

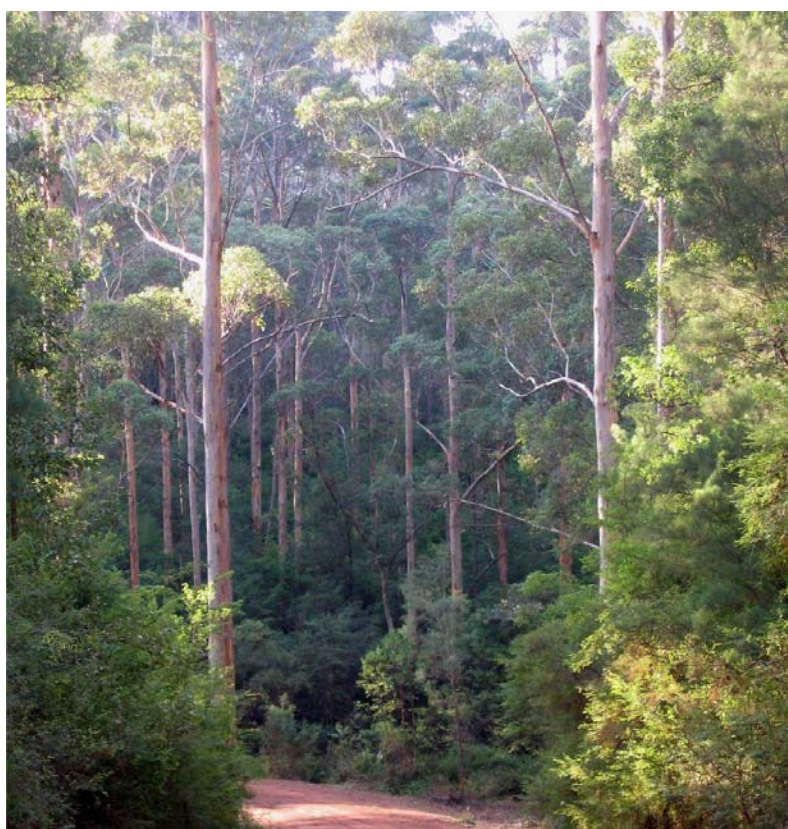
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August 2015

Reference material for karri forest silviculture

FJ Bradshaw



Department of Parks and Wildlife
Forest and Ecosystem Management Division



Department of
Parks and Wildlife



Department of Parks and Wildlife
Locked Bag 104
Bentley Delivery Centre WA 6983
Phone: (08) 9219 9000
Fax: (08) 9334 0498

www.dpaw.wa.gov.au

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This report/document/publication was prepared by Mr F J (Jack) Bradshaw

Questions regarding the use of this material should be directed to:

Senior Silviculturalist
Ecosystem Health Branch
Department of Parks and Wildlife
Locked Bag 104
Bentley Delivery Centre WA 6983
Phone: (08) 97255972
Email: danielle.wiseman@dpaw.wa.gov.au

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Reference material for karri forest silviculture

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Contents

Acknowledgments	vi
Introduction	vii
1 The karri forest	9
1.1 Forest types and communities.....	9
1.2 Climate	12
1.3 Geology and Soils	14
1.4 Hydrology	14
1.5 Influence of fire	16
1.6 Flora and Fauna	18
2 Silvics	19
2.1 Taxonomy.....	19
2.2 Flowering and seed production	20
2.3 Emergence and early survival	22
2.4 Seedlings.....	23
2.5 Stand development.....	24
2.5.1 Saplings, poles and mature trees.....	24
2.5.2 Competition	27
2.5.3 Site potential.....	29
2.6 Response to fire	32
3 Forest structure	35
3.1 Stand and forest structure	35
3.2 Stand initiation and structural diversity	40
3.3 Structural attributes of stand development stages	42
3.3.1 Establishment.....	43
3.3.2 Juvenile	43
3.3.3 Immature	44
3.3.4 Mature	45
3.3.5 Senescent	46
3.3.6 Old-growth.....	47
4 Sustainable forest management.....	48
4.1 Conservation of biodiversity	50
4.1.1 Conservation strategies.....	50
4.1.2 Biodiversity of the karri forest	51
4.1.3 Structural complexity	53

4.1.4	Genetic conservation.....	60
4.1.5	Predator control.....	61
4.1.6	Fire and biodiversity	61
4.1.7	Climate change	62
4.2	Maintenance of productive capacity	63
4.2.1	Factors influencing silvicultural practice	63
4.2.2	Fire management	78
4.2.3	Summary of silvicultural methods in the karri forest.....	81
4.2.4	Climate change	85
4.3	Maintenance of ecosystem health and vitality	86
4.3.1	Pests and disease	86
4.3.2	Fire management	87
4.3.3	Climate change	88
4.4	Conservation and maintenance of soil and water.....	89
4.4.1	Soil	89
4.4.2	Water.....	89
4.4.3	Nutrient recycling.....	91
4.4.4	Climate change	92
4.5	Maintenance of forests contribution to the global carbon cycle	92
4.6	Integrated management	93
Appendices		95
Glossary.....		115
References.....		120
Appendices		
Appendix 1	Changes in silvicultural practice	95
	Silvicultural systems.....	95
	Past silvicultural practice.....	96
Appendix 2	Vertebrate fauna that may be affected by timber harvesting	102
	Arboreal marsupials and terrestrial mammals.....	103
	Bats of the karri forest.....	105
	Hollow nesting birds.....	107
	Birds (not hollow using).....	110
	Reptiles and amphibians.....	111
	Fish	112

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Introduction

The karri (*Eucalyptus diversicolor*) forest is confined to the lower south-west of Western Australia. Renowned for the great size of the trees and its spectacular beauty, the forest is important for conservation, tourism, recreation and timber production. Most of the forest is on public land, vested in the WA Conservation Commission and managed on its behalf by the Department of Parks and Wildlife, hereafter referred to as the department. Approximately two thirds of the forest is reserved for conservation purposes and one third is managed for multiple uses.

Management of the forest available for timber harvesting is guided by the Forest Management Plan (FMP), a document prepared by the Department of Parks and Wildlife for the Conservation Commission with public input. The current FMP applies to the period 2014 to 2023. Silvicultural guidelines provide the next level of policy detail for the implementation of silviculture in the FMP.

This Reference Material has been prepared to provide a summary of the scientific and observational information that underpins the silvicultural guideline, manuals and procedures. It is intended to provide reference material for policy makers, foresters and interested members of the public. This document is intended to provide an understanding of why particular practices may be employed. Silvicultural guidelines, manuals and procedures are provided in separate documents.

Silvicultural practice in WA has been developed on the basis of scientific studies, observation of the response to natural or deliberate events, and understanding of the general principles of silvicultural response and forest dynamics. Preferred practice is also influenced by operational practicality, market opportunities for forest products, public opinion and ministerial or government policy.

A variety of practices have been used in the forests of the south-west of Western Australia over the past 130 years, and for the last 80 years have been generally well documented. Examples of the outcome of most past practices still exist in the field. Observation of these examples, and the benefit of hindsight together with specific scientific studies, is an invaluable resource for predicting the response to current activities and disturbances.

While the general background and ecological information presented here refers to the whole of the karri forest, information relating to silvicultural practice applies only to the forest available for timber harvesting. The discussion relates both to the karri forest type and to karri as a species.

1 The karri forest

1.1 Forest types and communities

Karri is a wet sclerophyll forest occurring in the higher rainfall areas of the lower south-west of Western Australia (Figure 1). Approximately 35 per cent of the forest occurs as pure karri, with the remainder occurring in mixture with marri (*Corymbia calophylla*) and to a lesser extent with jarrah (*E. marginata*). In the cooler south-east of its range it also occurs in mixture with red tingle (*E. jacksonii*), yellow tingle (*E. guilfoylei*) and Rates tingle (*E. brevistylis*) (Bradshaw *et al.* 1981; Wardell-Johnson *et al.* 1996; Bradshaw *et al.* 1997a). Karri may occur in intimate mixture with these species or as a mosaic of different forest types. Within what is described as the 'main karri belt', only one third of the area is occupied by karri. Karri forest types occur within a broader mosaic of other forest types and woodlands (mainly jarrah dominated), heathlands, sedgeland and swamps (Smith 1974; Christensen 1992; Bradshaw *et al.* 1997a). Karri occurs in more than 2000 discrete patches with 30 per cent of the area of the karri forest occurring in patches of less than 100 ha (Bradshaw *et al.* 1997b).

The area of karri forest at the time of European settlement is estimated at 232,000 ha. Eighty two per cent of the original extent remains, with approximately 174,000 ha occurring on public land (Conservation Commission 2013b).

The karri forest has been depicted in broad scale outline mapping in Fraser (1882), Ednie-Brown (1896), Lane-Poole (1921) and Kessell (1928), with various estimates of the area over which it occurred. From 1916 to the 1930s, ground surveys undertaken by Forests Department land classification teams refined the area of forest, but these maps were not published.

The first comprehensive and detailed maps of the karri forest were produced from the interpretation of aerial photography (API) in the 1950s and 1960s. This mapping covered all land tenures within an area from Mundaring to Albany (excluding the Swan coastal plain). The API mapping described forest type (the dominant and secondary canopy species), canopy density, structure and mature height. Subsequent mapping identified stands of pure karri, mapped isolated occurrences on the south coast, and mapped areas from which karri had been cleared for agriculture. A simplified version of this was published as *Forest Associations, Height of Tallest Native Vegetation and Karri Distribution Before European Settlement* by the Department of Conservation and Land Management in 1997 (Bradshaw *et al.* 1997a). Minor corrections were made to the database of these maps during the previous FMP 2004-2013 (Conservation Commission 2004).

Smith (1974) amalgamated the API data to produce maps of structural formations (after Specht (1970) at a coarser resolution. This was part of a larger series of maps of vegetation formations and associations for the south-west (Beard 1972-80), which has since been updated (Shepherd *et al.* 2001).

The southern forest area within which karri occurs has also been mapped on the basis of vegetation complex. Vegetation complexes are primarily based on the attribution of vegetation associations to a mapping unit developed from a combination of land form-soil units and rainfall. Because this is a classification within a continuum and because the

mapping is based on landform, the boundaries of these complexes are not readily identifiable in the field with any degree of precision. There can be considerable spatial variation between the inferred vegetation of this mapping and the actual occurrence of karri as mapped from API. A feature of the mapping is that it extends over cleared land to represent the original vegetation. The work was first completed for the northern jarrah forest (Hedde *et al.* 1980) and later extended to include all of the area covered by the Regional Forest Agreement (RFA) (Mattiske *et al.* 1998; Havel 2000). There are 23 Vegetation Complexes associated with karri forest types.

Vegetation Complexes have been amalgamated to create ecological vegetation systems, nine of which apply to karri forest (Havel *et al.* 1999). Christensen *et al.* (2005) used groupings of vegetation complexes to define fauna habitat types. Of the 54 types identified across the south-west, seven apply to the karri forest itself.

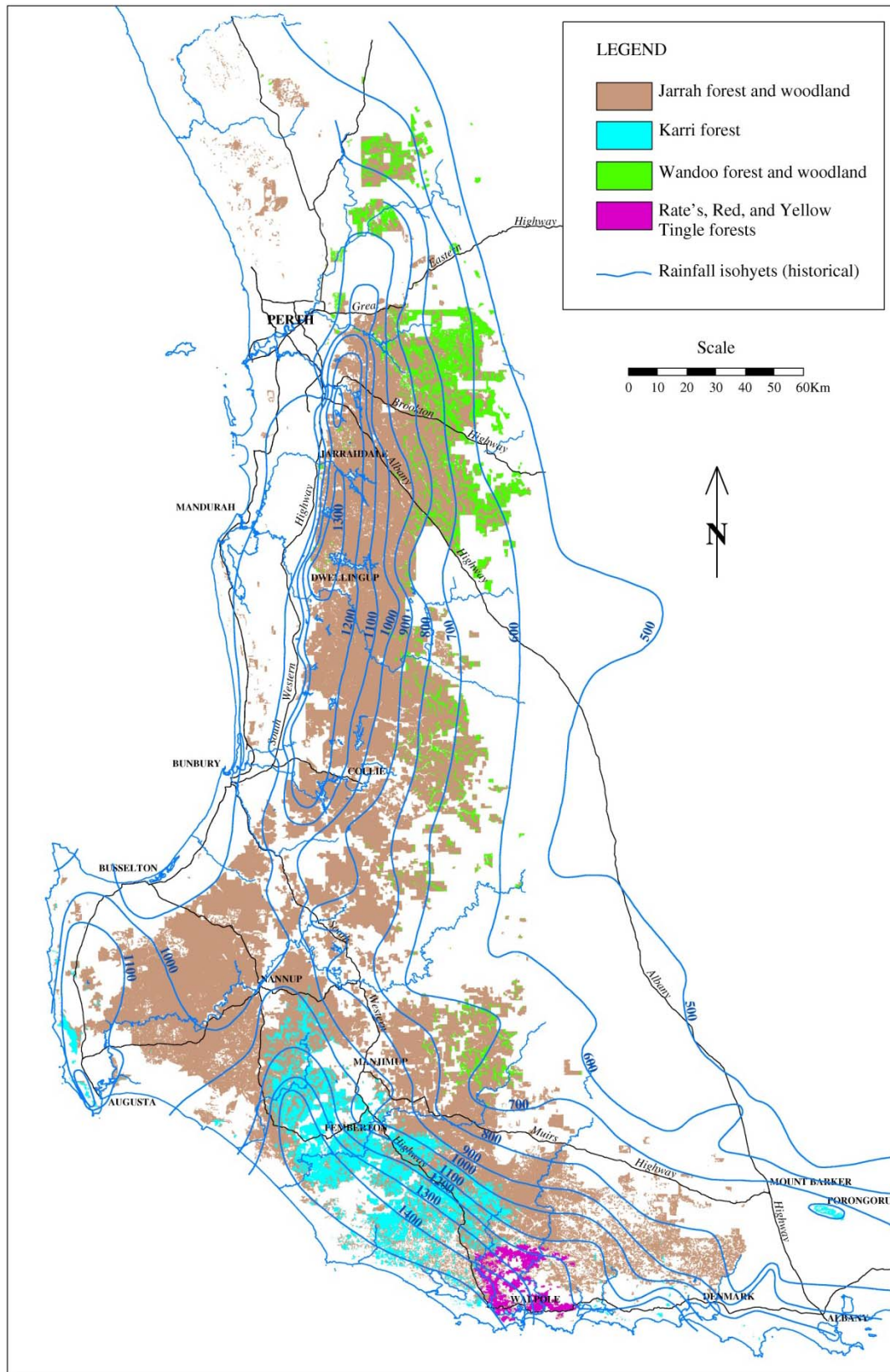


Figure 1: Distribution of major forest types on public land in the south-west. Isohyets are based on data to 1979 (Hayes et al. 1981).

Vegetation complexes are also used as the basis for mapping 30 Landscape Conservation Units (referred to in the FMP 2014-2023 as Landscape Management Units or LMUs), 15 of which contain karri forest (Mattiske *et al.* 2002).

A more detailed classification of the karri forest based on floristic attributes was developed by Inions *et al.* (Inions *et al.* 1990). This described five community groups and thirteen community types on the basis of the occurrence of non-herbaceous perennials in the understorey, a classification shown to be independent of disturbance. While too labour intensive to be used for broad scale mapping, the classification is useful for defining 'like types' for karri research projects.

As a basis for determining reservation targets for the RFA, vegetation complexes were combined with forest associations and geographic location to produce 26 forest ecosystems, five of which relate to karri forest (Bradshaw *et al.* 1997d; RFA 1998b Map 12). A further jarrah forest ecosystem (the Whicher Scarp) was added in 2011.

1.2 Climate

The south-west of Western Australia has a Mediterranean climate, and the occurrence of the karri forest is influenced by both annual and seasonal rainfall. Karri forest occurs where the long term median rainfall exceeds 1000mm per annum and the summer evaporation is less than 500mm (Gentili 1989) (see Figure 2 - based on average rainfall data to 1972). Summer (December, January, and February) rainfall historically exceeds 40mm per month. As a comparison, the annual rainfall at Dwellingup and Pemberton (prime jarrah and karri forest respectively) is similar at approximately 1200mm, but the Pemberton summer rainfall is double that of Dwellingup (Bureau of Meteorology). Karri outliers exist in areas with a median rainfall as low as 600mm, but these sites are associated with particular situations of high moisture availability.

The above data is based on long term rainfall patterns. Since the mid-1970s there has been a significant reduction in mean annual rainfall and in the median summer rainfall (Table 1), the longer term impact of which remains unknown at present.

Churchill (1968) suggests that the relative frequency of karri to marri has declined over the past 5000 years with the least favourable periods for karri being at 2500-450 BC and again at 500-1500 AD. He attributes these changes to changing rainfall rather than fire. This is disputed by Newsome and Pickett (1993), who argue that Churchill's methodology and interpretation are flawed and there is no evidence to suggest a significant change in vegetation or climate during that period.

Nevertheless, genetic data suggests that the outliers at Karridale, Rocky Gully and the Porongurup's were the first to become isolated from the main karri belt (Coates *et al.* 1989).

Table 1: Changes in rainfall at key weather stations in the main karri belt.

		Manjimup	Pemberton	Northcliffe
Median annual rainfall (mm)	before 1970 ^a	1059	1271	1441
	1970-2012	931	1091	1235
	Change since 1970	-12 per cent	-14 per cent	-14 per cent
Median summer rainfall (mm)	before 1970	45	73	75
	1970-2012	37	49	48
	Change since 1970	-18 per cent	-33 per cent	-36 per cent

^a Records begin in 1916 for Manjimup, 1942 for Pemberton and 1916 for Northcliffe

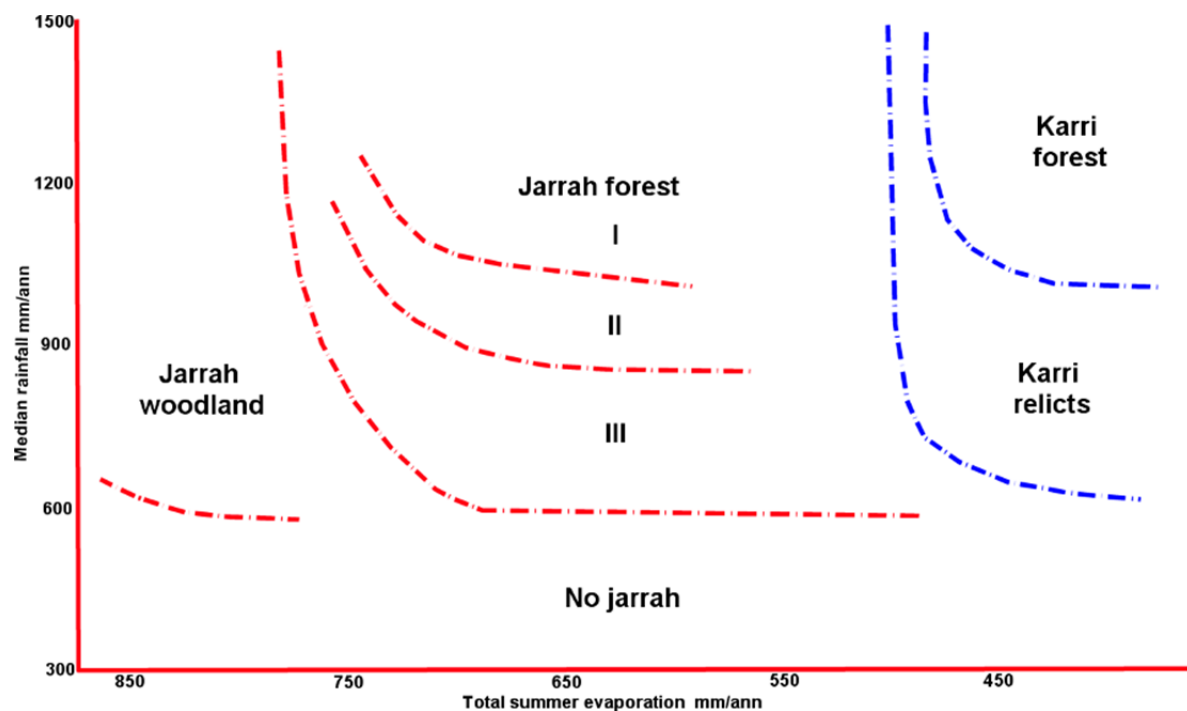


Figure 2: The distribution of karri forest in relation to median rainfall and summer evaporation. (Source: derived from Gentilli (1989)).

