand prescriptions.

Hollow development, loss and stand growth, step 8

Hollow development and tree age

Step 8 of Fig. 2 considers the recruitment and loss of hollow-bearing trees. Hollows larger than 20 mm wide and 100 mm deep begin appearing in trees of about 20 cm DOB and regularly occur in trees of about 50 cm DOB, which have a mean of one of these hollows per tree (Whitford 2002). For all but the largest hollow users, (red-tailed black cockatoo, and maternity hollows of common brushtail possum) it is not uncommon for usable hollows to occur in trees of 45 to 50 cm diameter (see Whitford and Williams 2002 for a more detailed discussion of hollow occurrence and tree age). Rhind (1996) noted that, for phascogales, hollow use became most apparent in trees of

40 cm DOB and larger. The diameter range of 40 to 50 cm corresponds to mean tree ages of 100 ± 3.4 to 124 ± 3.2 years (Whitford 2002). Individual trees may be much older or younger than this, and the trees of this size that actually bear usable hollows are likely to exhibit particular types of crown decline more common in older trees.

The fauna that use large hollows require separate consideration. Of 12 trees found by Whitford and Williams (2002) with hollows potentially suited to red-tailed black cockatoos, the youngest of the trees was 131 years old. This compares with a sample of 23 nest trees used by red-tailed black cockatoos in the jarrah forest that had a mean age of 233 years and a minimum estimated age of 141 years (R. Johnstone[7], unpublished data). This time for hollow development to occur falls within the planning and management period currently considered for most of the jarrah forest. The nominal minimum rotation length for most of the jarrah forest available for timber harvesting is 175 years (EPA 2003).

Loss of hollow-bearing trees

The survival of jarrah and marri habitat trees retained after timber harvesting and the factors associated with the natural fall of these large trees has been examined in the jarrah forest. Whitford and Williams (2001) observed 1,880 trees with DOB greater than 50 cm on 61 harvested sites and 18 sites that had never been harvested. The assessment of these trees covered periods from 5 to 77 years (mean, 46 years). Two percent of trees fell per decade. Hollowing-out of the butt of the tree by fire and subsequent breakage of the tree at this point was the most frequent mode of tree-fall (approximately 72 percent of cases) and few trees fell with their root plate (an extensive lateral root system) intact. Fire frequency and evidence of substantial termite infestation were associated with increased probability of tree fall. Considering only those trees large enough to meet the prescribed habitat tree size (DOB ≥ 70 cm) (CALM 1995), the mean rate of tree fall was 2.4 percent of trees per decade. Based on these rates, of the 500 primary habitat trees currently retained across 100 hectares of harvested forest, 120 are likely to fall within a 100-year period.

Tree growth

The relationship between diameter and age of jarrah trees is well understood, though variability in predicted age increases with increasing tree diameter (Burrows *et al*. 1995; Stoneman *et al*. 1997; Whitford 2002). The time-frame for the development of hollows is known. However, the time-frame reflects two distinct factors: predictable tree growth, and stochastic events with poor predictability – violent storms, intense fires, insect attack - that initiate hollow formation. Individual tree-growth models have not been successfully developed for jarrah, but stand growth can be reliably predicted in the medium term using a matrix approach (Turner 1998). Predicting the future availability of hollows from growth projections requires assumptions of the crown condition at some future time. The current distribution of Crown Senescence could be assumed to persist unchanged. However, the probability of irregular stochastic events that have a significant impact on crown condition and hollow development is unknown, and the validity of this assumption of crown condition is less reliable than growth projections. Although factors that influence jarrah growth rates have been identified and manipulated on experimental sites (Stoneman *et al*. 1997), little is known of how hollow development can be enhanced in jarrah. At this stage, our knowledge of the rate of hollow formation, tree growth, and tree-fall is useful in guiding management decisions and informing planning. However, numerically accurate prediction of hollow development and loss into the future is not yet possible.

PART 2: Management of tree hollows at a range of spatial scales

Guidelines to provide hollows at the stand scale

A silvicultural specification to retain trees for hollow-dependent animals in jarrah forest subject to harvesting was introduced in 1989, based on research by Inions (1985). A revision followed in 1991, and again in 1995, in consideration of preliminary results from the Kingston study (Burrows *et al*. 1994). A further revision that incorporates final results from the Kingston study is under preparation and the key silvicultural settings are outlined in the Forest Management Plan 2004-2013 (Conservation Commission 2004). The draft guidelines for silvicultural practices in jarrah forest were developed following consideration of the work of Wayne *et al*. (2000) and Whitford and Williams (2002) from the Kingston study, a review by Department of Conservation and Land Management zoologists and silviculturalists, and other reviews (Bradshaw 2002, Burrows *et al*. 2002).

The current silvicultural guideline (CALM 1995) specifies the retention of an average of four habitat trees per hectare on all harvested areas, and six to eight potential habitat trees for recruitment as primary habitat trees on areas harvested as gaps (an intensive removal of overstorey on an area less than 10 hectares to allow existing regeneration to develop). At least one hollow log or stump is retained per hectare. Revisions outlined in the Forest Management Plan 2004-2013 will increase the number of primary habitat trees from four to five per hectare, alter the defined crown condition to specify more highly senescent trees, require six to eight secondary habitat trees per hectare and the retention of four large grass trees per hectare where available. Grass trees provide valuable refuges for western ringtail possums. Grass trees are considered neither inferior to, nor a replacement for, hollows in trees as they provide complementary refuge sites that are used in conjunction with hollows in trees (Driscoll 2000). While previously only one hollow log per hectare was marked for retention, the draft guidelines require the retention of all hollow logs longer than three metres with a pipe larger than 10 cm in diameter. The proposed revisions also place greater emphasis on protecting the retained habitat from fire. The five primary habitat trees (DOB ≥ 70 cm) compare with an average of 17 trees per hectare with DOB > 70 cm on virgin jarrah stands across 7 forest blocks (range 11 to 28) (Bradshaw 2002), and no hollow-bearing trees retained at all on mined jarrah forest (the area of jarrah forest mined annually will become similar to the

[7]R. Johnstone, Western Australian Museum, Perth, WA.

area actually cut to gap; 500-1,000 ha/year; Conservation Commission 2002; Bradshaw 2002). Table 3 gives estimates of the mean number of potentially usable hollows present per hectare in a sample of five trees randomly selected under the primary habitat tree specification of the draft guidelines (i.e. jarrah and marri DOB ≥ 70 cm, Crown Senescence 4 – 7) (Conservation Commission 2004; Whitford 2002). On average, half of the five primary habitat trees retained per hectare would be hollow-bearing, and would bear a total of four hollows per hectare that are potentially suited to fauna of the jarrah forest.

The Forest Management Plan 2004-2013 also outlines the additional retention of marri trees, particularly large trees, in both jarrah and karri forest. This substantially increases the number of large marri trees that will be retained on harvested areas.

In addition to marked primary and secondary habitat trees that are retained on harvested areas, many other trees are also retained, and habitat trees are generally the lesser component of the remaining trees. Prescriptions or silvicultural guidelines focus on retained trees that are specifically marked; yet all trees on a stand contribute to habitat, and all stands contribute to biodiversity conservation. For example Armstrong and Abbott (1997) examined four harvest cells (36.5 ha) at Kingston Block and identified 14.4 trees per ha greater than 50 cm DOB that had been retained after timber harvesting. Half of these had been specifically marked for retention. The other half remained after being passed over in the harvesting operation because they did not contain a merchantable product. Most of these 270 unmarked trees of DOB>50 cm would bear at least small hollows (Whitford 2002) and a proportion of these would contribute to providing hollow refuges for some of the vertebrate species that use hollows. Such unmarked, but retained, trees should not be overlooked when considering the effectiveness of silvicultural guidelines.

McComb and Lindenmayer (1999) point out that it is unreasonable to expect all species to have their needs met on all stands, and that clear goals are the first step in developing stand prescriptions. Clear goals are essential if prescriptions are to evolve under an adaptive process. In stands subject to timber harvesting, stand-based prescriptions alone are unlikely to ensure the conservation of all species on all stands at all times, and the effectiveness of prescriptions that retain hollow trees should not be determined without consideration of other relevant conservation measures. In the jarrah forest, the conservation of hollow-dependent fauna relies on conservation measures applied across a range of spatial scales.

Conservation of hollow-bearing trees at a range of spatial scales

The retained habitat trees are the most visible strategy for conservation of hollows in harvested forest, even though many other habitat elements are retained on harvested areas. Although the knowledge gained from studying hollows at the stand scale is valuable for developing strategies that conserve hollows on stands, and for gaining perspective of the availability of hollows at the landscape and whole-of-forest scales, the stand scale is not the only scale for developing hollow conservation strategies. The conservation of hollow-dependent fauna occurs across the formal and informal reserve system, and the matrix of harvested forest.

Gibbons and Lindenmayer (2002) and Lindenmayer and Franklin (2002) list the principles of connectivity, landscape heterogeneity, and stand structural complexity as fundamental to the conservation of biodiversity in forests used for timber production. In Western Australia, the application of these principles is primarily served by reservation at a range of spatial scales, dispersion of timber harvesting in space and time, and silvicultural strategies. All of these strategies work together to provide habitat, including hollow-bearing trees, and contribute to an ongoing provision of habitat across spatial scales. These strategies have been evolving as part of forest management in Western Australian since at least 1973.

Connectivity

Connectivity allows for dispersal and mixing of populations and links areas with differing habitat qualities and attributes. Dispersal enables recolonization of suitable environments and movement away from areas that become unsuitable due to natural or imposed changes. Dispersal allows populations to enlarge and extend through the forest, and thus respond to environmental stochasticity. Connectivity enables genetic intermingling and counters the genetic degradation possible in small populations. Dispersal counters the demographic imbalances that can occur in small populations. Connectivity also enlarges the diversity of environments that are available to forest fauna by providing access to attributes such as large hollows in old trees that are more common in older stands, as well food sources in younger stands and water in riparian areas.

In the jarrah forest, connectivity is provided by:

- Reserves that can be classified into two types:
  - Formal reserves - conservation reserves established and protected by legislation;
  - Informal reserves - a zoning within State forest, established by a management plan, where timber harvesting is not allowed. These areas are either in a linear or patchy form;
- Fauna habitat zones; and
- Habitat elements retained on harvested coupes.

At the landscape and whole-of-forest scales, formal conservation reserves and forest conservation areas provide old-growth forest or relatively undisturbed habitat. Of the 1.81 million ha of jarrah forest, 9.8 percent, or 176,350 ha is currently in 350 formal reserves consisting of 137,190 ha of national parks and 39,160 ha of other formal reserves (Conservation Commission 2004). Additions in the Forest Management Plan 2004-2013 increase the number and size of formal reserves in the jarrah forest to 634,850 ha which represents 40.5 percent of jarrah forest managed by CALM, and 22.8 percent of the pre-European area of jarrah forest. Under the State Government's forest policy, all old-growth jarrah forest in Western Australia is excluded from timber harvesting (however it can be cleared for mining in State Agreement Act leases). Old-growth jarrah forest represents 14 percent of the current distribution of jarrah forest (Conservation Commission 2004). Figure 1 shows the distribution of existing and proposed formal conservation reserves at the whole-of-forest scale and Figure 3 is an example of the distribution of formal and informal reserves, and fauna habitat zones, at the landscape scale.

At the landscape and stand scales, informal reserves provide old-growth forest or relatively undisturbed habitat. Approximately 132,610 ha of land vested in the Conservation Commission are allocated to informal reserves (Conservation Commission 2004). Informal reserves in State forest are composed of patches of old-growth forest, river and stream zones, travel route zones, Diverse Ecotype Zones (DEZ), less well reserved vegetation complexes, poorly reserved forest ecosystems and linkage zones. Because of the linear form (river and stream zones, travel route zones) or small patch occurrence (old-growth forest, DEZ, less well reserved vegetation complexes, poorly reserved forest ecosystems) of most informal reserves, they create a fine-scale network of corridors and stepping stones of uncut forest that link from the stand scale to the landscape and whole-of-forest scales (Figure 3). Figure 4 shows a harvest plan for an area of jarrah forest at the stand scale. Typically, about 40 percent of a jarrah coupe consists of areas of various types that are not harvested during a particular cutting cycle. These areas may remain uncut (e.g. informal reserves) or be harvested at the next cut in 30 to 50 years.

Fauna habitat zones will contribute to connectivity by providing habitat at the stand and landscape scales. The Forest Management Plan 2004-2013 introduces the protection of between 50,000 and 55,000 ha of State forest and timber reserves that will be progressively set aside from timber harvesting as fauna habitat zones. Fauna habitat zones are to be 200 ha or more in area and will be dispersed across State forest, generally between two to four kilometres apart (Conservation Commission 2004). The size of these fauna habitat zones was based on data on the home range of the western ringtail possum and estimates of the viable sub-population size for medium sized mammals (50-200 individuals) (Burrows *et al.* 2001, Conservation Commission 2002). Fauna habitat zones will provide additional linkage as stepping stones between formal and informal reserves.

At the stand scale, habitat trees and other habitat elements retained on harvested coupes contribute to connectivity. These habitat elements are discussed under the section on *Stand structural complexity*.

Landscape heterogeneity

Landscape heterogeneity is the diversity, size and spatial arrangement of habitat patches. Landscape heterogeneity increases the likelihood that the diversity of resources required by some fauna species, such as hollows in large trees, food sources, and structural components of the vegetation, are all provided within an area of a size that is readily and efficiently accessed by populations of fauna.

Landscape heterogeneity in the jarrah forest exists as an outcome of natural variation in the landforms, soil and vegetation (Mattiske and Havel 2002), variability in natural disturbance regimes (Burrows *et al.* 1995, Lamont *et al.* 2002), variations in intensity of timber harvesting, and the range of silvicultural objectives and methods that have been applied over the period of exploitation of the jarrah forest for timber (Bradshaw *et al.* 1991; Bradshaw 1999, Heberle 1997). Maintenance of landscape heterogeneity in the jarrah forest is addressed through locating formal conservation reserves, informal reserves and fauna habitat zones in areas of importance for biodiversity conservation. The dispersion of harvesting in space and time, and forest structural goals, also contribute to landscape heterogeneity.