1.3 Geology and Soils

Within the area of suitable climate, karri distribution is determined by soil type. The main karri occurrence is on granite-gneiss bedrock with intrusions of basic rock that give rise to red earths and a red-brown to yellow-brown podsollic loamy sand. The pattern of distribution is largely determined by the degree of dissection of the lateritic duricrust of the Darling Plateau and the level of subsequent deposition. In the northwest of its range, karri occurs on the red earths (commonly referred to as karri loams) in the deeply incised valleys of the Donnelly River and its tributaries. Mixtures of karri/marri occur on the podsollic soils of the mid-slopes, giving way to jarrah on the lateritic gravels of the uplands. Further south where the lower elevation duricrust has been more extensively eroded, karri occurs from the valleys to the ridge tops. Still further south and south-east, much of the landscape has been inundated in the past resulting in extensive areas of sandy depositional podsoles supporting heath and sedgelands. In these areas, karri occurs on the red soils where the major rivers have dissected the sandy 'flats' and on the soils formed on the emergent granite-gneiss inselbergs (McArthur *et al.* 1975; Bradshaw *et al.* 1981; Christensen 1992). The major outlier at the Porongurup's is an example of the latter. The other major outlier on the west coast at Karridale occurs on limestone derived soils. Some outliers along the south coast occur on sands.

1.4 Hydrology

In high rainfall karri forest, the vegetation intercepts an estimated 10-20 per cent of rainfall and transpires about 60-70 per cent. Streamflow is typically 5-10 per cent of rainfall, 90 per cent occurring between May and October. According to Borg *et al.* (1988b), up to 85 per cent of the annual streamflow is from subsurface runoff (throughflow), with about five per cent from surface runoff and ten per cent from groundwater discharge. These observations were based on rainfall data to 1979 (Hayes *et al.* 1981).

The karri forest receives between 100 and 250 kg/ha/yr of salt deposited from rainfall depending on distance from the coast. Most of the salt is flushed through the system in these high rainfall areas, resulting in a current soil salt storage of 50-350 tonnes per ha and groundwater salinity of less than 1000 mg/L. In still forested areas, stream salinity is usually well below 500 mg/L and frequently less than 100 mg/L (Borg *et al.* 1988b).

In some catchments, streamflow in the southern forests has declined by as much as 50 per cent since records began (Figure 3). In the drier part of the karri belt, groundwater has dropped two to four metres since 1976 (April Road South and Lewin control catchment bores). Significant reductions in streamflow in most catchments occurred from 1975 following a step-down in rainfall from that time. Streamflow from most catchments in the karri forest have continued to decline since then but at a slower rate, at least until 2009 (DOW 2009). Data from four southern catchments studied in the 1980s indicate an annual streamflow reduction of 30mm per 100mm of rainfall below 1000mm (Bari *et al.* 1993). Data from the undisturbed Wattle catchment (mainly mature karri forest) illustrates the rapid reduction in streamflow when annual rainfall is less than 1100mm, reaching negligible flows at 700mm (Figure 4).

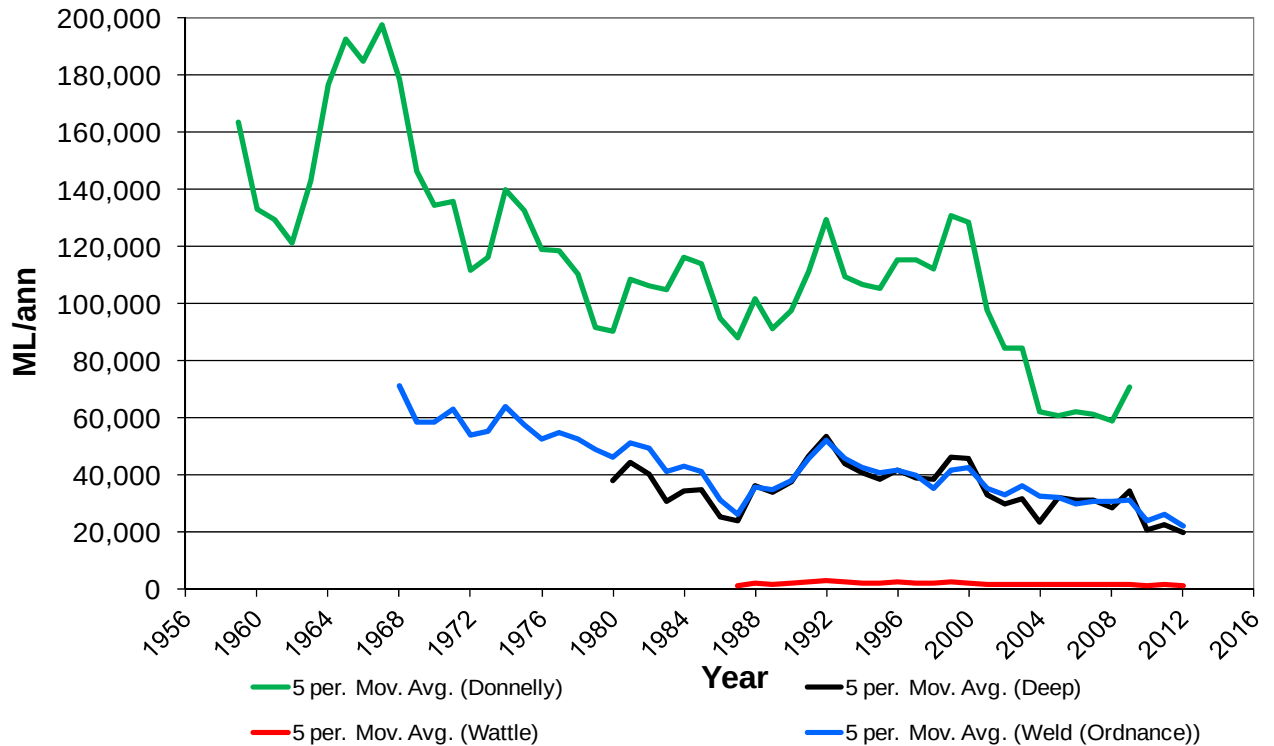


Figure 3: Five year rolling average streamflow in four mainly-forested catchments in the southern forests. The Donnelly River catchment has been extensively harvested. The Wattle catchment consists largely of virgin karri forest (Source data: Department of Water).

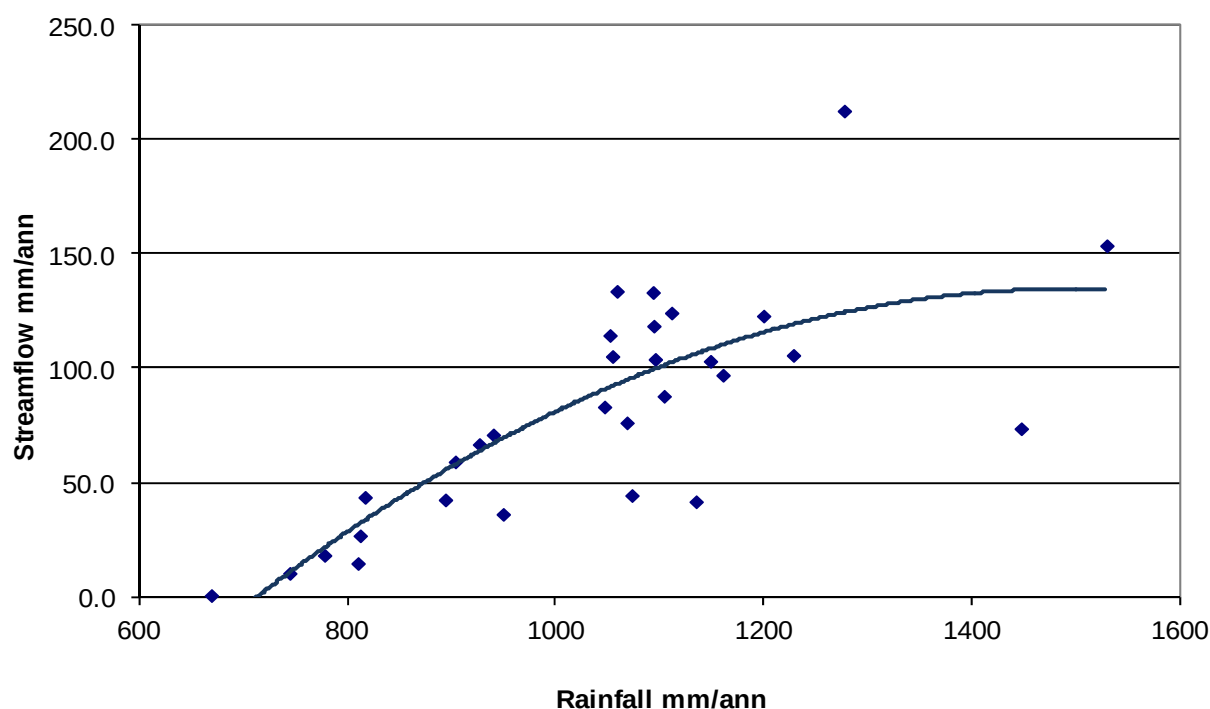


Figure 4: Streamflow versus rainfall in the undisturbed Wattle catchment from 1983 to 2012 showing the rapid decline in streamflow where the annual rainfall falls below 1100mm. (Source data: Department of Water).

1.5 Influence of fire

The south-west of Western Australia is a fire prone environment. There are about 110 rain-free days in the drier part of the year (Soil Dryness Index (SDI) >50mm) in the karri forest compared to about 80 days in the wet eucalypt forests of Tasmania and about 165 days in the northern jarrah forest¹. Left unburnt, the karri forest will accumulate up to 60 tonnes of flammable litter per ha after 60 years (McCaw *et al.* 2002). The dry summers of the Mediterranean climate in which it grows makes it possible for the litter fuel to sustain fire and for it to spread on about 80 days in the spring/summer/autumn period and for about 25 days at high intensity (McCaw *et al.* 2003). Lightning-caused fires are a regular occurrence throughout the karri forest (Underwood 1978).

Pre-European fire frequency is not known with certainty but Christensen and Annels (1985) have suggested that it may be in the order of twenty years based on the absence of temperate rainforest elements and late successional mammal species. However, the observation of early explorers and settlers suggest that it was also variable (Talbot 1973). It is believed that aboriginal use of the karri forest was limited but it is likely that relatively frequent fires would have encroached from the southern and eastern boundaries where aboriginal populations were higher, and from the regular aboriginal travel routes through the forest such as that now known as the Deeside Road (Dortch *et al.* 2010). While the first

¹ Based on long-term records.

European settlers arrived in the vicinity in the 1850s, there was little European activity in most of the karri forest till the 1920s when extensive 'group settlement' followed the establishment of sawmills a decade earlier (Gabbedy 1988).

In a study of six virgin karri stands that originated from 1630 to 1720, persistent regeneration events² or fire scarring events occurred within these stands at an average interval of 48 years prior to 1850, and every 17 years from 1850 until 1920. The absence of fire scars and the reduced number of regeneration events in the earlier period may be influenced by the age of the stands in this period, or be a result of more frequent but less severe fires (Rayner 1992b). In contrast to that, age mapping of the whole karri forest indicates that there were many more significant regeneration events in the period 1750 to 1850 than since that time (Bradshaw *et al.* 1997c). The reason for this apparent contradiction may be that the number of persistent regeneration cohorts may underestimate fire events or even regeneration events in immature stands, where the persistence of regeneration is limited by the vigour of the older (immature) cohort (Bradshaw *et al.* 1997b). This is discussed in more detail in Section 3.2.

The understorey of the karri forest supports a higher proportion of fire sensitive 'seeders' (regenerate by seed) relative to 'sprouters' (regenerate from lignotubers) than do drier sites that are subjected to more frequent fires. Approximately 50 per cent of the species are seeders but they constitute more than 80 per cent of the understorey vegetation cover, compared to only 10 per cent in the jarrah forest (Christensen *et al.* 1975). Regenerating from soil stored seed, most of the understorey seeders are relatively short lived. There are a number of longer lived fire sensitive species, such as *Trymalium floribundum*, *Chorilaena quercifolia*, *Acacia pentadenia* and *Bossiaea aquifolium* subsp. *Laidlawiana*, which can live up to 70 years (Figure 5). These appear to have replaced the fire-vulnerable rainforest species that no longer occur here (Christensen 1992; Burrows *et al.* 2003).

Vascular species richness and plant density decrease with time since fire. Some species are favoured by frequent fire and others by long periods between fire; while some will be disfavoured by both too frequent and too infrequent fire (McCaw *et al.* 2002; Burrows *et al.* 2003; Wardell-Johnson *et al.* 2004). The impact of fire on the karri forest has been summarised by Christensen and Abbott (1989).

The role of fire in regeneration and its influence on forest structure is discussed in Sections 2.2 and 3.2. Its role in nutrient cycling is discussed in Section 4.4.3.

² Indicative of fire occurrence. See Section 4.2 for further discussion of the role of fire in regeneration.



3 years old



30 years old



50 years old



75 years old

Figure 5: The change in understorey structure in relation to time since fire. By about 60 years most of the longest lived understorey species that regenerated with the fire have reached the end of their life and collapsed.

1.6 Flora and Fauna

The south-west of Western Australia is recognised as one of the world's 25 global biodiversity hotspots with approximately 7400 species of vascular plants, half of which are endemic. However, the high rainfall forest areas (annual rainfall 800-1500mm) are relatively species poor (Myers *et al.* 2000; Hopper *et al.* 2004). The karri forest is not particularly flora-rich and there are few endemic and no rare flora confined to the karri forest itself. The communities with which karri forest is associated (heaths, woodlands, granite outcrops and swamps) are much more species rich, especially in the south-east of the distribution (Christensen 1992; Hopper *et al.* 1992 ; RFA 1998b Map 5).

While there is no detailed list of flora species that are confined to the karri forest itself, a list of ferns, monocots, dicots, lichens, and larger fungi associated with southern forests (including southern jarrah forest and heaths) is provided by Christensen (1992).

There are approximately 450 vertebrate species in the south-west, about 25 of which are endemic. About 300 species occur within the broader forest area, about 60 per cent of which

are in locations or vegetation types that are not affected by timber harvesting and the majority of these are in the jarrah forest.

A comprehensive list of vertebrate species in the south-west forests, along with their probability of occurrence in mapped fauna habitat types (including karri forest types) and their vulnerability to the impacts of burning and timber harvesting, is contained in the forest Fauna Distribution Information System (FDIS) (Christensen *et al.* 2005).

In common with other areas, invertebrates are less well understood, but sources of this information relevant to southern forests has been summarised by Christensen (1992).

Further detail of the biodiversity of the karri forest is given in Section 4.1.2 and Appendix 2.

2 Silvics

2.1 Taxonomy

Karri is a tall, smooth barked tree typically reaching a height of 60-70m with rare individuals exceeding 80m. It is a member of the subgenus *Symphyomyrtus*, the largest subgenus of *Eucalyptus* that includes species that vary from large trees to mallee. The only other member of the subgenus with which it associates is yellow tingle (*E. guilfoylei*).

Genetic analysis of karri taken from samples throughout the main range and the outlying populations (Figure 1) indicates that karri has relatively high genetic diversity, including that within the small isolated populations. Within the main karri belt there is little genetic differentiation with the exception of the lower Warren area, which is more closely related to the Karridale outlier on the west coast. There is little differentiation between the main belt and the south coast outliers at William Bay and Mt Many Peaks, but the outliers at Rocky Gully and the Porongurup's form separate populations (Coates *et al.* 1989).

Family/provenance trials indicate superior height growth of provenances from the main karri belt, having higher within-provenance variation than between-provenance variation. An exception to this was the high performance of the outlying Porongurup provenance. It has been suggested that the poorer performance of other outliers may in part be due to inbreeding within these small populations (Mazanac *et al.* 1993).