Many formal conservation reserves have been located in areas with important habitat values such as high biophysical naturalness and areas of old growth forest (Conservation Commission 2004, Commonwealth of Australia and the State of Western Australia 1999). Informal reserves similarly protect areas of high biodiversity value around granite outcrops, streams, swamps and woodlands (Hopper *et al.* 1996, Main 1987, Postle *et al.* 1991). Fauna habitat zones are designed to specifically target for protection areas of habitat of importance for fauna (Conservation Commission 2004).

In terms of assessing the effects of timber harvesting on landscapes, Seymour and Hunter (1999) suggest that the key issues to consider are (i) average disturbance frequency, or the area regenerated annually within the stand, (ii) size distribution of gaps, and (iii) spatial configuration of gaps. Natural patterns of disturbance should be considered when planning harvest patterns. Over the past 15 years about one percent of the jarrah forest has been harvested for timber each year, of which 28 percent was cut to gap (Bradshaw 2002). Sixty percent of gaps were less than 2 ha (Conservation Commission 2002). The total area annually cut to gap is estimated to decline by about 75 percent from the previous 4,000 ha/annum to a cut of about 500 – 1,000 ha/annum over the next ten years (Bradshaw 2002). Indeed, over the last two years the area cut to gaps was about 1,500 ha in 2001 and 1,050 ha in 2002 which represents approximately 0.08 percent per annum of the extant area of jarrah forest or less than 0.2 percent per annum of the area available for timber harvesting (CALM 2002, 2003). Areas cut to gaps have not been equally distributed across all parts of the landscape. Whilst the key issues identified by Seymour and Hunter have been considered in landscape level decisions for jarrah forest, natural disturbance patterns and their effects on stand and landscape structure is not well understood, and improved knowledge of natural disturbance regimes would assist in refining harvest planning to better provide for landscape heterogeneity.

As jarrah predominantly exists in mixed aged stands, structural goals for the whole forest have been based on disturbance intensity, rather than stand age or size class. These structural goals have specified at least 25 percent of the of jarrah forest currently vested in the Conservation Commission be in formal reserves (minimally disturbed), at least five percent in informal reserves (low disturbance), and no more than one percent converted to the establishment stage in each year (CALM 1994). For the year 2000 the actual figures were 38 percent in existing and proposed formal reserves, 9 percent in informal reserves and 0.25 percent converted to the establishment stage in that year (Conservation Commission 2002). Recent large increases in proposed conservation reserves and the advent of fauna habitat zones substantially reduce the amount of forest available for timber harvesting. The increases in reservation and fauna habitat zones complement the previous emphasis on structural goals as a means of achieving landscape heterogeneity.

Stand structural complexity

Maintenance of stand structural complexity contributes to providing a wide variety of structural components on the stand. This structural diversity caters for the requirements of the various species that occur in the forest. Structural components include hollows of various sizes and orientations that occur on large and small trees, logs of various sizes and in various decay states, and a range of vegetation structural components that provide shelter, protection and food sources.

At the stand scale, jarrah forest harvesting is categorized as thinning, shelterwood, gap or dieback cutting. These four types of cutting produce a range of stand densities and structures. Each type of cut is applied to achieve a distinct silvicultural objective:

- 'Thinning with retained habitat' is used to promote growth on the trees retained in the stand.
- 'Shelterwood with retained habitat' is a partial removal of the overstorey that reduces competition and is employed to encourage establishment and development of seedlings.
- 'Gap with retained habitat' is a removal of up to 90 percent of the overstorey that releases regeneration and allows advance growth to develop into later growth stages.
- 'Dieback cutting' is applied to reduce the risk of disease intensification on *Phytophthora cinnamomi* infested forest by retaining and promoting resistant species and individuals (Bradshaw 1985, CALM, 1995).

The intensity of the cut varies with the silvicultural objective, the initial structure of the stand, and the number of non-commercial trees that remain after harvesting. For example, gap cutting, which is the most intensive of the four types of harvesting, may result in a wide range of stand densities and structures. Where there are few non-commercial trees on the stand, only these trees and the primary and secondary habitat trees will remain after harvesting, and the cut will create a gap in the overstorey. However, where many non-commercial trees, including large marri, remain after harvesting in addition to the primary and secondary habitat trees, the stand will have a much greater density and structural complexity. The intensity of harvesting on gap cuttings ranges between these two extremes.

Shelterwood cutting generally results in a more complex stand structure than gap cutting. The resulting stand may have a basal area anywhere between the extremes of 6 m^2 ha^{-1} to more than 30 m^2 ha^{-1} of retained trees, depending on the number of non-commercial trees that are retained.

Thinning is usually applied to stands that were harvested early in the last century, and in many areas few of the original overstorey trees were retained in this earlier harvesting. Thinning tends to leave the larger trees on the stand but also retains a range of trees sizes. Consequently, thinning is generally associated with stands that have greater structural complexity than gaps, but less structural complexity than shelterwood cuttings.

Dieback cutting is applied to *Phytophthora cinnamomi* infested forest and aims to result in the retention of at least 15 m^2 ha^{-1} of basal area, where this is available. The resulting stand structure is highly variable and dependent on the level of impact of *Phytophthora cinnamomi* on the site.

The eastern jarrah forest is naturally more open than the western forest because of the lower rainfall in the eastern forest, and the stocking of retained trees in harvested areas is less in the eastern forest than that described above for the western forest.

Selectively cut stands result from any of the types of cutting, where there are a large number of non-commercial trees remaining after harvesting.

Depending on the structure of the stands and status of regeneration in a proposed harvest area, all four types of cutting may occur as a mosaic, particularly in forest with a history of past harvesting. The resulting mosaic is itself highly variable and there is no typical mosaic pattern. Figure 4 is one example of the mosaic of harvesting and reserves in the jarrah forest. This mosaic of harvesting types contributes to structural diversity within stands and landscapes. In the past 15 years approximately 70 percent of cutting has resulted in mixed age stands dominated by trees in the larger size classes, and approximately 30 percent was gap cutting that resulted in stands dominated by regrowth (Bradshaw 2002).

In addition to the mosaic created by timber harvesting, stand structural complexity will be maintained in jarrah forest, including that managed for timber production, through various strategies (Conservation Commission 2002). These include:

- Informal reserves among timber harvesting areas: old-growth forest, river and stream zones, travel route zones, Diverse Ecotype Zones (DEZ), reservation of particular vegetation complexes, forest ecosystems and linkage zones;
- Fauna habitat zones dispersed throughout the forest;
- Temporary Exclusion Areas (TEAS)(15 to 60 years exclusion);
- Areas where the method or intensity of harvesting is modified to suit Visual Landscape Management objectives;
- Retention of 30 percent of each second-order catchment in the intermediate and low rainfall zones, either unharvested or with a retained basal area of at least 15 m^2 ha^{-1}, for a period of at least 15 years following harvesting;

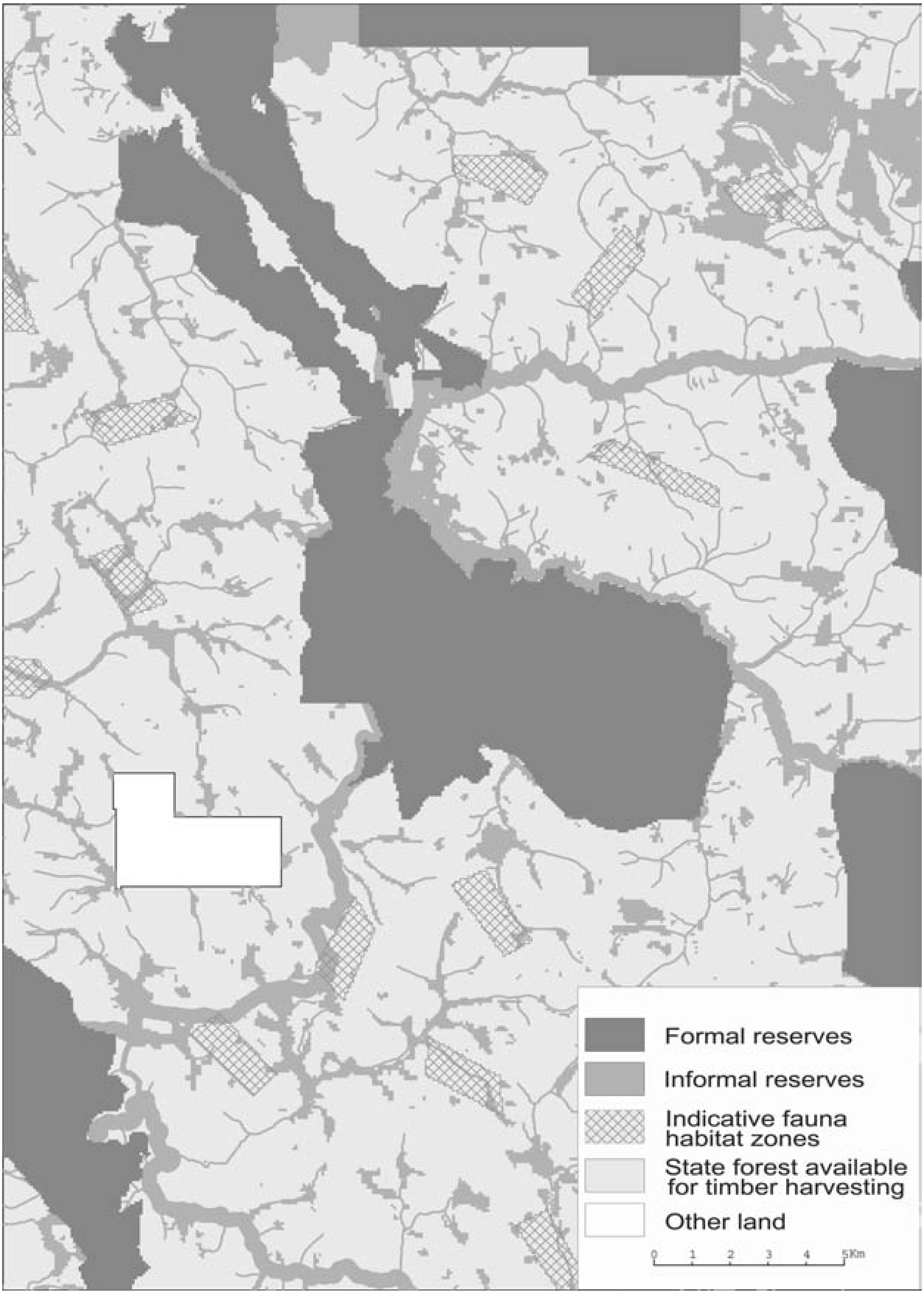

Figure 3. The network of existing and proposed formal conservation reserves, informal reserves and indicative fauna habitat zones at the landscape scale (Conservation Commission 2004). State forest and timber reserves available for timber harvesting are indicated. Other land not managed by the Department of Conservation and Land Management is also shown. The area shown is about 90,000 hectares and approximately 40 km to the east of Perth in the northern jarrah forest.

Figure 4. An example of the mosaic of harvesting and reservation in the jarrah forest at the stand scale. Harvesting consists of gap cutting, shelterwood cutting, thinning, selectively cut areas, and areas with a mosaic of types of cutting. Variations in the type and intensity of the cut produce variation in the stand structural complexity. Unlogged areas in this cutting cycle are informal reserves (river and stream zones, Diverse Ecotype Zones) and other unharvested areas. A range of other types of informal reserves can also occur in harvest areas. The area is approximately 2,750 ha of Hakea forest block in the northern jarrah forest.

- Retention of primary and secondary habitat trees, large marri and other non-commercial trees in all harvested areas;
- Retention of shelterwood trees during the regeneration establishment phase and the retention of crop trees in areas where the objective is to promote growth on retained trees;
- Relatively long rotation length, with most jarrah forest planned at more than 175 years.

Although supporting structural complexity at the stand scale, jarrah silviculture, particularly gap creation in areas cut to shelterwood over the last 15 years, could diminish stand structural complexity. Similarly the harvesting of TEAS strips in stands with a high proportion of earlier gap treatments could diminish structural complexity on particular stands. The impact of timber harvesting on structural complexity needs to be monitored and if the measures in the Forest Management 2004-2013 do not adequately provide for stand structural complexity then management practices will need to be reviewed.

Research priorities

A range of strategies is in place for the jarrah forest to provide habitat for hollow-dependent species across a range of spatial scales. The priorities for research in this area should be to provide knowledge that (i) assists in the assessment of the validity and necessity of these strategies and (ii) assists in the development and refinement of these strategies for providing hollows.

The strategies that we have discussed can be categorised as those that are in place to fulfil a range of biodiversity conservation functions and those that have been more specifically targeted at hollow-dependent species. This latter category, habitat trees and fauna habitat zones, therefore warrant specific attention. In the case of habitat trees, the types of trees required by hollow-dependent species has been well established, however the number of trees required for retention on stands can only be determined and assessed in the context of the habitat function provided by the surrounding forest including reserves and fauna habitat zones. Further research into the number of habitat trees required on stands would assist the development of forest management. Assessment of the validity and necessity of these strategies would be aided by the development of clear goals. Development of such goals requires knowledge of viable population sizes, population densities of jarrah forest species, and home range sizes and the numbers of hollows used by these species. As complete knowledge of the requirements of all hollow-dependent fauna is not currently available, adaptive management is appropriate. Studies of the relationship between population densities of hollow-dependent fauna and the densities of hollows in the forest would inform this process. Similarly, the role of fauna habitat zones, their size, configuration, and location is a worthy subject for adaptive management and research.

Although studies of any or all of the hollow-dependent species can provide useful knowledge, there are efficiencies in focusing research on species in the jarrah forest that are most likely to be impacted by the past and present harvesting activities. The species indicated by Abbott and Whitford (2002) as most likely to be impacted (Table 1), and the large and long-lived hollow-using cockatoos and owls, are useful species to study as they are most likely to provide crucial information that would enable refinement and development of strategies for conservation of hollow-dependent fauna.

Natural disturbance patterns and their effects on stand and landscape structure are not well understood, and improved knowledge of natural disturbance regimes would assist in refining harvest planning to better provide for landscape heterogeneity.