2.2 Flowering and seed production

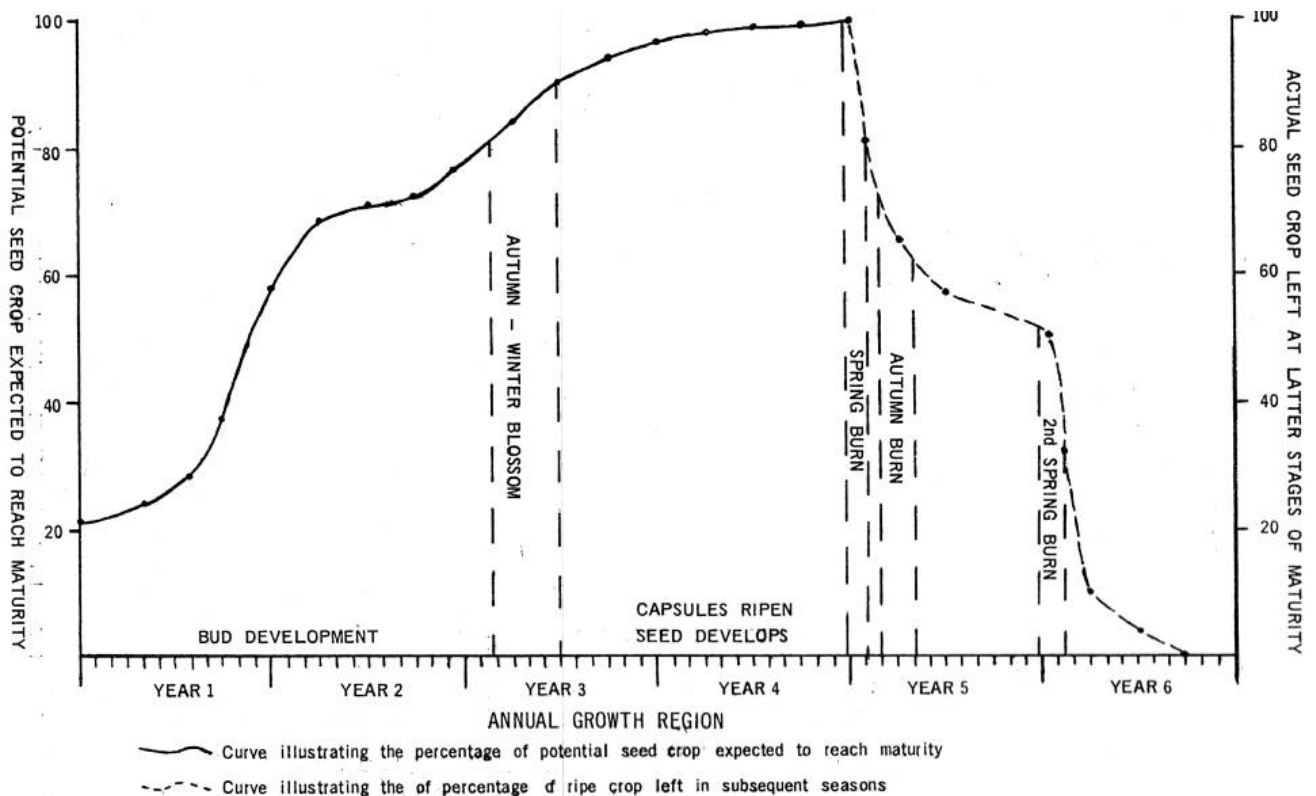


Figure 6: The generalised floral cycle of karri (White 1974b).

The karri floral cycle takes four to five years from bud initiation to the dissemination of seed (Figure 6). While a general seed crop occurs at about 5 year intervals, the pattern observed in a study from the 1950s to 1970s was that of alternating light and heavy crops (Loneragan 1979). The cycle is however more complex than this and up to five stages of floral development may exist on a tree at any one time, though the major crop will dominate others. Where a second cycle begins in a year following the first, the second cycle will 'catch up' with the first. Flowering that occurs prior to April will mature in the following summer, while later flowering will not mature until the summer after that. The various stages of development are shown in Figure 7. While there is a tendency towards synchronised flowering across the whole forest, there is within-stand and regional variation.

Karri floral cycle	Year and stages in the floral cycle (as observed 1958-1963)
YEAR JAN. FEB. MAR. APR. MAY JUNE JULY AUG. SEPT. OCT. NOV. DEC. YEAR 0 1963 INFLORESCENCE PIN BUDS CYLINDRICAL BUDS 1 2 PLUMP BUDS 2 3 BLOSSOM OPERCULA 2 3 HYPANTHIA 3 4 IMMATURE CAPSULES 4 5 MATURE CAPSULES 4 5 SEED 4 5 EMPTY CAPSULES 5 6	0-1 Inflorescence and pin buds of 1958
	0-2 Cylindrical buds of 1959
	2-3 Clavate (plump) buds of 1960. Blossom with separation of opercula from hypanthia (immature receptacles). Stage 1 fruit hypanthia of 1960
	3-4 Stage 2 fruit – immature capsules of 1961
	4-5 Stage 3 fruit – mature capsules and seed of 1962

Figure 7: Stages and timing of floral development (Loneragan 1979).

While karri that is thinned and had fertiliser applied may produce seed as young as seven years of age, seed production from unthinned immature stands is low, almost certainly due to the high competition for light and moisture in these stands. In mature stands, seed production increases with crown cover until crown cover reaches 20 per cent. Below 20 per cent, crown cover seed fall can be calculated from estimates of crown coverage, basal area and capsules per twig. Dominant and co-dominant trees are responsible for most seed production, especially in poor to moderate seed years (White 1974b; Loneragan 1979).

Under normal conditions karri produces about one seed per capsule, low by comparison with many eucalypts. Poor pollination appears to be the reason for this low number, with the introduction of bee hives increasing production by 25-50 per cent and manual pollination by 400 per cent. Purple-crowned Lorikeets are believed to be the important pollinators in karri (Christensen 1971a).

Loneragan (1979) found that cumulative deviation from annual rainfall is correlated with flowering and capsule production one and four years later respectively. Initiation of heavy crops was associated with spring/summer rainfall that is 40 per cent above average. Regular seed crops could only be expected when there is adequate soil moisture. However, the issue is complex. Cumulative rainfall has been in deficit since 1975, while the average spring/summer rainfall has remained largely unaltered, although it has exceeded 40 per cent of average on only one occasion since 1966. Despite that, the seed tree regeneration

program shows that a regular five year seed cycle has been maintained at least until 1994, after which seed tree regeneration largely ceased due to smaller coupe sizes and limitations on regeneration burning. Some seed tree regeneration was undertaken successfully each year from 1964 to 1994 (see Section 2.2). Nevertheless, increased stress resulting from reduced rainfall is likely to impact on seed supply. While formal monitoring has not been undertaken, there appears to have been a general decline in seed crops in recent years (Alan Seymour³, pers. comm.).

Low intensity fire has been shown to cause a reduction in all floral parts on the tree and a reduction in the initiation of floral parts over the subsequent two years. More severe fire that scorches the trees may cause the shedding of all floral parts and it may take several years to recover.

Seed shed from the capsules on the tree occurs as the mature capsules begin to dry out. About 40 per cent are shed in the first summer and autumn, with the remaining 60 per cent in the second summer/autumn (Figure 6). Moderate and intense fire stimulates seed fall with scorched branches shedding most of their seed within 2-10 days of the fire, with almost all seed shed within three weeks. Capsule opening on scorched branches appears to be a consequence of direct desiccation, while those on green branches are a result of the development of a fire-induced abscission layer. A milder fire resulted in little abscission layer formation and has little impact on seed-shed (Christensen 1971b).

Most seed is shed within one tree height, with the seed at tree height being half that beneath the seed tree. Suitable seed tree spacing for 65m trees is therefore 3 to 5 per ha (White 1974b).

Insects that either nip off or bore into the buds have been recorded as causing the loss of more than 75 per cent of the seed in some years, but their impact is sporadic. Seed predation by invertebrates is significant and the application of insecticide to seed can significantly increase germination, but the extent to which this prevents seed removal or damage to germinants is unclear (Christensen *et al.* 1979). The mass seed fall on burnt ground following fire minimises the foraging impact of invertebrates on seed availability.

A number of factors influence the production of seed including moisture availability and nutrition. These factors are discussed in some detail by Breidahl and Hewett (1995).

2.3 Emergence and early survival

The germination rate of karri is in the order of 95 per cent under nursery conditions, with germination occurring within 2-3 weeks. In field conditions, germination per cent has been shown to vary from about 2 to 32 per cent and tree survival (the number of seedlings that become established as a percentage of the seeds sown) from 0.6 to 6.3 per cent (Loneragan 1979). Field germination and early survival are influenced by a number of factors.

For karri germinants to survive, they require access to mineral soil. The removal of litter and understorey can be achieved by mechanical means or fire. Litter prevents access to soil and also has an allelopathic effect on germinants. Understorey creates shade and severe competition for moisture. Soil type has been shown to have no influence on germination

³ Manager, Silviculture, Forest Products Commission.

rates, but seedlings of red loams grew faster than those on gravely podsols. Seed bed has a significant influence on germination. Disturbed soil is superior to compacted soil or unburnt, undisturbed sites. The centre of ashbeds are often hydrophobic, resulting in poor germination, but germination rates on the edges of ashbed or on less intense ashbed is high. Natural seed fall into unburnt understorey, even with partial disturbance by timber harvesting, results in negligible establishment. Seedling growth on ashbeds is significantly higher than on other seedbeds (Breidahl *et al.* 1995). Loneragan and Loneragan (1964) showed that the ashbed response in karri seedlings was largely due to the increase in P from the burning of the litter and increases in N and P from soil heating. This is similar to that reported for *E. regnans* (Chambers *et al.* 1994).

Seed germinates after the first rains (typically) in April. April germination is highest (47 per cent) with a rapid deterioration until there is virtually no germination from June onwards as soil temperature drops (Breidahl *et al.* 1995). This may vary with the date of the opening rains. While seed fall following a spring burn is substantially higher than that from an autumn burn, predation and possibly desiccation of seed over summer results in lower total germination (Christensen 1970). Germination of karri is complete within the first three months of autumn/winter (White 1971). The survival of nursery stock is high if seedlings are planted once the top 25cm of soil is moist and prior to the end of July. This allows time for seedlings to become established prior to the summer 'drought'. While frost can kill young seedlings in their first year, it is relatively uncommon in the karri forest.

Damping-off fungi can affect germinants and while *Phytophthora cinnamomi* has been shown to attack karri roots in pot trials, it has not been observed as an issue in the field (White 1974b). Poor survival of seedlings has been observed on areas that have been adversely affected by the root pathogen *Armillaria luteobubalina*. Fungicide treatment of seed produced variable results, in some cases reducing germination. It is possible that it may have inhibited mycorrhizal attachment.

Browsing damage by quokkas (*Setonix brachyurus*) occurred in karri forest until 1935 where harvesting coupes were small and surrounded by unburnt forest (Stewart 1936, 1937). By 1938, quokkas had largely disappeared due to fox predation and browsing damage did not occur again in the karri forest until 2004, following successive years of fox baiting. The extent of damage to regeneration by quokka is spatially variable and is a concern only in areas where population numbers are high. While damage is normally confined to a distance of about 50m from a stream zone, in some cases it has occurred to least 150m from a forest edge (A. Seymour, pers. comm.).

2.4 Seedlings

Shoot growth of seedlings occurs between October and May (Skinner 1972). Seedlings may be successfully established following fire in otherwise undisturbed forest, or from 'seed trees' retained in harvested forest. The coincidence of burning and available seed is critical to the success of natural regeneration or seed tree operations. Successful regeneration can also be achieved with either fire or physical soil disturbance using artificial seed dispersal, or the

planting of 'wildlings'⁴, open-rooted or container-grown nursery stock. Planted seedlings respond positively to fertiliser application at time of planting, with most response due to phosphorus (P). High rates of nitrogen (N) decreased survival (Breidahl *et al.* 1995). The application of 0.03g of NPK (nitrogen, phosphorus and potassium) 'Red' fertiliser just before nursery dispatch has been shown to be as effective as 25g of DAP (di-ammonium phosphate) applied at planting time (Hewett 1991).

Karri seedlings are severely affected by competition from established understorey. Infill planting into sites with understorey as young as one year old have survival rates ranging from 23-82 per cent and their growth rate is about 10 per cent of established seedlings. Ripping that disturbs the soil as well as removing the understorey from the immediate planting site increases growth rate by four times. Shallow ripping is necessary to achieve satisfactory survival and growth on patches of soil that have been compacted by timber harvesting, but it also produces a positive response in undisturbed soils (Hewett 1992).

Growth and survival of seedlings are also negatively affected by retained trees, the impact being felt within two years of germination, by which time most seedlings beneath the crown of a retained tree will have died, most probably as a consequence of competition for moisture (Breidahl *et al.* 1995).

In the regeneration of mixed karri/marri stands, marri will regenerate from seed (if it is available) or from lignotubers. Lignotuber survival is high and the growth rate from marri lignotubers will exceed that of karri for the first few years, after which it is rapidly overtaken by karri (White 1971). While jarrah ground coppice generally has good survival, the survival of jarrah seedlings in mixed karri/marri/jarrah stands is more problematic due to the strong competition from both overstorey and understorey.

2.5 Stand development

2.5.1 Saplings, poles and mature trees

In the absence of competition from existing trees, regeneration grows rapidly, out-competing understorey that regenerates at the same time. Annual growth rate varies with soil type and seed bed type, but is generally in the order of 1 - 1.5m from age 2 - 10 years (White 1971). Differential growth rate, largely as a consequence of seedbed type, initiates dominance sorting between saplings, being reinforced at crown closure, which normally occurs from about 5 - 8 years of age. It has been observed in natural stands and in spacing trials that between 17 per cent and 30 per cent of stems remain dominated and suppressed regardless of stocking and do not remain as effective competitors (Schuster 1978; Bradshaw *et al.* 1991). The reason is unknown, but may be due to these trees coming from self-pollinated seed or due to their lack of attachment to mycorrhiza, which is known to be highly spatially variable. In natural or seed tree regeneration areas, by the time crown closure has occurred, just 3,000 individual stems may have survived from as many as 1,000,000 germinants.

⁴ Wildlings are seedlings that have been transplanted from areas with excessive numbers of seedlings originally established from seed.

Despite the fact that karri overtakes marri in mixed stands, marri continues to contribute to the dominant strata because the mixture is not homogeneous, but tends to occur as small scale patches (Hewett 1993).

Saplings have relatively strong apical dominance, abscising and shedding side branches as they become shaded and less efficient (Jacobs 1955). Despite the long clean boles evident in later life, karri is a relatively coarse-branched species compared to many others. Close spacing and a dense understorey play an important role in reducing branch size and encouraging branch shed.

Crown cover increases rapidly to reach the equivalent of the mature forest in ten years and then exceeds it by about 15 per cent at least for the next 50 years (Stoneman *et al.* 1988). The pole stage is reached at about 25-30 years of age when apical dominance begins to lessen, side branches become more persistent and the crown changes to a broader, more rounded shape. By this time, 400-500 dominant and co-dominant trees remain. Periodic annual volume increment peaks and trees have achieved about 75 per cent of their final height and basal area (Rayner 1992a) (Figure 8). Stem numbers continue to reduce rapidly until about 50 years of age, after which mortality slows to result in a stocking of about 100 stems per ha at around 120 years of age (Figure 11).

The relationship between the diameter at breast height over bark (dbhob) and age of dominant trees is linear, but varies with site (Rayner 1992b). For trees older than 50 years, the relationship for:

1. Karri API height class A: (greater than 50m) $\text{age} = (\text{dbhob} - 26.2)/0.61$
2. Karri API height class B: (40-50m) $\text{age} = (\text{dbhob} - 42.8)/0.4$ (Bradshaw *et al.* 1997b)

Final height varies from 30-70m at about 120 years of age, depending on site, with a few rare individuals eventually exceeding 80m (Rayner 1990). See Figure 8 for the generalised pattern of development over time for an average quality even-aged karri stand.

In the absence of fire, dead mature trees retain their main crown structure for up to 30 years, then begin to break down from the top with few 'stags' remaining beyond 60-70 years. By this time, the boles of those that have fallen and any remaining stumps contain little solid wood. In practice, most dead stags and logs are consumed by fire in a much shorter period.

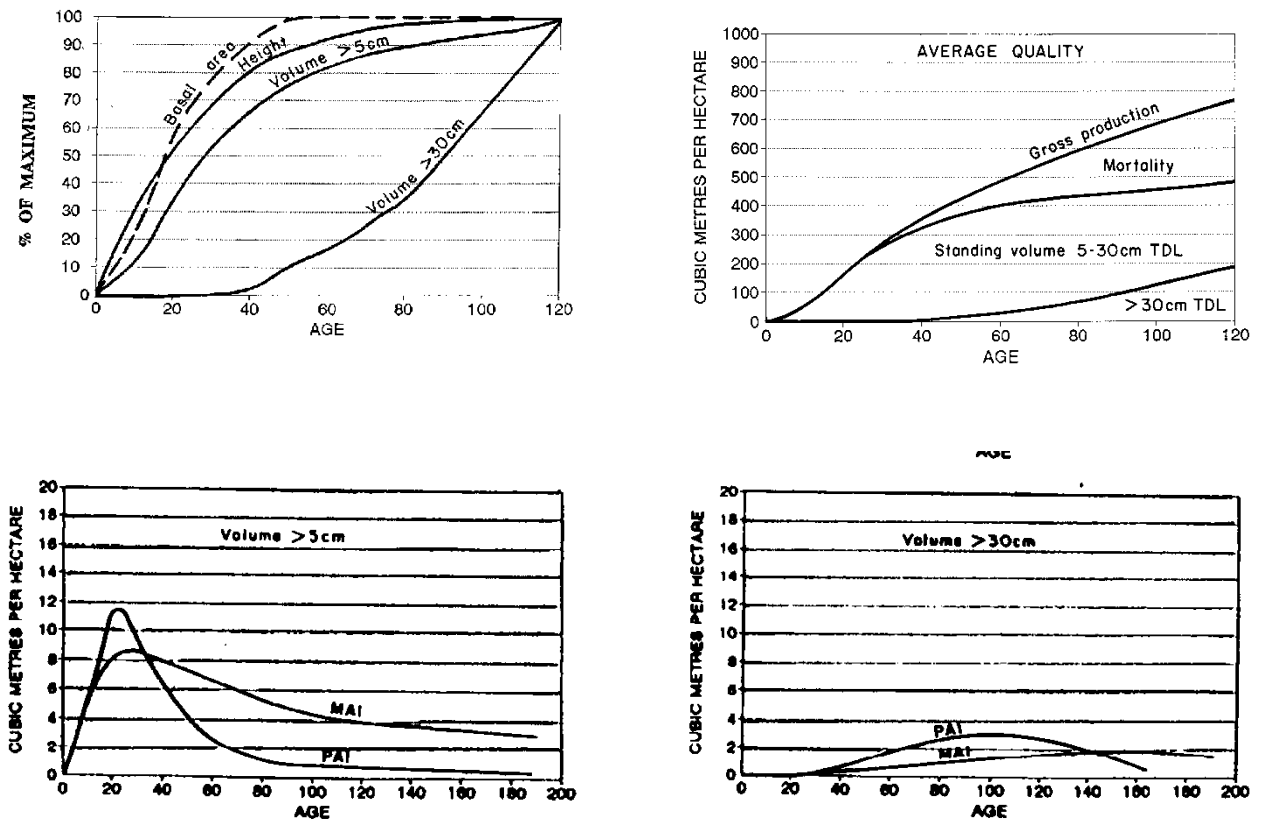


Figure 8: Patterns of stand development for average quality even-aged karri regrowth stands (Source: KARSIM v4/91 developed by Rayner (1992a)).