The work of Faunt (1992) and Williams and Faunt (1997) has provided data on the occurrence of logs on the ground and the abundance of hollows in logs. Although the current availability of hollow logs is not a concern in the jarrah forest, and the Forest Management Plan 2004-2013 requires all hollow logs be retained, further work on the long-term recruitment of ground logs, their size distribution and their decay, combined with knowledge of the use of ground logs by forest mammals, would provide a knowledge base for the long-term management of the hollow logs.

Conclusion

Management of the jarrah forest includes a range of strategies at the stand, landscape and whole-of-forest scales, which contribute to conservation of hollow-bearing trees and the fauna that use them. These strategies are consistent with the principles of connectivity, landscape heterogeneity, and stand structural complexity. The strategies operate across spatial scales and spread risk.

The available knowledge of hollows and hollow-dependent fauna species is informative and has aided the development of guidelines for retention of habitat trees at the stand scale. However, knowledge of the population densities and home range sizes of jarrah forest fauna and the numbers of hollows used by these species is insufficient to allow quantification of target population density for fauna species and target density of habitat trees. Quantitative targets or goals are an important part of an adaptive process as they help to provide transparency and a basis for discussion and modelling of strategies. Without refinements that lead to more specific goals for hollow-tree retention, management of hollow trees must rely on risk-spreading strategies such as those currently in place.

Acknowledgements

We thank I. Abbott, L. McCaw, N. Burrows, and three anonymous referees for commenting on a draft of this paper, Greg Strelein for providing data from the Jarrah Forest Inventory, P. Collins, M. Robinson and D. Winter for preparing maps, K. Faunt, W. McComb, M. Craig, A. Wayne, I. Abbott, G. Strelien and R. Johnstone for contributions to this work, and M. Williams and J. McGrath, for support, comments, and advice.

References

Abbott, I. 1998. Conservation of the forest red-tailed black cockatoo, a hollow-dependent species, in the eucalypt forests of Western Australia. *Forest Ecology and Management* 109: 175-185.

Abbott, I. and Whitford, K. 2002. Conservation of vertebrate fauna using hollows in forests of south-west Western Australia: Strategic risk assessment in relation to ecology, policy, planning, and operations management. *Pacific Conservation Biology* 7: 240-255.

Abbott, I. and Burrows, N. 2004. Monitoring biodiversity in jarrah forest in south-west Western Australia: the Forestcheck initiative. Pp. 947-58 in *Conservation of Australia's forest fauna* (second edition), edited by D. Lunney. Royal Zoological Society of NSW, Mosman, NSW.

Ambrose, G.J. 1982. An ecological and behavioural study of vertebrates using hollows in eucalypt branches. PhD thesis, La Trobe University, Melbourne.

Armstrong, J.A. and Abbott, I. 1997. Habitat trees and logging - ecology of the Brush-tailed Phascogale in jarrah forest. Unpublished report. Forest Science Library[8], Department of Conservation and Land Management, Kensington, W.A.

Bailey, C. 1996. Western Shield - bringing wildlife back from the brink of extinction. *Landscope* 11(4): 41-48.

Bootle, K. 1983. Wood in Australia - *Types, properties and uses*. McGraw-Hill Book Company, Sydney.

Bradshaw, F.J. 1985. Treemarking and silviculture in the jarrah forest. A training brief for operational staff. Silviculture branch Department of Conservation and Land Management, Western Australia.

Bradshaw, F.J. 1999. Trends in silvicultural practices in the native forests of Western Australia. Australian Forestry, 62: 255-264.

Bradshaw, F.J. 2002. Advice to the Department of Conservation and Land Management on the report: "Towards ecologically sustainable forest management in Western Australia. A review for Draft Jarrah Silviculture Guideline". F.J. Bradshaw, Forest Consultant. June 2002. Unpublished Report for the Department of Conservation and Land Management, Western Australia.

Bradshaw, F.J., Adams, R., Sneeuwjagt, R., Low, K., Havel, J.J., Bartle, J.R. and Stoneman, G.L. 1991. The jarrah forest: a case study in multiple use. Pp. 1-21 in *Forest Management in Australia* edited by F.H. McKinnell, E.R. Hopkins and J.E.D. Fox. Surrey Beatty & Sons Pty Ltd., Chipping Norton, NSW.

Brown, S., Mo, J., McPherson, J. K. and Bell, D. T. 1996. Decomposition of woody debris in Western Australian forests. *Canadian Journal of Forest Research* 26: 954 - 966.

Buchanan, J.B., Irwin, L.L. and McCutchen, E.L. 1993. Characteristics of spotted owl nest trees in the Wenatchee National Forest. *Journal of Raptor Research* 27: 1-7.

Burrows, N. and Christensen P. 2002. Long-term trends in native mammal capture rates in a jarrah forest in south-western Australia. *Australian Forestry* 65(4): 211 - 219.

Burrows, N., Friend, G., Morris, K., Stoneman, G., Craig, M., and Williams, M. 1994. An integrated study of the effects of timber harvesting on the jarrah forest ecosystem unpublished report, Department of Conservation and Land Management, Forest Science Library[8], Kensington, Western Australia.

Burrows, N.D., Ward, B. and Robinson, A.D. 1995. Jarrah forest fire history from stem analysis and anthropological evidence. *Australian Forestry* 58: 7-16.

Burrows, N., Christensen, P., Hopper, S., Ruprecht, J., Young, J. 2001. Ministerial condition 11: Panel report Part 1. Report for the Department of Conservation and Land Management. Western Australia.

Burrows, N., Christensen, P., Hopper, S., Ruprecht, J., Young, J. 2002. Towards ecologically sustainable forest management in Western Australia. A review for Draft Jarrah Silviculture Guideline 1/02. Panel report part 2 for the Conservation Commission of Western Australia. http://www.conservation.wa.gov.au/files/docs/15.pdf

Calder, T.G., Golding, B.G. and Manderson, A.D. 1983. Management for arboreal species in the Wombat State Forest. Monash University, Graduate School of Environmental Science Environmental Report No. 16.

CALM. 1987a. Regional Management Plan 1987-1997, Northern Forest Region. Department of Conservation and Land Management, Perth Western Australia.

CALM. 1987b. Regional Management Plan 1987-1997, Central Forest Region. Department of Conservation and Land Management, Perth Western Australia.

CALM. 1987a. Regional Management Plan 1987-1997, Southern Forest Region. Department of Conservation and Land Management, Perth Western Australia.

CALM 1994. Forest Management Plan 1994-2003. Department of Conservation and Land Management. Perth, Western Australia.

CALM 1995. Silvicultural practices in the jarrah forest. Silvicultural Guidelines 1/95. Department of Conservation and Land Management, Como, Western Australia, Australia.

CALM 2002. Annual Report 2001-2002. Department of Conservation and Land Management, Western, Australia.

CALM 2003. Annual Report 2002-2003. Department of Conservation and Land Management, Western, Australia.

Christensen, P.E.S., Liddelow, G., and Hearn, R. 2001. Fauna assessment in timber harvesting coupes in Western Australia. Unpublished report. Forest Science Library[8], Department of Conservation and Land Management, Kensington, W.A.