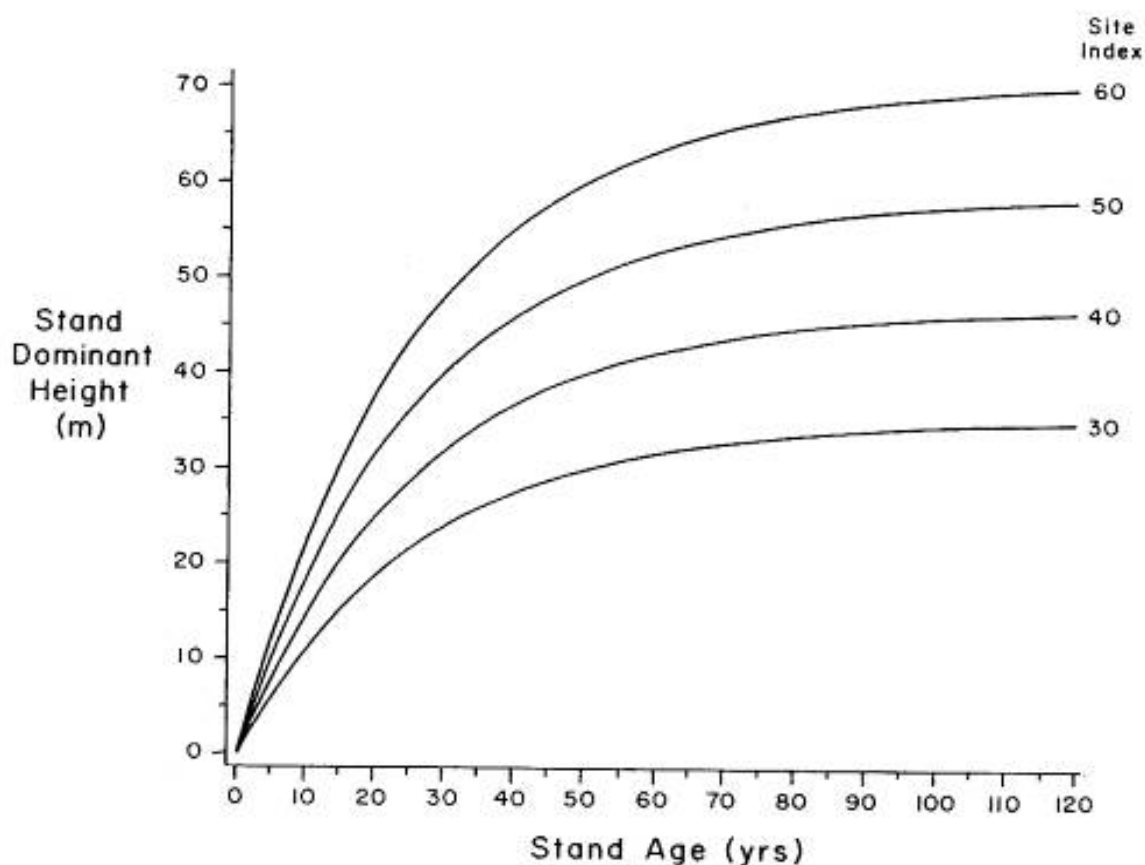


Figure 9: Modelled height development of karri regrowth over time, by site index classes (from Rayner 1990).

The principal indicator of site quality used for karri regrowth stands is site index, the 'index' being the top height at 50 years of age (Figure 9). This is a particularly useful indicator because it is largely unaffected by stand density or past management and it can be derived at any age beyond 20 years, before which dominant height is too variable to provide a reliable basis.

2.5.2 Competition

Competition for resources begins at the seedling stage, trees first competing with the understorey, and then as saplings develop and dominate the understorey, they begin to compete with each other from the time of crown closure at about 8 years. They rapidly sort themselves into dominance classes.

Lower site index (SI) stands are slower to differentiate into dominance classes, resulting in a higher proportion of the trees in smaller diameter classes (Figure 10).

The largest trees are responsible for most of the stand growth in high site quality stands. For stands of SI 50, the largest 100 stems per ha (spha) are responsible for approximately 70 per cent of the basal area growth in unthinned stands at 12 years and at 50 years of age⁵.

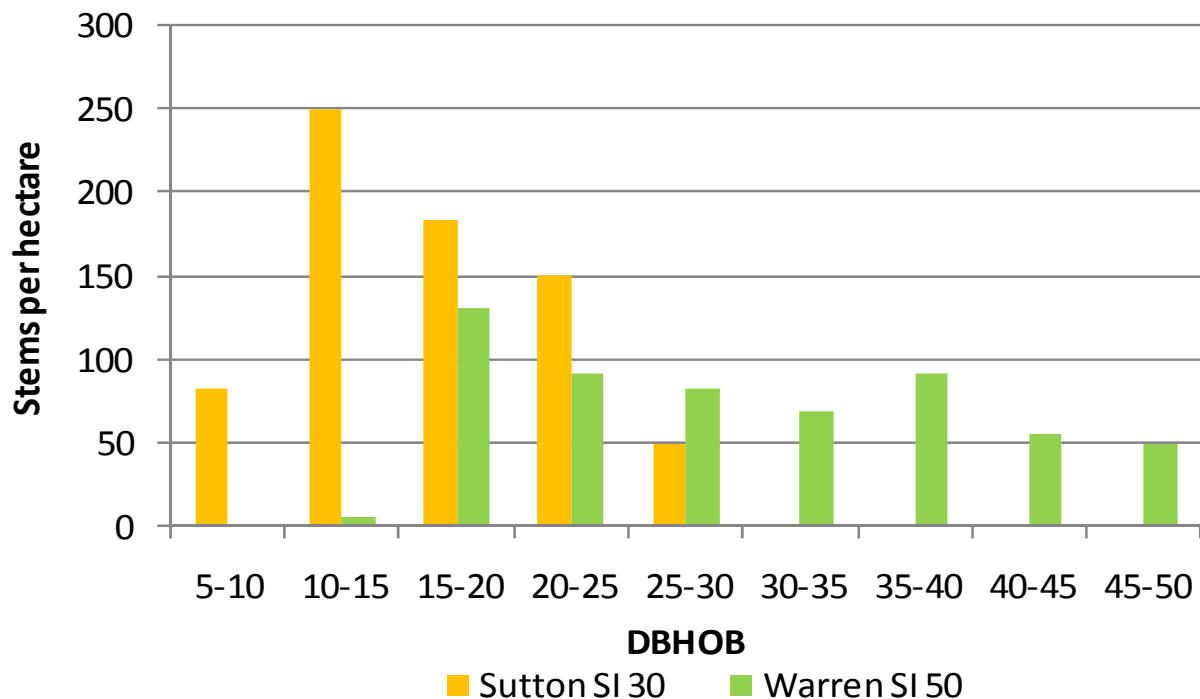


Figure 10: Lower SI stands (Sutton) tend to be dominated by smaller stems, while trees in high SI stands (Warren) sort themselves into dominance classes more effectively, resulting in a wider range of diameters. Both stands are 23 years old.

Self-thinning occurs rapidly from the time that inter-tree competition begins and continues throughout the life of the stand, resulting in a rapid reduction in tree numbers over time (Figure 11). While the rate of reduction in tree numbers slows down over time, competition for the remainder of the life cycle is no less severe.

⁵ Source: Warren and Treen thinning plot data

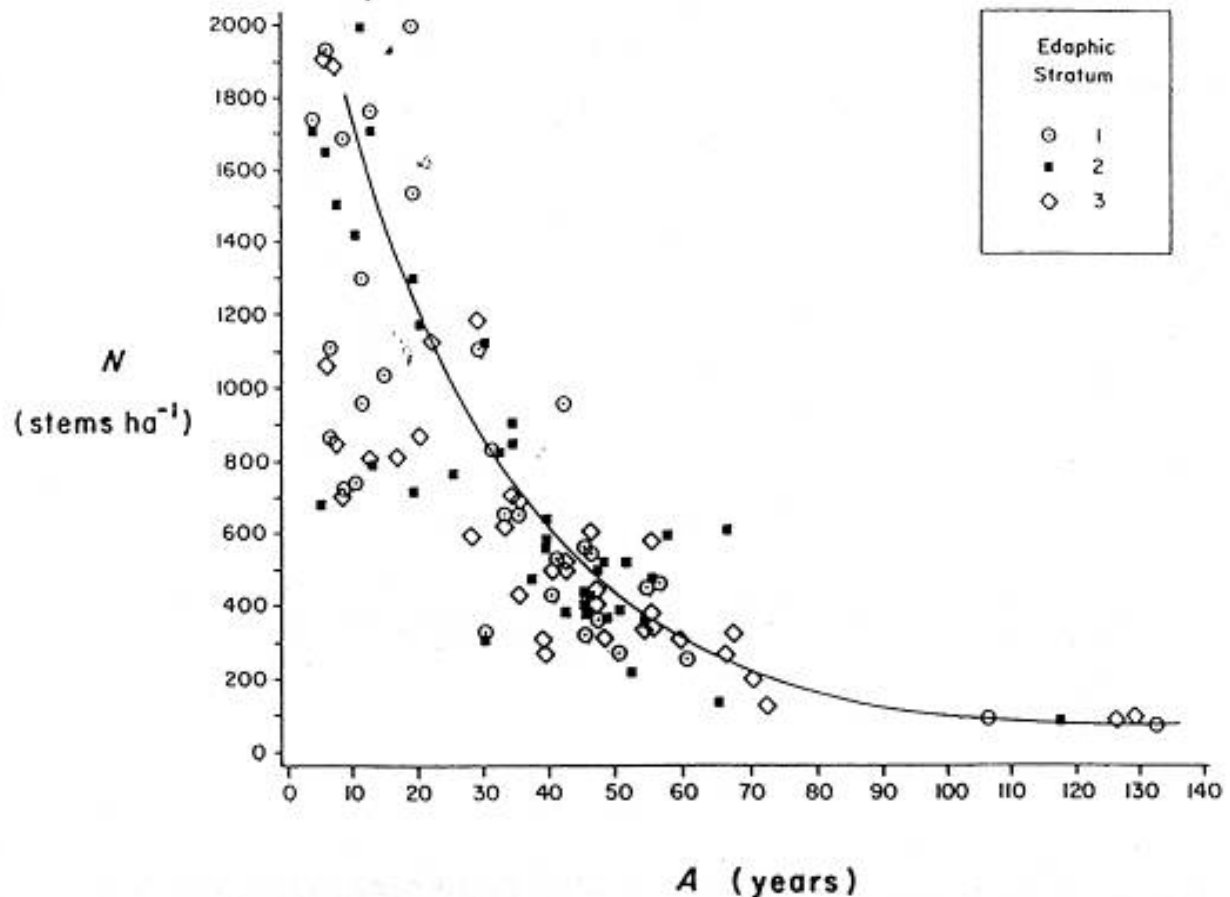


Figure 11: Modeled stem numbers over time over a range of maximum density naturally regenerated regrowth stands (after Rayner 1992a).

2.5.3 Site potential

Site potential, or the biomass that a stand can support before it is limited by nutrients and moisture, varies according to soil, position in the landscape, rainfall, evaporation and age. Commonly expressed as above ground volume site potential influences leaf area, canopy cover, height and basal area. Basal area is particularly useful because it is easy to measure, though its limitations for some purposes, such as determining site index is acknowledged.

Inions (1990) attempted to relate karri productivity to a range of soil characteristics, but failed to find a relationship between soils groups and site index. He considered this to be due to the restricted set of variables used in the study, but he also highlighted the difficulties of working with the extreme spatial variability of many of the variables. He attributed the lower site index of karri in the extreme south-east of the main karri belt, where it gives way to the tingle forest, to lower temperature and radiation.

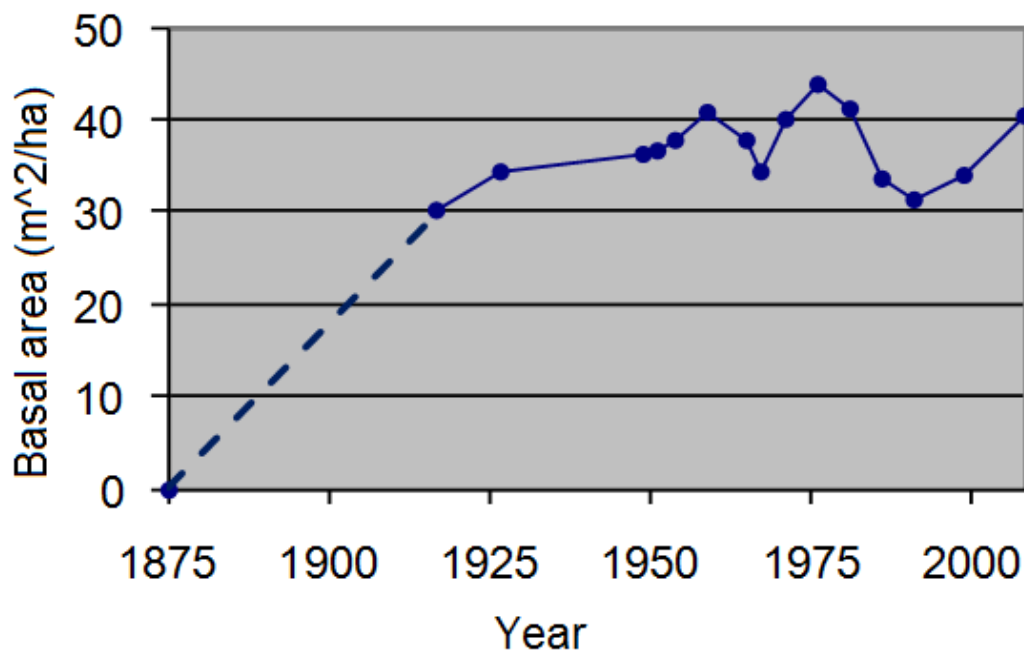


Figure 12: Accumulated basal area of Plot 823 (SI 47) ('100 year forest') over time showing the attainment of maximum basal area from about 40 years of age. Episodic mortality maintains equilibrium at about 35 m²/ha on this site. The amplitude of the variation in density is likely to increase with stand age since the loss of an individual tree has a greater impact on stand density. (Source: Forest Management Branch permanent plot data).

The capacity to utilise the full site potential is also influenced by stand age, in that young stands reach maximum density and undergo suppression and mortality at a lower density than older stands (Figure 11). From the time stands reach maximum density for the site, growth merely replaces mortality, maintaining the stands in a state of relatively constant density (i.e. stands increase in density above this level for a period and then fall below it with episodic mortality) (Rayner 1992a) (see Figure 12).

Figure 13 illustrates that the volume of trees in a stand at any one time represents only a proportion of the total production of the site over that time, with substantial proportions having grown and subsequently died. While the vast majority of trees die in the first 30 or 40 years, they contribute only a small part of the total volume of the mortality because the trees are small.

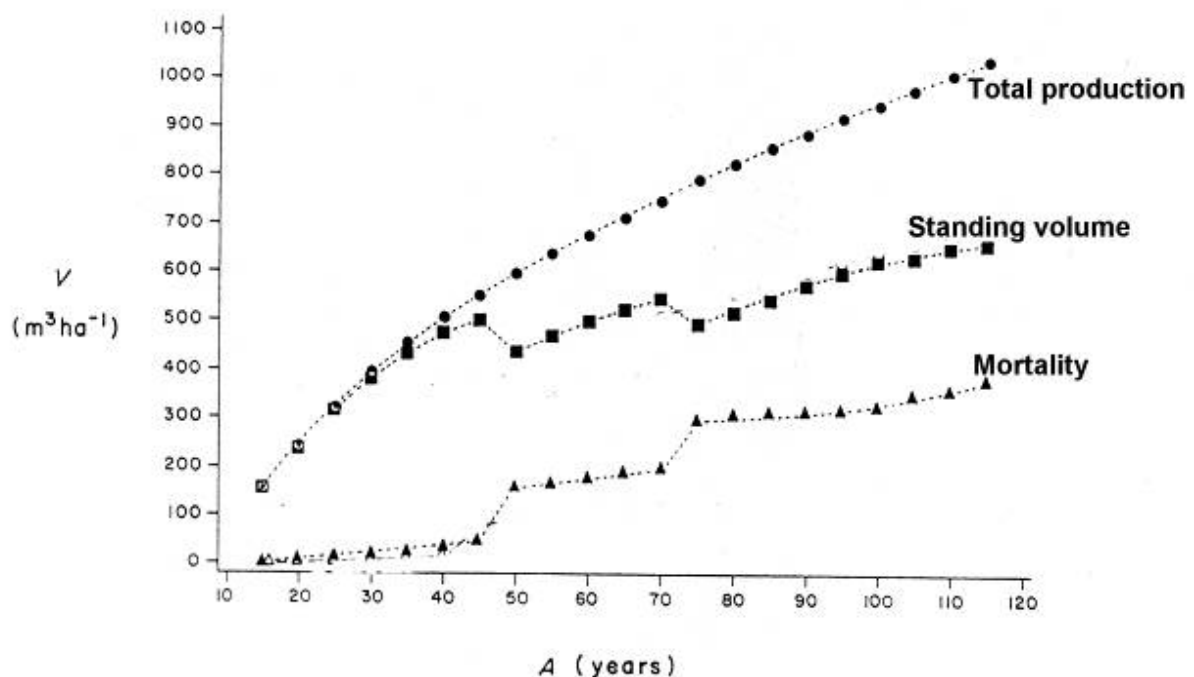


Figure 13: Karri volume accumulation over time showing the standing volume, the volume that has died and by addition, the total volume produced over the period for a stand of site index 52. The steps in the mortality and standing volume represent modeled episodic mortality. (Source: KARSIM (Rayner 1992a)).

The potential impacts of climate change on forests will depend on their exposure to climatic variation, the sensitivity of the forest to changes in climate and the adaptive potential of the forest (Bruce *et al.* 2014). Climate change has implications for site potential. If rainfall is reduced, there will be increased competition for moisture. Conversely, increases in CO₂ concentration may increase water use efficiency and may provide some amelioration (Keenan *et al.* 2013). Current modelling of the temperate plantation estate in Australia shows the impact of climate change on productivity and mortality will be limited if species are responsive to elevated CO₂ concentrations (Bruce *et al.* 2014). However, the relationship between CO₂ concentration, temperature, moisture and nutrient availability on growth rate and site potential is complex and the final state of equilibrium that will develop is uncertain. Rayner (1992a) found a broad correlation between reduced growth rate and mortality in fully stocked stands with historical periods of drought stress. The triggers and mechanism for drought related mortality depend on the species and the response strategies it employs ((Mitchell *et al.* 2012). During periods of water stress, eucalypts tend to maximise gas exchange, which leaves them vulnerable to hydraulic dysfunction.

In a trial to determine response to fertiliser application, the addition of N and N/P fertiliser increased the total stand growth rate of a high quality 12 year old karri stand over the first five years, but there was no response, and possibly a slight depression in the growth of unthinned stands, suggesting that moisture was the limiting factor at a density of about 30

m²/ha (*Figure 14*). This is consistent with work carried out in jarrah forest regrowth that indicated that applying fertiliser to unthinned stands increased moisture stress (Stoneman *et al.* 1996). There was also no increase in maximum basal area (site potential) attained as a result of fertiliser application. There was no positive response in the subsequent five year period.