Churchward, H. M. and Dimmock, G. M. 1989. The soils and landforms of the northern jarrah forest. Pp. 13-21 in *The Jarrah Forest* edited by B. Dell, J. J. Havel and N. Malajczuk. Dordrecht, Kluwer Academic Publishers.

Commonwealth and W.A. RFA Steering Committee 1998. Comprehensive Regional Assessment – Maps. Vol. 2. A Regional Forest Agreement for Western Australia. Commonwealth and Western Australian Regional Forest Agreement Steering Committee, Canberra.

Commonwealth of Australia and the State of Western Australia 1999. Regional Forests Agreement for the south-west forest region of Western Australia, May 1999. The Commonwealth of Australia and the State of Western Australia.

Conservation Commission 2002. Draft forest management plan. July 2002. Conservation Commission of Western Australia. http://www.conservation.wa.gov.au/downloads.htm

Conservation Commission 2004. Forest Management Plan 2004-2013. December 2003. Conservation Commission of Western Australia. http://www.conservation.wa.gov.au/downloads.htm

[8]Forest Science Library holdings (http://science.calm.wa.gov.au/conslib.php) are available through inter-library loan.

de Tores, P. 1999. Control and ecology of the red fox in Western Australia - prey response to 1080 baiting over large areas. Unpublished report prepared for Environment Australia, National Feral Animal Control Program, April 1999. ISF06638. Department of Conservation and Land Management, Perth, Western Australia.

Dickman, C.R. 1991. Use of trees by ground-dwelling mammals: implications for management. Pp. 125-136 in *Conservation of Australia's Forest Fauna* edited by D. Lunney. Royal Zoological Society of NSW, Mosman, NSW.

Driscoll, A. W. 2000. A comparison between the insulation qualities of hollows in trees and nests in balga (*Xanthorrhoea preissii*): study relating to habitat used by the western ringtail possum (Pseudocheirus occidentalis). Honours thesis, School of Resource and Environmental Management, Department of Forestry, The Australian National University, Canberra.

EPA, 2003. Proposed Forest Management Plan 2004-2013, Conservation Commission of Western Australia, Report and recommendations of the Environmental Protection Authority. Environmental Protection Authority, Perth Western Australia, Bulletin 1113, September 2003.

Faunt, K. 1992. Formation, frequency and longevity of hollows in jarrah: Interim report. Unpublished report. Forest Science Library[8], Department of Conservation and Land Management, Kensington, W.A.

Florence, R.G. 1996. Silviculture of un-even aged forests, Chapter 10, in *Ecology and Silviculture of Eucalypt Forests*. Australian National University CSIRO Publishing Collingwood.

Forests Department 1982. 50 years of forestry in Western Australia, Forests Department of Western Australia.

Friend, J.A., Thomas, N.D. 2003. Conservation of the numbat (Myrmecobius fasciatus). Chapter 31, in *Predators with pouches: the biology of carnivorous marsupials* edited by M. Jones, C. Dickman, and M. Archer. CSIRO Publishing, Collingwood Victoria.

Gehlbach, F.R. 1994. Nest-box versus natural-cavity nests of the eastern screech-owl: an exploratory study. *Journal of Raptor Research* 28: 154-157.

Gellman, S.T. and Zielinski, W.J. 1996. Use by bats of old-growth redwood hollows on the north coast of California. *Journal of Mammalogy* 77: 255-265.

Gentilli, J. 1989. Climate of the jarrah forest. Pp. 23-40 in *The Jarrah Forest* edited by B. Dell, J. J. Havel and N. Malajczuk. Dordrecht, Kluwer Academic Publishers.

Gibbons, P. and Lindenmayer, D. 2002. *Tree hollows and wildlife conservation in Australia*. CSIRO Publishing, Collingwood, Victoria.

Gibbons, P., Lindenmayer, D.B., Barry, S.C. and Tanton, M.T. 2002. Hollow selection by vertebrate fauna in forests of southeastern Australia and implications for forest management. *Biological Conservation* 103: 1-12.

Grove, T. S., O'Connell, A.M., and Dimmock, G.M. 1986. Nutrient changes in surface soils after an intense fire in jarrah (*Eucalyptus marginata* Donn ex Sm.) forest. *Australian Journal of Ecology* 11: 303-317.

Gysel, L.W. 1961. An ecological study of tree cavities and ground burrows in forest stands. *Journal of Wildlife Management* 25: 12-20.

Haseler, M. and Taylor, R. 1993. Use of tree hollows by birds in sclerophyll forest in north-eastern Tasmania. *Tasforests* 5: 51-56.

Heberle, G. 1997. Timber harvesting of Crown land in the south-west of Western Australia: An historical review. *CALMScience* 2: 203-224.

Hopper, S.D., Harvey, M.S., Chappill, J.A., Main, A.R., and Main, B.Y. 1996. The Western Australian biota as Gondwanan heritage - a review. Pp 1-46 in *Gondwanan Heritage: Past, Present and Future of the Western Australian Biota* edited by S.D. Hopper, J.A. Chappill, M.S. Harvey and A.S. George. Surrey Beatty & Sons. Chipping Norton, NSW.

How, R.A. 1981. Population parameters of two congeneric possums, *Trichosurus spp*. in north-eastern New South Wales. *Australian Journal of Zoology* 29: 205-215.

How, R.A. and Hillcox, S.J. 2000. Brushtail possum, *Trichosurus vulpecula*, populations in south-western Australia: demography, diet and conservation status. *Wildlife Research* 27: 81-90.

How, R.A. and Kerle, J.A. 1995. Common Brushtail Possum *Trichosurus vulpecula*. In *The Mammals of Australia* edited by R. Strahan. Reed books, Chatswood, NSW. p 273-275.

Inions, G. 1985. The interactions between possums, habitat trees and fire. BSc Honours thesis, Department of Forestry, The Australian National University, Canberra.

Inions, G., Tanton, M.T. and Davey, S.M. 1989. Effect of fire on the availability of hollows in trees used by the common brushtail possum, *Trichosurus vulpecula* Kerr, 1792, and the ringtail possum, *Pseudocheirus peregrinus* Boddaerts, 1785. *Australian Wildlife Research* 16: 449-458.

Jackson, J.A. 1978. Competition for Cavities and Red-cockaded Woodpecker Management. In *Endangered Birds Management Techniques for Preserving Threatened Species* edited by S.A. Temple. University of Wisconsin Press, Madison, Wisconsin.

Jones, B. 1995. Western Ringtail Possum *Pseudocheirus occidentalis*. Pp 252-254 in *The Mammals of Australia* edited by R. Strahan. Reed books, Chatswood, NSW, Australia.

Jones, B. and Hillcox, S. 1992. A survey of the possums *Trichosurus vulpecula* and Pseudocheirus *occidentalis* and their habitats in forest at Ludlow, Western Australia. *The Western Australian Naturalist* 139- 150.

Jones, B.A., How, R.A. and Kitchener, D.J. 1994. A field study of Pseudocheirus occidentalis (Marsupialia: Petauridae). 11. Population studies. *Wildlife Research* 21: 175-87.

Joseph, L., Emison, W.B. and Bren, W.M. 1991. Critical assessment of the conservation status of red-tailed black cockatoos in south-eastern Australia with special reference to nesting requirements. *Emu* 91: 46-50.

Kehl, J. and Borsboom, A. 1984. Home range, den tree use and activity patterns in the greater glider, *Petauroides volans*. Pp. 229-236 in *Possums and Gliders* edited by A. Smith and I. Hume. Surrey Beatty and Sons, and Australian Mammal Society, Chipping Norton, NSW.

Kerle, J.A. 1984. Variation in the ecology of *Trichosurus*, its adaptive significance. Pp. 115-128 in *Possums and Gliders* edited by A. Smith and I. Hume. Surrey Beatty and Sons, and Australian Mammal Society, Chipping Norton, NSW.

Kinnear, J. E., Onus, M.L., and Sumner, N.R. 1998. Fox control and rock-wallaby population dynamics – II. An update. *Wildlife Research* 25: 81-88.

Lamont, B. B., Ward, D. J.; Eldridge, J.; Korczynskyi, D.; Colangelo, W. I.; Fordham, C.; Clements, E.; Wittkuhn, R. 2003. Believing the Balga: a new method for gauging the fire history of vegetation using grasstrees. Pp 147-169 in *Fire in Ecosystems of South-west Western Australia: Impacts and Management* edited by I. Abbott and N. Burrows. Backhuys Publishers, Leiden.

Lindenmayer, D.B., Cunningham, R.B., Tanton, M.T., Smith, A.P. and Nix, H.A. 1991a. Characteristics of hollow-bearing trees occupied by arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-east Australia. *Forest Ecology and Management* 40: 289-308.

Lindenmayer, D.B., Cunningham, R.B., Tanton, M.T., Nix, H.A. and Smith, A.P. 1991b. The conservation of arboreal marsupials in the montane ash forests of the Central Highlands of Victoria, south-eastern Australia: III. The habitat requirements of Leadbeater's Possum *Gymnobelideus leadbeateri* and models of the diversity and abundance of arboreal marsupials. *Biological Conservation* 56: 295-315.

Lindenmayer, D.B. and Franklin, J.F. 2002. Conserving forest diversity – A comprehensive multiscaled approach. Island Press, Washington, D.C.

Lindstedt, S.L., Miller, B.J., and Buskirk, S.W. 1986. Home range, time and body size in mammals. *Ecology* 67, 413- 418.

Long, J.L. 1990. The breeding biology of four species of parrots in the south of Western Australia. Agriculture Protection Board, Department of Agriculture of Western Australia Technical Series No. 6.

Luck, G.W. 2001. The demography and cooperative breeding behaviour of the rufous treecreeper, *Climacteris rufa*. *Australian Journal of Zoology* 49: 515-537.

Luck, G.W. 2002. The habitat requirements of the rufous treecreeper (*Climacteris rufa*). 1. Preferential habitat use demonstrated at multiple spatial scales. *Biological Conservation* 105: 383-394

Main, B.Y. 1987. Ecological disturbance and conservation of spiders: Implications for biogeographic relics in southwestern Australia. Pp 88-97 in *The Role of Invertebrates in Conservation and Biological Survey. Proceedings of a Workshop held during the 18th Scientific Conference of the Australian Entomological Society* edited by J.D. Majer. Department of Conservation and Land Management, Western Australia.

Mattiske, E.M., and Havel, J.J. 2000. Vegetation mapping of south-west forest region of Western Australia, prepared for CALMScience, Department of Conservation and Land Management and Environment Australia.

Mattiske, L., and Havel, J.J. 2002. Delineation of landscape conservation units in the southwest region of Western Australia. Prepared for the Department of Conservation and Land Management by Mattiske Consulting and Havel Land Consultants.

Maxwell, S., Burbidge, A.A., and Morris, K. (eds) 1996. The 1996 Action Plan for Australian Marsupials and Monotremes. Endangered Species Program, Project No. 500. ANCA, Canberra.

McComb, W.C. 1992. Prospectus. Hollow dynamics in jarrah forest: A conceptual model and identification of research opportunities. In; *Mapping potential habitat for vertebrates in forests of Western Australia*, Final report for the Department of Conservation and Land Management, Western Australia. Bill McComb, 1994. Forest Science Library[8], Department of Conservation and Land Management, Kensington, Western Australia.

McComb, W., Faunt, K., Bradshaw, F.J. and Biggs, P. 1994. Hollow availability for selected vertebrates in jarrah forests, Western Australia. In; *Mapping potential habitat for vertebrates in forests of Western Australia*, Final report for the Department of Conservation and Land Management, Western Australia. Bill McComb, 1994. Forest Science Library[8], Department of Conservation and Land Management, Kensington, Western Australia.

McComb, W. and Lindenmayer, D.B. 1999. Dying dead and down trees. Pp. 335-372 in: *Managing Biodiversity in Forest Ecosystems* edited by M. L. Hunter, Jr. Cambridge University Press, Cambridge, UK. 698pp.

McComb, W.C. and Noble, R.E. 1981. Nest-box and natural cavity use in three mid-south forest habitats. *Journal of Wildlife Management* 45: 93-101.

Medway, L. and Marshall, A.G. 1970. Roost-site selection among flat-headed bats (*Tylonycteris spp.*). *Journal of Zoology, London* 161: 237-245.

Menkhorst, P.W. 1984a. The application of nest boxes in research and management of possums and gliders. Pp. 517-525 in *Possums and Gliders* edited by A.P. Smith and I.D. Hume. Surrey Beatty and Sons and the Australian Mammal Society, Chipping Norton, NSW.

Menkhorst, P.W. 1984b. Use of nest boxes by forest vertebrates in Gippsland: acceptance, preference and demand. *Australian Wildlife Research* 11: 255-264.

Morris, K., Johnson, B., Orell, P., Gaikhorst, G., Wayne, A., Moro, D. 2003. Recovery of the threatened chuditch (*Dasyurus geoffroii*): a case study. Chapter 30 in *Predators with pouches: the biology of carnivorous marsupials*, edited by M. Jones, C. Dickman, and M. Archer. CSIRO Publishing, Collingwood, Victoria.

Newton, I. 1994. The role of nest sites in limiting the numbers of hole-nesting birds: a review. *Biological Conservation* 70: 265-276.

Nilsson, S. G. 1984. The evolution of nest-site selection among hole-nesting birds: the importance of nest predation and competition. *Ornis Scandinavica* 15: 167-175.

Oldroyd, B.P., Lawler, S.H. and Crozier, R.H. 1994. Do feral honey bees (*Apis mellifera*) and regent parrots (*Polytelis anthopeplus*) compete for nest sites? *Australian Journal Ecology* 19: 444-450.

Pinkowski, B. C. 1976. Use of tree-cavities by nesting eastern bluebirds. *Journal of Wildlife Management* 40: 556-563.

Postle, A.C., Majer, J.D. and Bell, D.T. 1991. A survey of selected soil and litter invertebrate species from the northern jarrah (*Eucalyptus marginata*) forest of Western Australia, with particular reference to soil type, stratum. Seasonality and the conservation of forest fauna. Pp 193-203 in *Conservation of Australia's Forest fauna* edited by D. Lunney. Royal Zoological Society of New South Wales, Mosman, NSW.