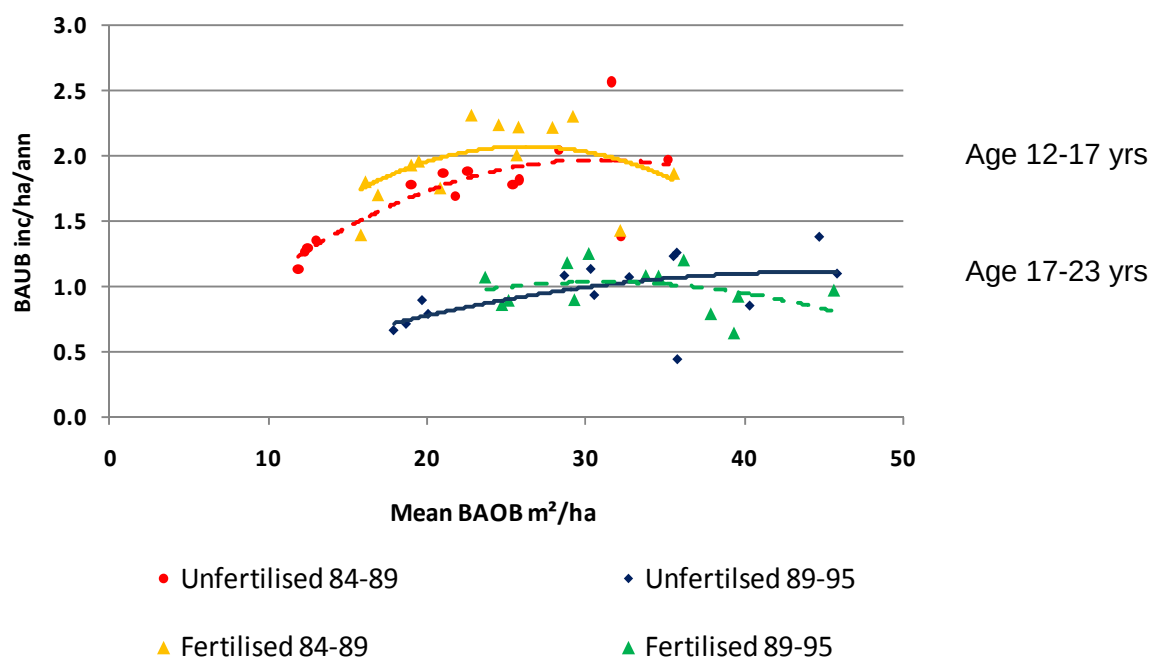


Figure 14: Response to fertiliser application to young, high quality (SI 50) regrowth stands measured over two consecutive five year periods, indicating that moisture is limiting at about 30 m²/ha.

2.6 Response to fire

Fire is critical in the regeneration process, removing the litter layer to expose mineral soil to the small seed, accelerating seed shed onto receptive seedbed, temporarily reducing understorey competition and regenerating nitrogen-fixing understorey, as discussed in earlier sections. The role of fire in stand dynamics is discussed in Section 3.2.

Tolerance of karri to fire varies with its age and stage of development. Tolerance may be considered in terms of resistance to damage and response to damage. In February 2012, a fire in Boorara and Babbington karri forest blocks affected 325ha of state forest. Around 217ha of 25-30 year old regrowth was completely defoliated, with most of the upper stem dead. Many stems subsequently produced epicormic shoots from the lower stem.

Karri regrowth in state forest, which is burnt prior to, or as part of the regeneration process, is at lower risk from fire for the first five years because of the relatively low fuel loads. However, saplings are intolerant to even low intensity fire, becoming increasingly tolerant as their bark thickens and height increases. By the time they have reached about 30m in height, the dominant trees can withstand low intensity fire. Where less than one third of the crown is

scorched, some trees less than 15cm dbhob will be killed, but the majority of trees will suffer no stem damage. Total scorch will kill most trees less than 15cm dbhob and cause significant damage to the stems of the remainder (McCaw *et al.* 1997a). The stems of 16 year old karri greater than 20cm dbhob that were fully scorched were killed back 1m from the growing tip, while those that were defoliated were killed back 6-7m (McCaw *et al.* 1994).

Damage is also related to the dryness of the fuel, the quantity consumed/fire intensity and the residence time of the fire. Most trees less than 15cm dbhob will recover from crown scorch and maintain an apical crown. Levels of damage to the stems and crowns will increase when burnt under an SDI of more than 1000. Most damage to the stems is caused by the burning of large debris (greater than 75mm diameter) close to the base of the tree.

Most saplings under 15m in height that are defoliated will be killed above ground, but will coppice and develop into trees, though some will have double stems and possibly other related wood defects. Few double stems are found in older virgin forest, suggesting that one coppice stem will dominate over time. Saplings and small poles taller than 15m that survive defoliation will suffer varying degrees of crown death and stem damage, which can result in numerous forked stems at varying heights (Figure 15 *Figure*).

It is likely that coppice plays a significant role in natural forest by regenerating stands of sapling or poles that are burnt before they are old enough to seed. For example, observation and historical records suggest that parts of the Karridale forest regrowth are derived from the burning and subsequent coppice development of regeneration that had originated from natural seed fall after timber harvesting that took place from 1880-1910.

Mature trees are relatively tolerant of fire, protected by their bark thickness (about 2.5cm) and height. The most intense fires rarely if ever kill all of the trees in a stand, though there is some observational evidence that successive severe fires may result in almost total mortality. Trees over 25m tall that are fully scorched or defoliated will incur severe damage to the cambium especially where the SDI is greater than 1000.



Figure 15: Left: A 22 year old stand damaged by bushfire, resulting in multiple coppice stems developing part way up the stem. Right: A stem that has been deliberately coppiced following the same fire showing the development of multiple coppice stems at ground level.

3 Forest structure

3.1 Stand and forest structure

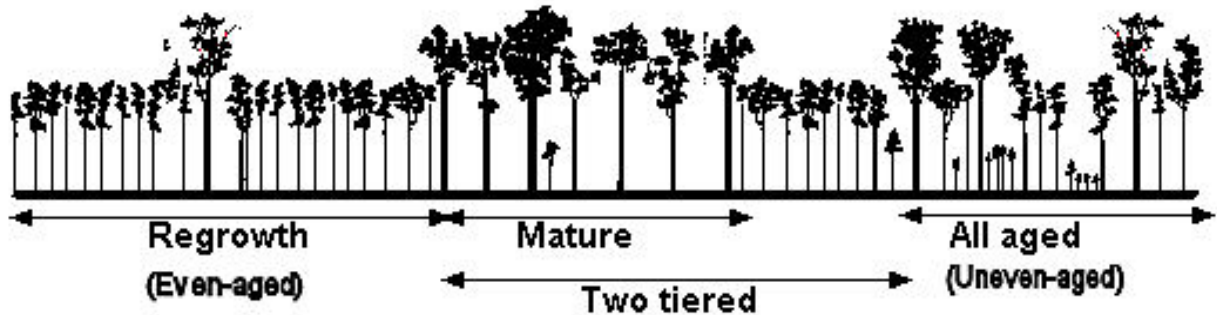
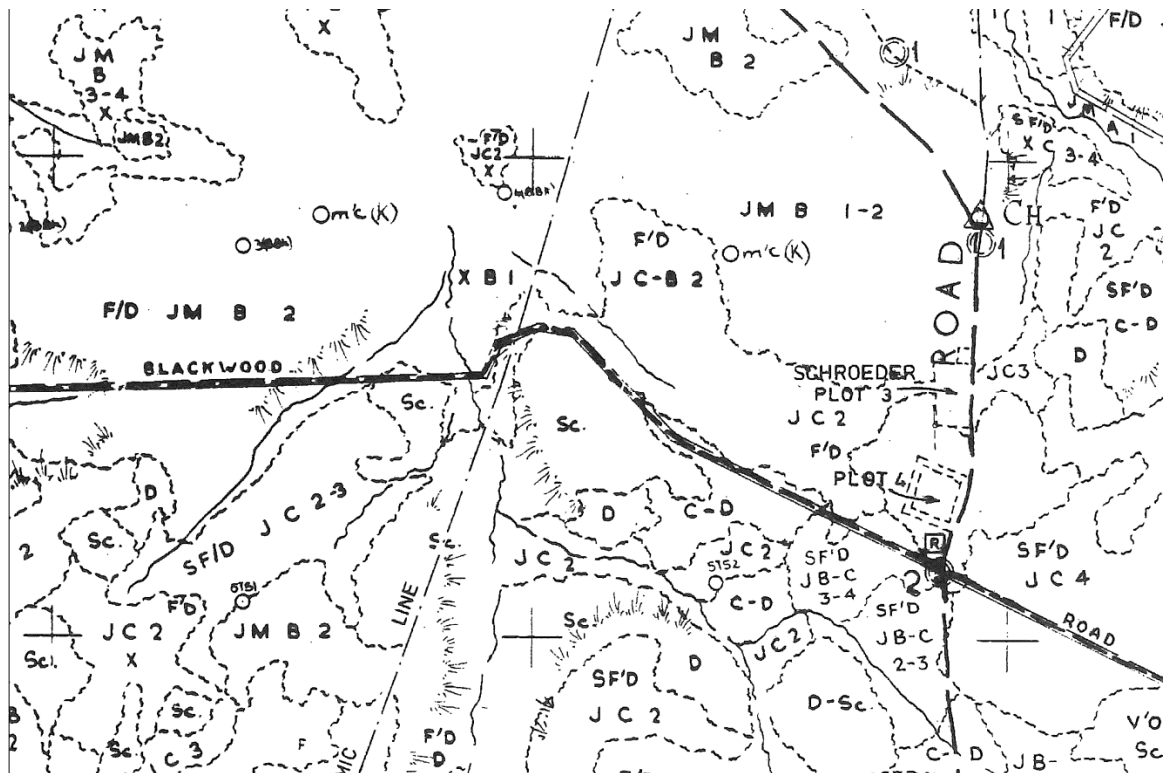


Figure 16: Diagrammatic representation of the structural variation of the karri forest with common descriptions of the various elements.

The first comprehensive mapping of stand structure was undertaken in the 1950s and 1960s using aerial photo interpretation (API). API mapping classified the forest on the basis of either species, height and crown cover (crown cover classification) or stand structure, crown cover, species and height class (structural classification) to a 2ha resolution (Figure 17) (see (Bradshaw *et al.* 1997a). The crown cover classification system was adequate for describing forest dominated by a single structure or strata. The system was not adequate for describing forest which had been harvested or had distinct strata. A structural classification system was devised for describing forest with distinct strata and was applied to all forest subject to API from 1959.

While the API information on structure is now dated, the original tree height and total density are still relevant as indicators of site quality and site potential. Species composition is also relevant, although it is recognised that reliability varies with the quality of the aerial photography taken at the time.



API mapping using crown cover classification



API mapping using structural classification

Figure 17: An example of API mapping of the karri forest using both the crown cover classification (top) and structural classification (bottom).

Subsequent mapping identified even-aged stands, pure karri, and development stages (Bradshaw *et al.* 1997a; Bradshaw *et al.* 1997b). Stand age and species type are updated for each area that is regenerated following timber harvesting using the silvicultural recording system, SILREC.

Never harvested karri forest is a multi-aged forest with cohorts of different regeneration age occurring from the broad scale (even-aged stands) to the finer mosaic of uneven-aged stands. Stand replacing fires create even-aged stands⁶, but they rarely exceed 20 ha, reflecting the relatively high tolerance to fire compared to other wet eucalypt forests such as *E. regnans*. The size class distribution of the never harvested forest at the whole-of-forest scale shows a forest dominated by large trees, with two thirds of the basal area in trees larger than 100cm dbhob (Figure 18). The structure is much more complex at the stand level. Bradshaw and Rayner (1997b, 1997c) classified the then-virgin karri forest in 1995 (63,000 ha) into development stages (Section 3.3) and estimated the age of the dominant and secondary cohorts⁷ across the forest. The virgin forest ranged in age from 30 to 370 years, with an average age of 170 years, approximately half of which is uneven-aged (Figure 19).

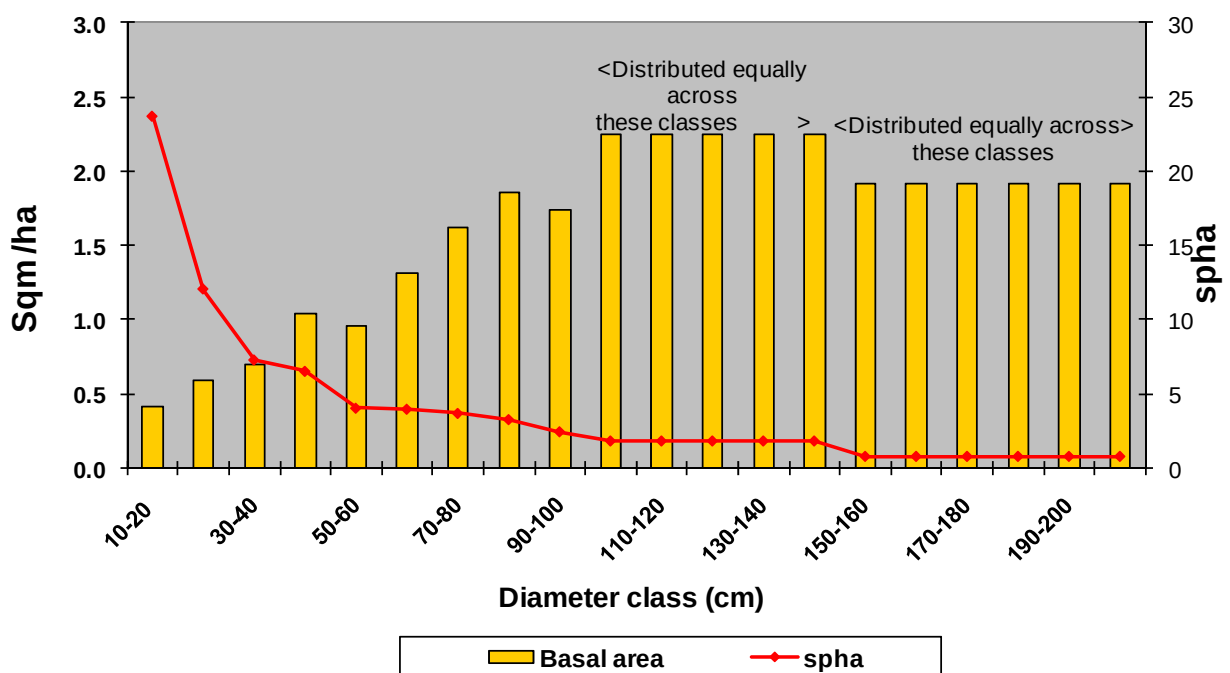


Figure 18: Stems and basal area per ha in virgin karri forest at the forest scale. Based on data from 654 (0.8 ha) inventory plots established in virgin forest that existed in the 1960s in Deanmill, Northcliffe, Pemberton, Shannon and Quinninup areas. Total mean basal area over bark (BAOB) is 33m²/ha. Total mean spha is 81.

⁶ Under the definition used in mapping, these stands may contain up to 15 per cent crown cover of an older cohort.

⁷ The dominant cohort is the oldest cohort with a crown cover of more than 25 per cent, the level at which it exerts a suppressing influence on younger cohorts. A secondary cohort is a younger cohort with a crown cover of 25 per cent or more. Other cohorts with less than 25 per cent crown cover may also occur.

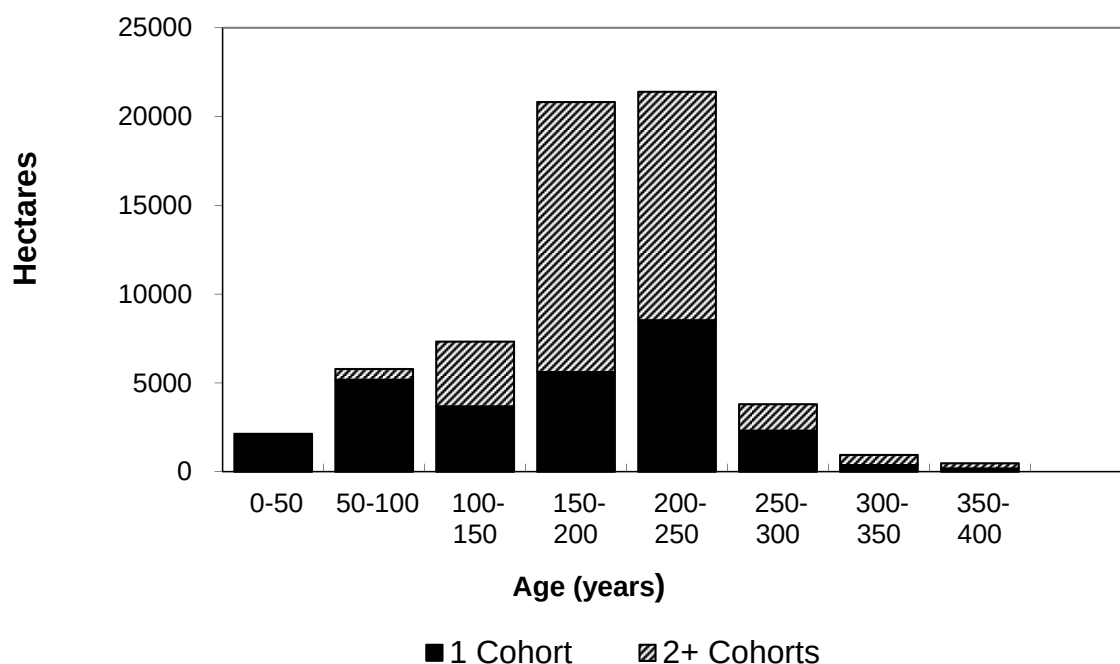


Figure 19: Age class distribution of the dominant cohort in the virgin forest in 1995 showing the proportion of stands with single and multiple age cohorts.

The age of the secondary cohorts for the 50-100 year old stands is 30 years; for the 100-150 year old, 30-90 years; for the 150-200 year old, 70-100 years; for the 200-250 year old, 50-200 years; for the 250-300 year old, 130-160 years; for the 300-350 year old, 150-200 years; and for the 350-400 year old, 70-280 years. (Source: (Bradshaw *et al.* 1997b))

The structure of the remaining original forest has been altered by disturbance (principally timber harvesting and silviculture) since the 1880s, resulting in a higher proportion of younger even-aged stands in today's forest (Figure 20). The history of timber harvesting and silviculture is described in Appendix 1.