Rhind, S.G. 1996. Habitat tree requirements and the effects of removal during logging on the marsupial Brush-tailed Phascogale (*Phascogale tapoatafa tapoatafa*) in Western Australia. *Western Australian Naturalist* 21: 1-22.

Rhind, S.G. 1998. Ecology of the brush-tailed phascogale in jarrah forest of southwestern Australia. PhD, Murdoch University Western Australia.

Rose, A.M. 1996. Territorial and breeding behaviour of the rufous treecreeper (*Climacteris rufa*) in the Stirling Ranges, Western Australia. *Corella* 20: 55-61.

Saunders, D.A. 1979. The availability of tree hollows for use as nest sites by white-tailed black cockatoos. *Australian Wildlife Research* 6: 205-216.

Saunders, D.A., Smith, G.T. and Rowley, I. 1982. The availability and dimensions of tree hollows that provide nest sites for cockatoos (Psittaciformes) in Western Australia. *Australian Wildlife Research* 9: 541-556.

Scarff, F., Rhind, S.G. and Bradley, J.S. 1998. Diet and foraging behaviour of brush-tailed phascogales in the jarrah forest of southwestern Australia. *Wildlife Research* 25: 511-526.

Schmidt, L. 1979. Nest-hollow temperatures and water-turnover of nestling galahs (*Cacatua roseicapilla*, Vieillot 1817). BSc Honours thesis, University of Western Australia, Perth.

Seymour, R.S. and Hunter M.L. , Jr. 1999. Principles of ecological forestry. Pp 22-61 in *Managing Biodiversity in Forest Ecosystems* edited by M. L. Hunter, Jr. Cambridge University Press, Cambridge, UK. 698pp.

Soderquist, T.R. 1993. Maternal strategies of *Phascogale tapoatafa* (Marsupialia: Dasyuridae). II. Juvenile thermoregulation and maternal attendance. *Australian Journal of Zoology* 41: 567-576.

Soderquist, T.R. 1995a. Spatial organization of the arboreal carnivorous marsupial *Phascogale tapoatafa*. *Australian Journal of Zoology* 237: 385-398.

Soderquist, T.R. 1995b. Phascogale *Phascogale tapoatafa*. Pp. 104-106 in *The Mammals of Australia* edited by R. Strahan. Reed books, Chatswood, NSW, Australia.

Spencer, R.D. 1992. Application of modern inventory techniques in the forests of Western Australia. Western Australian Department of Conservation and Land Management, CALM Occasional Paper No. 1/92.

Stoneman, G.L., Crombie, D.S., Whitford, K., Hingston, F.J., Giles, R., Portlock, C.C., Galbraith, J.H. and Dimmock, G.M. 1997. Growth and water relations of *Eucalyptus marginata* (jarrah) stands in response to thinning and fertilization. *Tree Physiology* 17: 267-274.

Stoneman, G.L., Rayner, M.E. and Bradshaw, F.J. 1997. Size and age parameters of nest trees used by four species of parrot and one species of cockatoo in south-west Australia: Critique. *Emu* 97: 94-96.

Taylor, R.J. and Savva, N.M. 1988. Occurrence of potential nest trees and their use by birds in sclerophyll forest in north-east Tasmania. *Australian Wildlife Research* 15: 637-645.

Tidemann, C.R. and Flavel, S.C. 1987. Factors affecting the choice of diurnal roost site by tree-hole bats (microchiroptera) in South-Eastern Australia. *Australian Wildlife Research* 14: 459-473.

Traill, B. J. 1995. Use of tree hollows by two sympatric gliding possums. Chapter 4, in Co-existence and competition in a community of forest vertebrates. PhD thesis, Department of Ecology and Evolutionary Biology, Monash University, Melbourne.

Traill, B.J. and Coates, T.D. 1993. Field observations of the Brush-tailed Phascogale *Phascogale tapoatafa* (Marsupialia: Dasyuridae). *Australian mammalogy* 16: 61-65.

Turner, B.J. 1998. An appraisal of methods and data used by CALM to estimate wood resource yields for the South-West Forest Region of Western Australia. Published by the joint Commonwealth and Western Australian Regional Forest Agreement (RFA) Steering Committee.

Turner, B.J., Ferguson, I., and Fitzpatrick, N. 1999. Report by the expert panel on the calculation of a sustainable sawlog yield for the jarrah and karri forests of WA. Sustainable yield expert panel report. Commonwealth and Western Australian Regional Forest Agreement (RFA). www.calm.wa.gov.au/forest_facts/pdf_files/rfa_panel.pdf.

Van Balen, J.H., Booy, C.J.H., Van Franeker, J.A. and Osieck, E.R. 1982. Studies on hole-nesting birds in natural nest sites 1. Availability and occupation of natural nest sites. *Ardea* 70: 1-24.

van der Ree, R., Soderquist, T.R., and Bennett, A.F. 2001. Home-range use by the brush-tailed phascogale (*Phascogale tapoatafa*) (Marsupialia) in high-quality, spatially limited habitat. *Wildlife Research* 28, 5: 517-525.

Vonhof, M.J., and Barclay, R.M.R. 1996. Roost-site selection and roosting ecology of forest-dwelling bats in southern British Columbia. *Canadian Journal of Zoology* 74: 1797 –1805.

Wardell-Johnson, G. 1986. Use of nest boxes by mardos, *Antechinus flavipes leucogaster*, in regenerating karri forest in south Western Australia. *Australian Wildlife Research* 13: 407-417.

Wayne, A., Ward, C., Rooney, J. and Wheeler, I. 2000. The immediate impacts of timber harvesting and associated activities on the Ngwayir (*Pseudocheirus occidentalis*) in the Jarrah forest of Kingston State Forest Block. Internal report, Forest Science Library[8], Department of Conservation and Land Management, Kensington, Western Australia.

Whitford, K.R. 2001. Dimensions of tree hollows used by birds and mammals in the jarrah forest: improving the dimensional description of potentially usable hollows. *CALMScience* 3: 499-511.

Whitford, K.R. 2002. Hollows in jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*) trees: I. Hollow sizes, tree attributes and ages. *Forest Ecology and Management* 160: 201-214.

Whitford, K.R. and Williams, M.R. 2001. Survival of jarrah (*Eucalyptus marginata* Sm.) and marri (*Corymbia calophylla* Lindl.) habitat trees retained after logging. *Forest Ecology and Management* 146: 181-197.

Whitford, K.R. and Williams, M.R. 2002. Hollows in jarrah (*Eucalyptus marginata*) and marri (*Corymbia calophylla*) trees: II. selecting trees to retain for hollow-dependent fauna. *Forest Ecology and Management* 160: 215-232.

Williams, M.R. and Faunt, K. 1997. Factors affecting the abundance of hollows in logs in jarrah forest of south-western Australia. *Forest Ecology and Management* 95: 153-160.

Woinarski, J.C.Z. 1973. A comparison of the ground-nesting of two species of Pardalote. *Emu* 74: 219-222.

Woinarski, J.C.Z. and Bulman, C. 1985. Ecology and breeding biology of the Forty-Spotted Pardalote and other Pardalotes on North Bruny Island. *Emu* 85: 106-120.