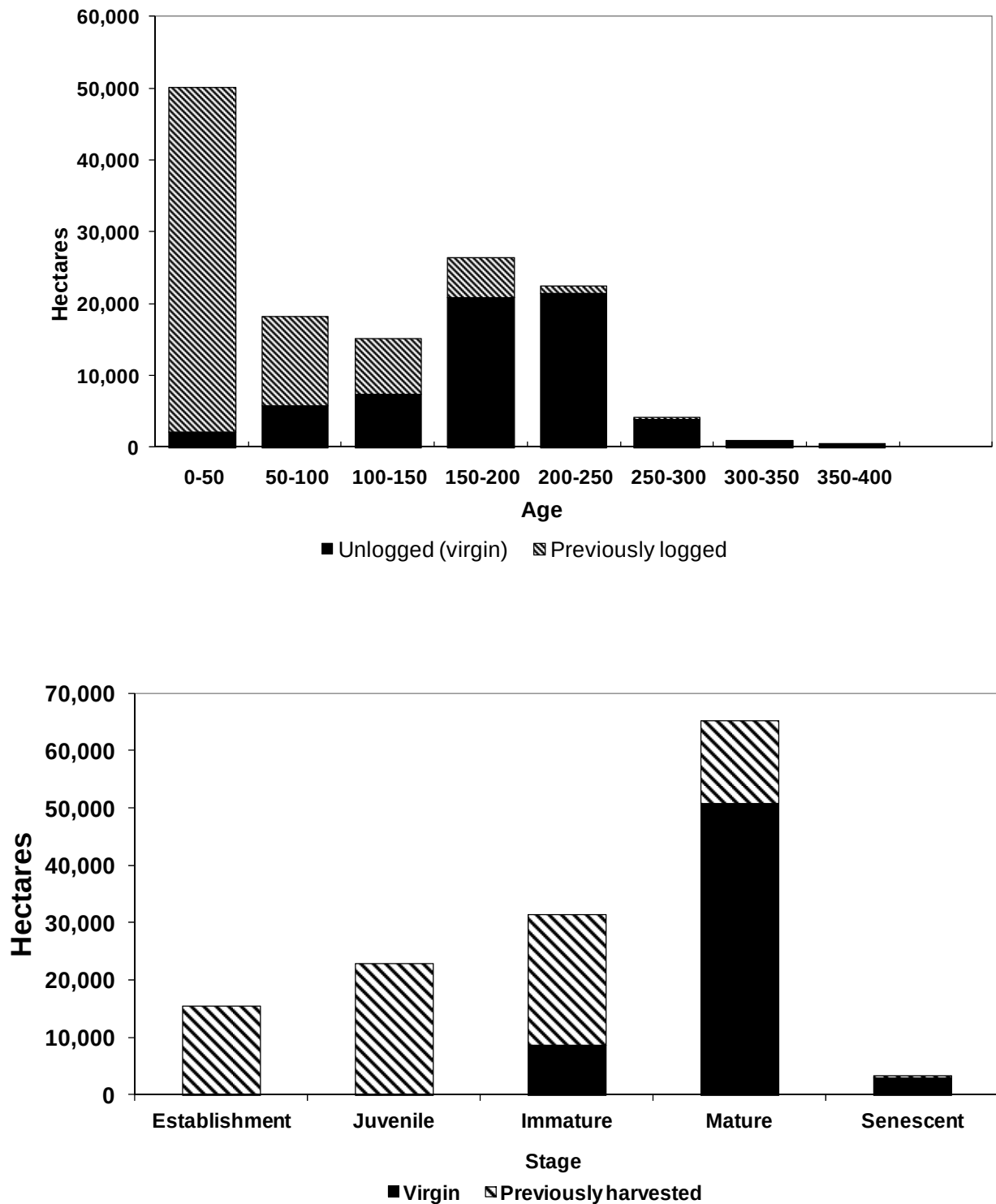


Figure 20: Stand age class distribution of the karri dominant forest in 1995 (top). Proportion of the forest in each development phase at 1995 (bottom). (Source: Bradshaw & Rayner (1997b)). See Section 3.3 for a description of development stages.

3.2 Stand initiation and structural diversity

Fire is the key driver of stand initiation and structural complexity in the virgin karri forest.

Fire plays a key role in regeneration by removing the litter layer to expose mineral soil to the small seed, causing concentrated seed shed onto the seedbed when it is most receptive, creating 'ashbed' that promotes rapid seedling growth, temporarily reducing understorey competition, regenerating nitrogen-fixing understorey and in some cases removing overstorey competition. Fire intensity plays a key role in the determination of stand structure.

Low intensity fire will burn the understorey and while it does little to encourage seed fall, it may be sufficient to initiate regeneration if there is seed in the crowns at the time of the fire. Where the forest canopy is intact, this regeneration will not survive for more than a year or two. Should a gap exist in the canopy, regeneration may persist and over time develop into saplings, poles or mature trees depending on the size of the gap. If the regeneration occurs in a small gap (for example that caused by the death of a single tree) within an immature stand, the regeneration may be rapidly suppressed by the growth and extension of influence of the immature trees surrounding the gap. This regeneration may ultimately die and all evidence of its existence may then disappear. Because of this, regeneration events in even-aged immature stands are less likely to result in the development of an uneven-aged stand. If however the gap is surrounded by mature or senescent trees that are no longer capable of extending their influence into the gap, the regeneration will persist and an uneven-aged stand will be initiated.

If no seed is available at the time of the fire, no regeneration will develop; the gap in an immature stand will be occupied by the growth of the surrounding trees, while the gap in the mature and senescent forest will remain unoccupied by trees and the stand will remain below full stocking until the next fire provides a new opportunity for regeneration. The cyclical nature of karri seeding may see gaps in mature virgin stands remain unoccupied for many years.

Moderate intensity fire that scorches trees will cause a similar response, except it will stimulate seed fall if seed exists. Moderate intensity fire is therefore more likely to result in regeneration than a less intense fire.

High intensity fire will kill some trees and create new gaps in the canopy. Younger cohorts (saplings and small poles) may be killed and re-establish from coppice development. If seed is available, the new canopy gaps will be regenerated in the same way as described above. In addition, some regeneration will develop beneath the canopy of damaged trees and will survive until the damaged trees re-establish their canopy and influence with new epicormic growth.

If only some of the trees are killed, an uneven-aged stand will be created or maintained. If all or most of the trees in an area are killed by very severe fire, then an even-aged stand of regrowth will result. Even-aged stands will develop through the stages described in Section 3.3 unless the process is truncated by another severe fire. The range of different structures that may be initiated by fire is illustrated in Figure 21.

Figure 19 shows that about half of the virgin forest in 1995 was uneven-aged, and that the age distribution follows a more or less normal distribution, unlike the negative exponential distribution of more fire sensitive species in north America, where stand replacement fires

are more common (van Wagner 1978; Johnson *et al.* 1995). While the distribution from 1995 is not necessarily typical of virgin forest at different times in history, projections into the future suggest that this distribution is likely to be relatively stable and typical of moderately fire sensitive species where uneven-aged forest is more common (Bradshaw *et al.* 1997c). A consequence of this distribution is that the proportion of the forest that would be expected to exist in the 'old-growth' condition will be higher than it is for more fire sensitive species.

Tree species composition and diversity is also affected by fire. While karri, marri and tingle distribution is largely dependent on climate and soil, it is also influenced by fire and its relationship to regeneration and the seed cycle. Marri seeds more frequently than karri and also has the capacity to regenerate from a lignotuber. Where marri is present, it will be advantaged by a fire that occurs in an off-seed year for karri. However karri will out-compete marri so that a full stocking of karri is not necessary for its ultimate domination, though it is likely to be disadvantaged by too-frequent fire. Yellow tingle seeds every year and regenerates readily. While there is usually some seed in the crowns of karri in most years, burning in a non-seed year for karri can cause a major shift in composition.

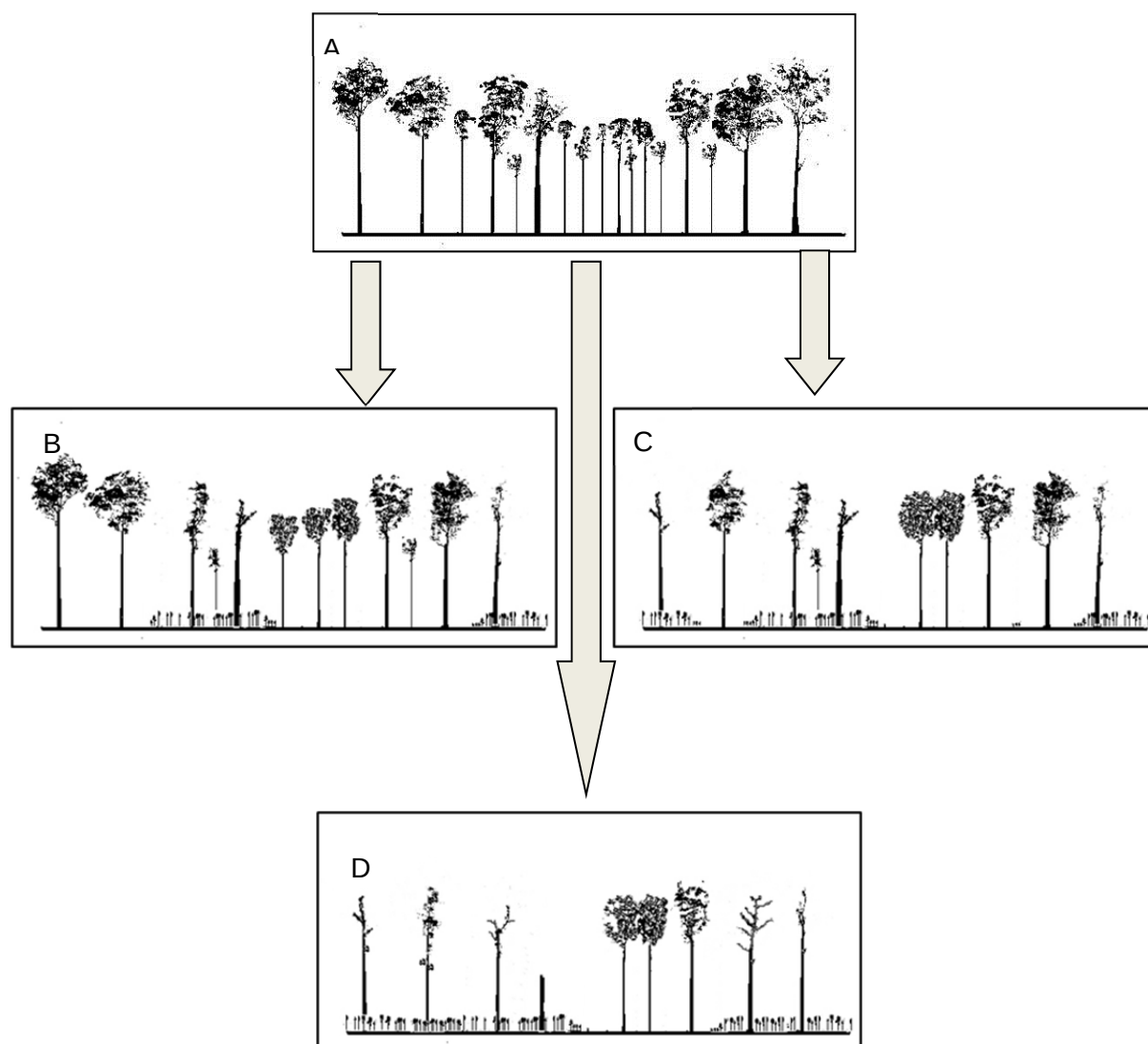


Figure 21: A diagrammatic illustration of the structural response to fire. Stand B is the outcome of age-induced mortality and regeneration that occurs in the gaps as a result of mild fire that burns the understorey and exposes mineral soil. The regeneration process may occur over many years and result in an uneven-aged stand. Stand C is the outcome of fires that are intense enough to kill some of the trees, resulting in an uneven-aged stand. Stand D is the outcome of very severe fires that kill most of the trees (i.e. stand-replacing fires that result in an even-aged stand with a few veteran 'legacies').

3.3 Structural attributes of stand development stages

Stands of even-aged karri develop through several distinct stages from establishment to senescence, which are summarised below. These descriptions are based on that from Bradshaw and Rayner (1997b), an expanded version of those previously described by Butcher (CALM 1992). Uneven-aged stands that contain cohorts at different stages of development within uneven-aged stands will have similar characteristics.

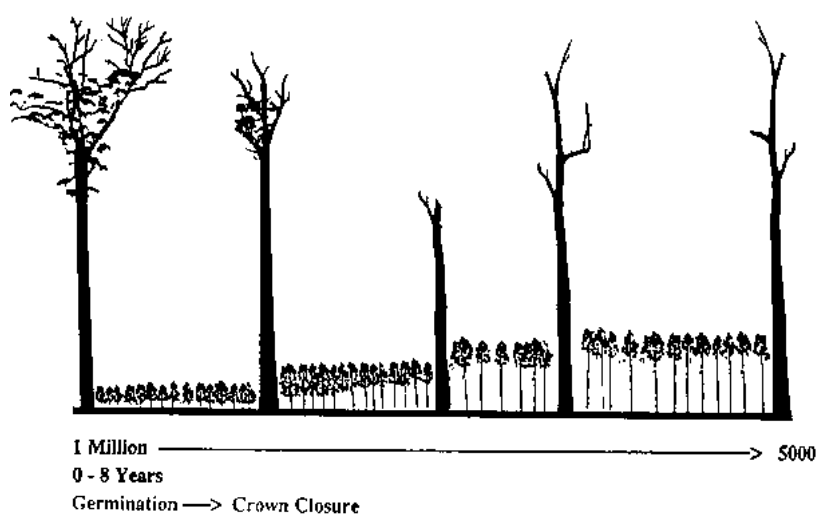
3.3.1 Establishment

In a natural forest, the establishment stage begins with the death of a tree or a group of trees in the original overstorey of such a size that the space that is vacated is not “re-occupied” by the surrounding trees (Breidahl and Hewett 1995).

Rapid growth of the seedlings ensures dominance over the understorey and the establishment stage ends with canopy closure of the saplings at about eight years of age. By this age, competition has reduced numbers from perhaps 1,000,000 germinants to 5,000 saplings per ha and the saplings are about 6 – 10 metres tall.

Net nutrient demand is highest during this period (O'Connell *et al.* 1982).

ESTABLISHMENT

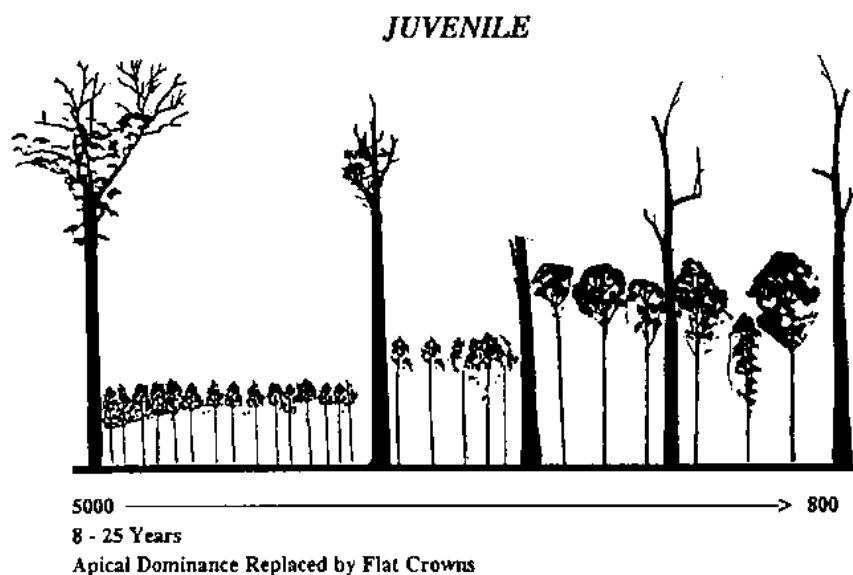


3.3.2 Juvenile

The juvenile stage begins with crown closure and is characterised by a period of intense intra-specific competition, which results in the emergence at the end of this stage of about 400-500 dominant and co-dominant trees from an original 5000 individuals at the start of the period. Current annual volume increment peaks towards the end of this period.

The juvenile stage ends when the stand is about 25-30 years old and the crown shape of the dominant and co-dominant trees has begun to alter from the previous conical form to a more rounded shape. This is due to the retention of branches at the base of the crown as they become too large to be shed cleanly from the bole. The appearance of the stand changes at the end of the period due to the shedding of the lower dead branches, causing the stand to take on the more open appearance of the ‘pole’ stand.

By the end of the juvenile stage and without further fire disturbance, the understorey also changes in character. Much of the short lived ‘fire weed’ species have died and together with the leaf litter and coarser debris, form a partially suspended, well aerated dead litter layer which is still accumulating. There is a sparse second-storey of green understorey canopy at about 4m. Changes in understorey composition over time have been illustrated by Christensen (1972). *Trymalium* is one of the more persistent species and *Allocasuarina* becomes more prominent.



3.3.3 Immature

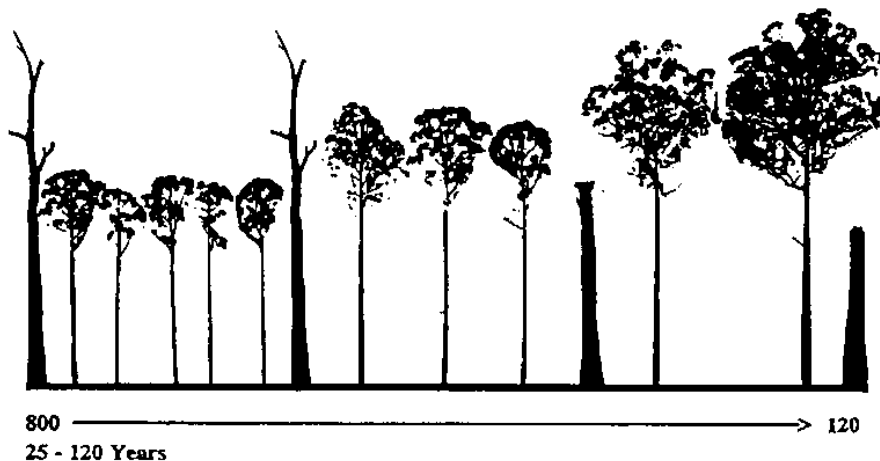
Competition continues throughout the immature stage though mortality occurs at a less rapid rate than in the juvenile stage. Dominants and co-dominants reduce from about 300 to 150 stems per hectare. Small gaps in the canopy resulting from the death of individual trees are quickly re-occupied by the vigorously growing adjacent trees. Net basal area increases until about 50 years of age, from which time it fluctuates about a plateau according to the limitations of the site and periodic mortality events (Rayner 1992b). See Figure 12.

Height growth continues at a slower rate but the dominants achieve 90 per cent of their final height by the time they are 60-70 years old (Rayner 1992c).

Although bole length continues to increase, the lower dead branches are no longer shed cleanly. The dead branches are now shed by first breaking off several centimetres from the bole and then either rotting away or being overgrown as the bole diameter increases (Jacobs 1955). There is opportunity for the development of hollows to be initiated at this stage where larger branches are involved. Toward the end of the immature stage, the crown 'shaping' branches become larger and more persistent.

This stage ends when individual tree crowns have reached the size above which they are no longer capable of expansion, regardless of the space available. This occurs at about 120 years of age.

In the absence of further fire disturbance, the understorey becomes more open. The highly suspended litter of the juvenile stage has broken down, a few long lived individuals of understorey species (such as *Chorilaena quercifolia*) remain, and dry matter (less than 25mm) accumulation has stabilised at levels in excess of 50 tonnes per ha (McCaw *et al.* 2002). The situation beyond 80 years is a matter of speculation since this is the longest (documented, studied) unburnt site known to exist in the karri forest (Figure 5).

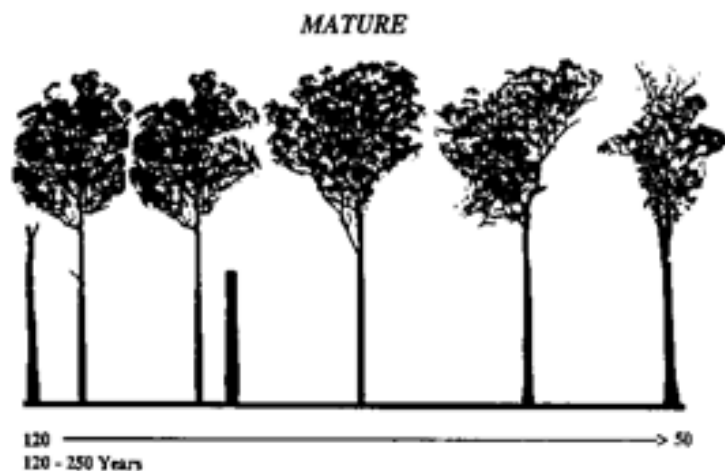
IMMATURE**3.3.4 Mature**

The rapid growth stage ends as the physical limitations of each individual are reached. They can neither occupy more of the site nor increase their crown dimensions; only tree diameter will increase. Competition induced mortality is much reduced and the stand enters a period of relative stability.

The crown has reached its maximum size (about 20-25m diameter) and permanent or shaping branches form the outline of the crown. As the extremities of the primary crown become less efficient, epicormic shoots develop within the crown (Jacobs 1955). The branches will periodically break, resulting in replacement through epicormic development without changing crown dimensions (Mackowski 1984). Individuals will slowly decline in vigour, although the growth rate of a dominant tree is largely maintained (Rayner 1992b). Where an individual tree dies, the remaining trees are unable to take up the available growing space, leaving a break in the canopy and allowing regeneration to occur. In many other non-eucalypt forest types, this equates to the time when the regeneration of tolerant species will be released and begin to develop (Oliver *et al.* 1990).

The majority of hollow development in crowns is probably initiated early in this stage as large shaping branches break and larger branch stubs overgrow. As with jarrah (Inions *et al.* 1989), their development will be accelerated by bushfire.

The end of this stage occurs when the stand is about 200-250 years old.



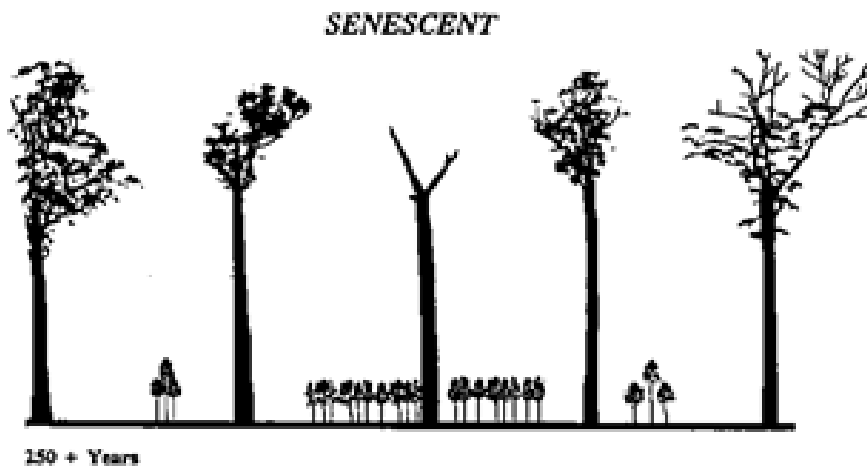
3.3.5 Senescent

This is the stage of rapid decline in health, vigour and the number of original trees. The trees have a reduced control over the site. The process of crown renewal slows and, as major branch components are lost, they are replaced by epicormic shoots lower on the branch or bole of the tree. This damage also provides entry points for fungi, which further weaken the tree's structure. In effect, the trees are in decline and will slowly break up. How much of this decline is due simply to the ravages of age and how much to the probability that trees of this age have been subjected to a greater number of damaging events is difficult to determine. However, even relatively young trees may become 'senescent' if severely injured and unable to support their existing structure following events such as storms or severe fires.

Previous studies of tree ages (Rayner 1992b) have shown that few living individuals are known to exist beyond 350 years. The age distribution of large living individuals in the forest indicates a rapid decline in numbers between 200 and 280 years, followed by a more gradual reduction until there are only a few rare individuals recorded at 350 years (Figure 20).

Opportunities for regeneration increase as the control of the site by the overstorey diminishes with the increasing death rate of individuals in the stand. This is the beginning of a new establishment stage overlapping with the senescent stage. In the absence of a suitable co-incidence of fire and seed availability, these canopy gaps may remain unoccupied by trees for a number of years. For that reason senescent stands will often support a lower density (basal area) than younger stands.

In contrast with some forests, karri has no tolerant climax species waiting to eventually replace it in the absence of disturbance. Furthermore, life-long absence of disturbance from fire is inconceivable in this climate (Underwood 1978). This model is based on the presumption that fire at least of an intensity to create seed bed conditions will occur several times during the natural life span of karri. In the final phase of development, there is no late seral phase of alternate species, but simply old karri forest. Given even modest fire disturbance at the senescent stage, the stand will be replaced by a new generation of karri forest.



3.3.6 Old-growth

Old-growth forest is defined in the 1992 National Forest Policy Statement as *Forest that is ecologically mature and has been subjected to negligible unnatural disturbance such as logging roading and clearing. The definition focuses on forest in which the upper stratum or overstorey is in the late mature to overmature growth phases (NFPS 1992).* The focus is on the stand condition rather than individual trees. The term was originally coined to describe the particular characteristics and the special functional attributes of late seral stage forests in North America. These were forests where the attributes of old age, lack of disturbance and successional change went hand-in-hand. The generalised characteristics have been summarised by Bauhus (2009). However, the interpretation of this broad concept depends on the ecological attributes of each forest type. Under the FMP, old-growth forest is defined as *ecologically mature forest where the effects of unnatural disturbance are now negligible. The definition focuses on forest in which the upper stratum or overstorey is in a late mature to senescent growth stage (Conservation Commission 2013a).*

In forest that is subject to regular naturally occurring fire, such as the karri forest, 'ecologically mature' refers to the overstorey only. The understorey may consist of the full range of seral stages. In the karri forest old understorey is more commonly found in juvenile and immature forest than it is in immature or senescent forest (Figure 26). In this respect it differs from old-growth forest types that have reached the late mature and senescent stage in the long absence of fire. There are no late successional overstorey species in the karri forest. Several attributes commonly associated with old-growth forest such as deep litter, layered canopy, presence of several age cohorts, large volumes of coarse woody debris (Bauhus *et al.* 2009) are not unique to old-growth karri forest. The understorey, litter depth and coarse woody debris (CWD) attributes are more a function of harvest and silvicultural history, burn age and frequency, than they are of overstorey age. For example, cutover forest may have a greater volume of CWD than virgin forest.

Under a definition of old-growth as being undisturbed late mature and senescent forest, approximately 50 per cent of the not-previously-harvested forest would qualify as 'old-growth'. This is a much higher percentage than that commonly found in more fire sensitive species (van Wagner 1978; Johnson *et al.* 1995). However, in the mapping of stand

development stages in the karri forest (Bradshaw *et al.* 1997b), early and late mature forest could not be readily differentiated from air photo interpretation and as a result all not-previously-harvested mature and senescent forest areas have been classified as 'old-growth' (80 per cent of the not-previously-harvested forest)(See Figure 20).

The value of old-growth forest for flora and fauna has been widely discussed in relation to various forests. However, the wide variation of attributes that exist in the karri forest suggests that studies of the function of individual attributes such as structural complexity, hollows and CWD will be more productive than attributing functionality of old-growth forest *per se*. These attributes occur at various levels within the full range of forest conditions from regrowth to old-growth.

4 Sustainable forest management

The objective of forest management outlined in the FMP is *"for biodiversity to be conserved, the health, vitality and productive capacity of ecosystems to be sustained, and the social, cultural and economic benefits valued by the community to be produced in a manner taking account of the principles of ecologically sustainable forest management"* (ESFM). It further defines ESFM as *"a set of guiding principles that seeks to sustain ecosystem integrity, while continuing to provide ongoing social and economic benefits to the community through the sustainable access to wood and non-wood forest resources and enjoyment of other forest values"* (Conservation Commission 2013b).

The FMP adopts the slightly modified Montreal Criteria of sustainability as the framework within which to identify management actions, in line with the principles of ESFM. (Conservation Commission 2013b).

The FMP's overall goals seek to:

- conserve biodiversity and self-sustaining populations of native species and communities and to allow for the recovery of biodiversity from disturbance operations
- maintain ecosystem health and vitality
- protect soil and water resources
- adapt to climate change and sustain the contribution of the areas covered by the plan to global carbon cycles, consistent with relevant legislation and the achievement of other goals
- sustain the productive capacity of native forest ecosystems and plantations as they progressively adapt to changing climatic conditions
- protect and maintain Noongar and other Australian cultural heritage
- sustain social and economic benefits, through the provision of a range of goods and services valued by the community
- ensure that management is undertaken in a systematic manner in accordance with the plan and is continually improved so as to achieve desired outcomes.

Silviculture has a direct role to play in a number of these goals. While the maintenance of socio-economic values is a goal of much of the silvicultural practice, it is dealt with in other policy and planning documents.

The FMP acknowledges that finding a balance between sometimes competing goals, and the management activities required to manage for those goals, is challenging. Silvicultural practice aims to integrate and give effect to all of the goals that are relevant to those areas available for timber harvesting.

This section deals primarily with the management of karri forest on which timber harvesting is permitted, however the biodiversity conservation strategies for areas where timber harvesting is excluded are also briefly discussed. The broad context is that:

- 53 per cent of the karri forest is within large formal reserves such as National Parks, Nature Reserves and other formal conservation areas
- 13 per cent is in informal reserves such as stream reserves and old-growth
- 33 per cent is available for timber harvesting (Proposed in Conservation Commission 2013b, 2013a).

Between 2004 and 2011, an average of 460 hectares of karri forest was harvested for regeneration each year, representing 0.8 per cent of the forest available for timber harvesting or 0.26 per cent of the karri forest. An average of approximately 1100 ha was thinned during the same period (Figure 22, Figure 39 and, Figure 40). The annual area of thinning is expected to increase to about 1500 ha under the current FMP (Conservation Commission 2013b).

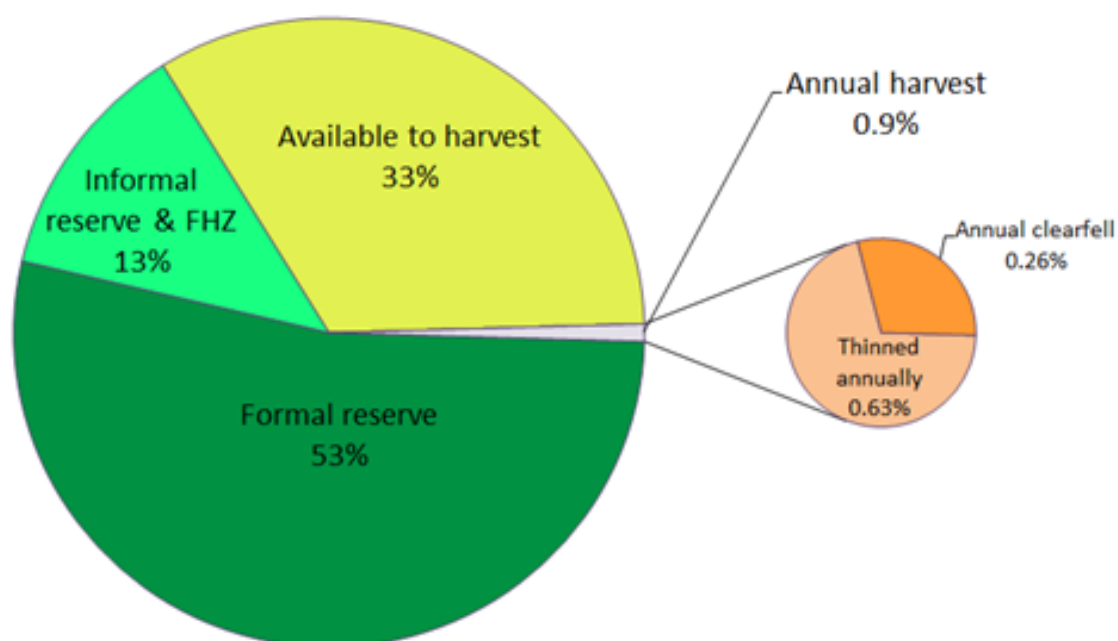


Figure 22: The proportion of publicly owned karri forest in the three forest regions by land use category. At right - the proportion of the annual harvest area in each timber harvesting category (Source: (Conservation Commission 2012) and DEC/DPaW Annual reports).

4.1 Conservation of biodiversity

Biological diversity (biodiversity) refers to the variability in structure and function of living organisms and the ecosystems of which they are a part. Conserving biodiversity requires maintenance of a diversity of habitats and ecological processes at various spatial scales, from entire forested landscapes to specific localised areas. It also includes sustaining populations and maintaining their genetic diversity. Conserving biodiversity is also aimed at assisting ecosystems to remain productive and resilient to disturbance and to changes to the environment in which they exist.

4.1.1 Conservation strategies

Lindenmayer *et al.* (Lindenmayer *et al.* 2006) propose five general principles for conservation of biodiversity in forested landscapes and a range of strategies that may assist in providing for these principles. These are:

- maintenance of connectivity
- maintenance of landscape heterogeneity
- maintenance of stand structural complexity
- maintenance of intact aquatic ecosystems
- knowledge of natural disturbance regimes to inform human disturbance regimes.

Each of these principles can be considered at several different scales within the whole forest. The objectives and strategies adopted at each of these are designed to complement each other, thereby strengthening conservation values of the whole.

Formal reserves

Large formal reserves (National Parks, Nature Reserves and Conservation Reserves) that are excluded from mining and timber harvesting disturbances are seen as one of the fundamental strategies in the conservation of biodiversity. Many of these reserves have been selected to be large enough to support self-sustaining populations of native species and communities (RFA 1998a; Conservation Commission 2013b). The expectation is that these ecosystems will continue to function under the influence of natural or low level disturbances. Some of these areas are managed according to separate management plans. All of the five forest ecosystems recognised in the karri forest are represented in these reserves (Bradshaw *et al.* 1997d; RFA 1999).

Landscape scale management

At the landscape scale, management seeks “to allow for the recovery of biodiversity between one timber rotation and the next” (Conservation Commission 2008). To assist in achieving that, informal reserves with specific management purposes are dispersed throughout areas available for timber harvesting. These reserves also contribute to biodiversity conservation at the broader landscape level. Informal reserves include old-growth forest larger than two hectares in size, buffers of varying width on all streams, ‘diverse ecotypes’, fauna habitat zones (FHZ) and travel route zones on selected roads and trails. The distribution of informal reserves throughout the landscape is illustrated in Figure 23.

These areas are designated at the planning stage and separate guidelines exist for their management (Department of Environment and Conservation 2009).

Local scale management

The application of silvicultural treatment at a coupe and patch scale is the principal focus of this material and the associated silviculture guidelines. These practices are designed to ensure a wide range of structural and compositional diversity is maintained at the local and the landscape scale. These measures complement the management of reserves and are not intended as stand-alone strategies for biodiversity management.

4.1.2 Biodiversity of the karri forest

The karri forest supports a lower diversity of plants and animals than either the jarrah or wandoo forests. There are about 160 native vertebrate species within the forest area, comprising 24 mammals, 88 birds, 24 reptiles, 16 amphibians and 8 fish.

FDIS identified eight different habitat types for vertebrate fauna across the karri forest (Christensen *et al.* 2005). However, not all of these habitat types are affected by silvicultural treatments. It has been estimated that in the wider forested area of the south-west of WA that 60 of species are in locations or vegetation types that are not affected by timber harvesting, instead occurring within the extensive formal and informal reserve system. FDIS contains a comprehensive list of vertebrate species in the south-west forests, along with their probability of occurrence and their vulnerability to the impacts of prescribed burning and timber harvesting. FDIS is used to analyse and predict the likelihood of fauna occurring in a particular habitat type, and to provide recommendations for management, paired with existing silvicultural guidance to mitigate potential impacts of timber harvesting and prescribed burning on particular species.

Vascular plants

Within the general forest area, the most species-rich types are the sedge lands and fringes of rock outcrops associated with the forest, rather than in the forest itself. The majority of declared rare and deemed priority flora are within the Blackwood Plateau and the south-east fringe of the jarrah forest to the east of the Franklin River (RFA 1998b: Maps 5 & 15; Gioia *et al.* 2000). Centres of endemism, relictual and disjunct flora are similarly located.

Where more permanent disturbances such as roads, mines, or basic raw materials pits are proposed, the planned activity is preceded by an on-site flora survey to identify declared rare and priority flora, and Threatened Ecological Communities (TEC) and Priority Ecological Communities (PEC). In areas subject to timber harvesting the disturbance is transient. Harvesting is preceded by a desk-top search to identify the location of known populations, and to assess the likelihood of the occurrence of populations based on an analysis of the vegetation types and landform. Known sites or populations of declared rare, and priority flora, and TECs and PECs are excluded from timber harvesting, and potential sites may be required to be field surveyed prior to the operation commencing. The actions required to address the flora values are recorded as part of the *Planning checklist for disturbance activities* (Department of Parks and Wildlife 2014a).

Mammals

The karri forest supports approximately 24 native mammals including arboreal marsupials, terrestrial mammals and bats which occur across a range of habitats. The five arboreal

marsupial species generally occur where karri is a minor component of the overstorey or where it occurs in isolated patches. Species that occur in karri forest are; the brush-tailed phascogale (*Phascogale tapoatafa*), common brushtail possum (*Trichosurus vulpecula*), western pygmy possum (*Cercartetus concinnus*), honey possum (*Tarsipes rostratus*) and western ringtail possum (*Pseudocheirus occidentalis*). The western ringtail possum is found on the Margaret River Plateau and the Leeuwin Naturaliste Coast where the WA peppermint (*Agonis flexuosa*) is a major part of the second-storey layer.

The karri forest supports approximately 10 terrestrial native mammals, however few of these species are widespread across the karri forest. Four species require hollow logs for refuges or breeding. The quokka (*Setonix brachyurus*) and quenda (*Isodon obesulus*) prefer habitats with dense cover.

Nine species of bat occur in the karri forest, including the white-striped freetail bat (*Tadarida australis*), lesser long-eared bat (*Nyctophilus geoffreyi*), Southern freetail bat (*Mormopterus planiceps*), chocolate wattled bat (*Chalinolobus morio*), Gould's wattled bat (*Chalinolobus gouldii*), Gould's long-eared bat (*Nyctophilus gouldi*), greater long-eared bat (*Nyctophilus timoriensis*), western false pipistrelle (*Falsistrellus mackenziei*) and the Southern forest bat (*Vespadelus regulus*). They occupy all the niches in the forest from above the canopy to the forest floor and while some are solely dependent on tree hollows as roosting sites, others are not and use tree hollows opportunistically.

Birds

Birds provide the greatest species richness of all the vertebrate fauna and the karri forest supports approximately 88 different bird species that occupy the entire stratum of the karri forest. The bird species found in the karri forest vary greatly from each other in terms of breeding cycles, social systems, nest placement, group types (flocks or solitary birds), foraging habits and migratory inclinations.

Of the 88 species, sustainable timber harvesting and karri silviculture may affect 22 species (Christensen *et al.* 2005). These are listed in Appendix 2. Of the 22 species, 19 rely on tree hollows for nesting sites either solely or occasionally. Two species listed as endangered or vulnerable rely on tree hollows for nesting. They are the forest red-tailed black-cockatoo (*Calyptorhynchus banksii naso*) and Baudin's black-cockatoo (*Calyptorhynchus baudinii*). The priority 3 masked owl (*Tyto novaehollandiae novaehollandiae*), priority 2 barking owl (*Ninox connivens*) and the specifically protected peregrine falcon (*Falco peregrinus*), also rely on tree hollows for nesting.

There are two other bird species found in the karri forest which may be adversely impacted by timber harvesting but do not rely on hollows - the crested shrike-tit (*Falcunculus frontatus*) which is listed as Priority 4, and the malleefowl (*Leipoa ocellata*) which is listed as vulnerable.

Reptiles

The karri forest supports approximately 24 species of reptiles, comprising one species of dragons and monitors, one species of gecko, one species of legless lizards, 15 species of skinks and six species of snakes. Of these only one species, the jewelled south-west ctenotus (*Ctenotus gemmula*) is listed as priority 3, but is not considered to be adversely

affected by timber harvesting or prescribed burning. Two species, bobtail (*Tiliqua rugosa*) and heath monitor (*Varanus rosenbergi*) have been identified in FDIS as species to consider in the management of prescribed fire.

Amphibians

The karri forest supports 16 species of frogs. Of these species, there are two species that are listed as endangered or vulnerable, being the orange-bellied frog (*Geocrinia vitellina*) VU and the white-bellied frog (*Geocrinia alba*) CR. Based on FDIS it is expected that none of these species will be adversely impacted by timber harvesting activities (Christensen *et al.* 2005).

Fish

The karri forest supports 12 species of fish. Of these species, there are two species that are listed as endangered or vulnerable, being Balston's pygmy perch (*Nannatherina balstoni*) VU and the mud minnow (*Galaxiella munda*) VU. There are two priority species, the black-striped minnow (*Galaxiella nigrostriatas*) P3 and the pouched lamprey (*Geotria australis*) P1. Finally, there are four introduced species, brown trout (*Salmo trutta*), mosquito fish (*Gambusia affinis*), rainbow trout (*Salmo gairdneri*) and redfin perch (*Perca fluviatilis*). Based on FDIS it is not expected that any of these species will be adversely impacted by timber harvesting (Christensen *et al.* 2005).

Appendix 2 provides a list of vertebrate fauna that could be adversely affected by timber harvesting.

4.1.3 Structural complexity

Forest scale

An important element of sustainability of habitat is the maintenance of an appropriate spatial and temporal representation of all stages of forest development. An appropriate age structure at the forest and landscape scale provides for the relative stability of structural habitat in the presence of inevitable and continuous change at the local level. Local level change is initiated by mortality and renewal that occurs in both natural and actively managed forest. The assumption is that if the various stages of development are sustained, then the other elements of biodiversity that are related to those stages are also more likely to be sustained.

In the past, the maintenance of sustained yield of timber has been seen as a surrogate for the sustainability of other values since sustaining timber values also requires maintaining a range of age structures across the forest. While timber yield was aimed at producing large sized logs (coming from old, large trees) and harvesting was less intense (no removal of non-sawlog material), this assumption was largely valid. However as management becomes more intense, the protection of other values that may have been provided for by default, must be more explicitly provided for. Structural goals provide a more direct means of addressing the issue of structural complexity and a more direct means of evaluating the sustainability of all forest values. Whole-of-forest and landscape level structural goals have been reviewed by Bradshaw and Burrows *et al* (2002). Ferguson *et al* (2003) have suggested that managing structural goals would shift the emphasis to sustainability of forest

and landscape structure and values, and away from a reliance on non-declining yield of sawlogs as the primary regulator of sustainability.

A study of the present and predicted age distribution of the karri forest, prior to the expansion of the conservation reserve system and the protection of all old-growth forest, showed that the age structure of karri is likely to remain relatively stable over time at the whole of forest scale (Bradshaw *et al.* 1997c).

Landscape scale

Diversity at the landscape level has important implications for biodiversity. Some fauna species are temporarily adversely affected by the immediate impact of timber harvesting and prescribed burning within the disturbed area. Displaced species will colonise the regrowth as it develops over time. For example, the bush rat (*Rattus fuscipes*), the mardo (*Antechinus flavipes*), the quokka (*Setonix brachyurus*), the Southern brown bandicoot (*Isodon obesulus*), the common brushtail possum and the Western ringtail possum return to karri regrowth within two to 10 years after regeneration (Christensen 1992).

The abundance and diversity of bird species are affected by the size of the clearfelled coupe (proportion of 'edge') and the age or development stage of regrowth in the coupe. In the first four years post-harvest, about half of the species are affected either positively or negatively (Wardell-Johnson *et al.* 2000). These differences are reduced as the regrowth develops into the juvenile stage (Atkinson 2003), although different groups are affected differently and there are seasonal differences. Most studies have been done in areas that have been recently clearfelled, where differences are most extreme and while there are indications that most mature species are present in 50 year old regrowth (Tingay *et al.* 1984), there is a need for more comprehensive studies to determine the changes that occur as the regrowth develops through different structural stages.

Reptiles such as the tiger snake (*Notechis scutatus occidentalis*), the mourning skink (*Egernia luctuosa*), Smith's skink (*Egernia napoleonis*) and the burrowing skink (*Hemiergis peronii*) return to karri regrowth within a few years of regeneration. While some invertebrates such as relict spiders with poor powers of dispersal could be adversely affected, these are mostly located in areas that are not subject to timber harvesting (Christensen 1992). Several reviews of the impact of disturbance have been undertaken (Christensen 1986; Wardell-Johnson *et al.* 1992; Christensen 1997).

An important element of biodiversity conservation is the maintenance of undisturbed forest widely spread through the forest (Figure 23), and the dispersal of timber harvesting in space and time, so that there are sufficient areas from which re-colonisation may occur (Christensen 1986; Wardell-Johnson *et al.* 1991; Wardlaw *et al.* 2012).

Diversity at the landscape scale is principally achieved through the distribution of formal and informal reserves across the landscape, the dispersal of timber harvesting in time and space and variation in the size of timber harvesting coupes. Diversity of understorey age at the landscape scale is achieved through the passage of time and rotational prescribed burning within LMUs (Mattiske *et al.* 2002; Conservation Commission 2008: KPI 16), and at the local scale through patchiness of prescribed burns. Bushfires will also impose change at the landscape scale to both the overstorey and the understorey, with the extent of change linked to fire intensity.

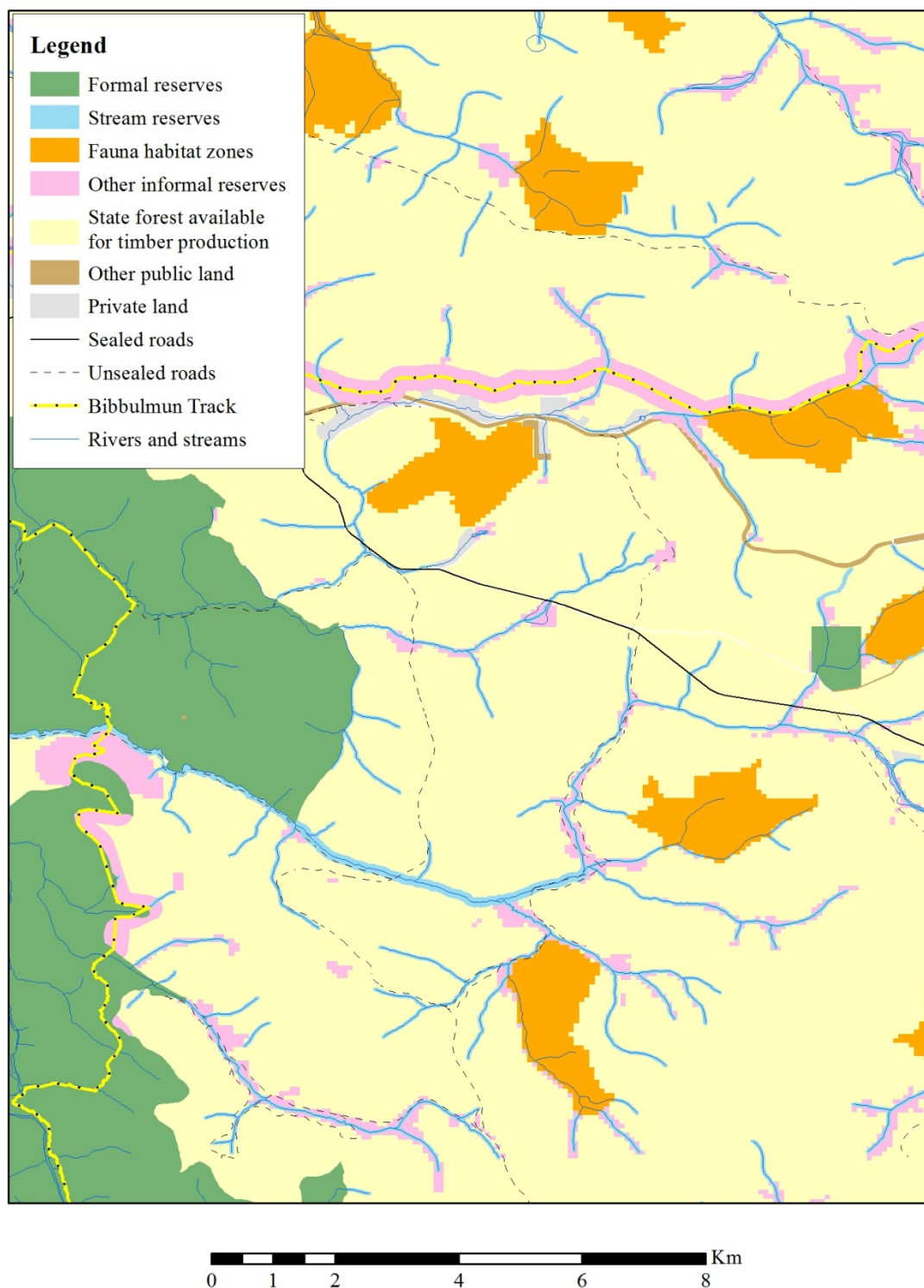
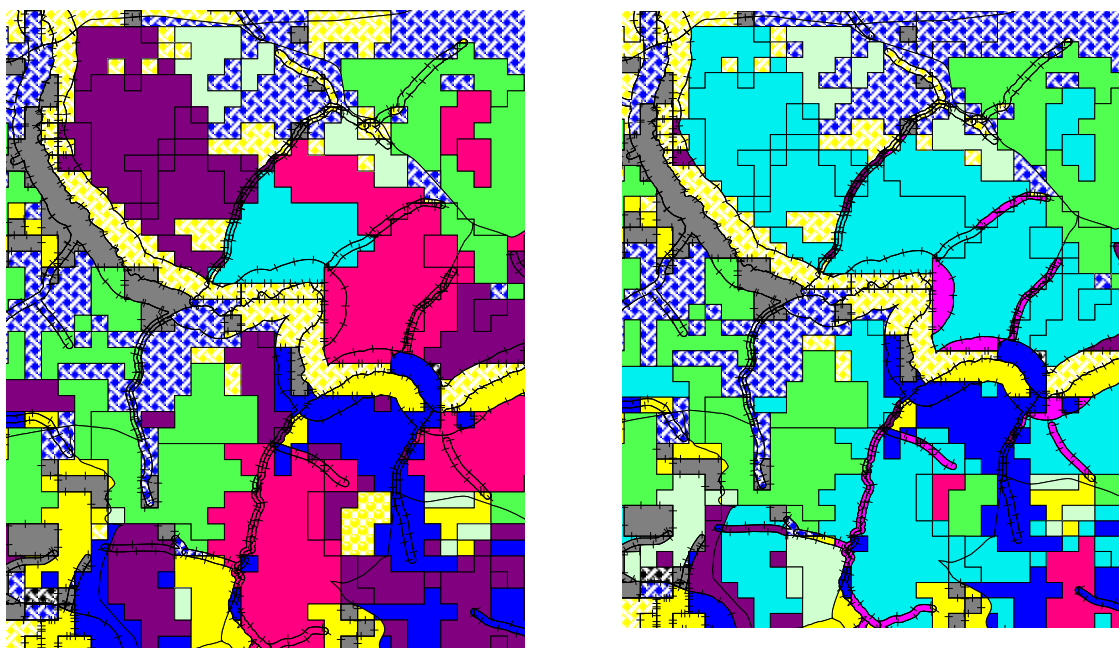


Figure 23: A portion of the south-west forest showing formal reserves and the distribution of informal reserves throughout the forest available for timber harvesting. Informal reserves include stream, diverse ecotype, travel route zones, and old-growth forest remaining outside existing and proposed conservation reserves.

Structural complexity as a basis for biological diversity changes with time, as does the spatial relationship between different structural elements. For example, the juxtaposition of a stand in the establishment phase and one in the immature stage may represent a significant structural difference. Over time however, the differences diminish as both stands proceed towards the same development stage and changes again where timber harvesting takes place. These changes can be simulated to evaluate the impact of different management options not only on structure but also on biodiversity. It is possible to develop a range of metrics appropriate to structural complexity that could be used to evaluate alternative management strategies and monitor outcomes. An example of the changing structural relationships over time is illustrated in Figure 24. For this illustration, the details the development stages are not included, as the emphasis is on the dynamics of change.



a: time zero

b: a after 40 years

Figure 24: An illustration of the changing spatial relationship between stands at different stages of development over a 40-year period. Changes from establishment, juvenile, immature, mature and senescent, represented by different colours in a mixed jarrah/ karri landscape over a 40-year period. Within-patch structural variation adds to this complexity. Overlain on this changing pattern is the constantly changing age of the understorey associated with time since the last burn operating to a different cycle. These relationships have important implications for biodiversity and fire protection strategies.

Local scale

Overstorey trees with hollows

While there are no hollow nesting species confined to the karri forest, the occurrence of hollow bearing trees across the landscape is important for a number of species (Appendix 2). Habitat trees (trees with or with the potential for hollows) have not generally been retained in karri areas subject to clearfelling. This is mainly because of their impact on regeneration (Rotherham 1983), but also their vulnerability to windthrow in the long term and associated safety concerns. A number of practices have been used to counter the absence of habitat trees within a coupe. This included the maintenance of buffers to all streams and certain roads and the retention of mature forest areas at a maximum distance of 400m (Wardell-Johnson *et al.* 1991), with the assumption that mature trees were provided in these areas. The latter no longer applies since the introduction of FHZ.

While historically mature habitat trees have not been retained in most clearfelled coupes, younger potential habitat trees have been retained. These are younger, smaller trees that are expected to live for a further 150-200 years, providing habitat potential after existing mature trees would have died but before the regrowth is old enough to develop hollows. Their exposure in the first few years of regeneration will promote epicormic development on the bole, especially if they are scorched during the regeneration burn. As the regeneration develops, shading of the epicormic branches will cause them to die, providing the opportunity for early development of hollows along the upper bole (Figure 25). Their impact on regeneration suppression is significantly less than that of mature trees. Significant but variable numbers of retained marri in mixed karri/marri stands also serve as habitat trees in these areas. The retention of some dead and senescent trees in some clearfelled areas was introduced in the current FMP to enhance structural complexity (Department of Parks and Wildlife 2014b; Burrows *et al.* in review).



Figure 25: Epicormic shoots that develop on retained potential habitat (left) will develop but be suppressed by the developing regrowth. The dead branches that are created along the bole of the tree (right) have the potential to develop hollows at an earlier age and on younger trees than would normally be the case.

Coarse woody debris

The value of CWD for biodiversity and other purposes has been recognised in a number of studies around the world, including recent work in the wet eucalypt forests of Tasmania (Grove *et al.* 2003; Hollis *et al.* 2009; Wardlaw *et al.* 2009; Gates *et al.* 2011). It has also been shown to be important for the re-colonisation of the fern *Asplenium aethiopicum* after clearfelling of karri (McCaw 2006). No specific, detailed study of the amounts, condition and species associated with CWD has been undertaken in the karri forest. Timber harvesting has the capacity to add or remove CWD to/from the site. Areas that were clearfelled and regeneration burnt prior to chipwood utilisation contained more than 350 tonnes per ha of CWD greater than 75mm and 250 tonnes per ha greater than 600mm (Jones 1978). Clearfelling with chipwood utilisation reduced CWD to 200 tonnes per ha greater than 75mm and 80 tonnes per ha greater than 600mm, which is still substantially more CWD than occurs in unharvested areas (Jones 1978; Smith *et al.* 1993). The current regeneration practice of pushing harvesting debris into heaps and burning them results in lower rates of CWD than broadcast burning (see Section 4.2.1 for an explanation of the reasons for this practice). Excluding some logs from heaps will increase CWD. In areas where marri has been retained, felling some trees after regeneration can also create additional CWD, depending

on the tree's relative value as a standing habitat tree or a ground log. It is thought such a practice would be unlikely to have a significant impact on regeneration.

Large logs in advanced stages of decay cannot be quickly replaced and need to be deliberately retained where they exist and protected from the regeneration burn where it is practical to do so. Accordingly, the exclusion of some large logs from regeneration burning has been introduced under the current FMP.

While it might have been expected that rotation lengths of 100 years may lead to a reduction in the potential for maintaining CWD, Grove and Meggs (2003) found no relationship between decomposition rates and log size in *E. obliqua*. Utilisation, site preparation method, bushfire and periodic prescribed burning are likely to have a greater impact on the survival of CWD in the karri forest. Prescribed burning conducted in spring with low SDI is less likely to burn CWD. Where required, the recruitment of CWD can be maintained by occasional felling of trees during the rotation.

Second-storey trees and understorey layer

The intensity of site disturbance and regeneration burning of harvested areas can be expected to kill nearly all of the obligate seeders and remove the above ground portion of root stock species. Therefore, susceptible species that require protection must be identified and excluded from harvest. Areas proposed for timber harvesting are initially examined in a desk-top survey to determine the likelihood of rare flora presence. All areas proposed for permanent (roads) and semi-permanent (landings, basic raw material pits) clearing are subject to an in-field survey prior to construction.

The maintenance of understorey diversity is dependent on their regenerative capacity. The impact on the understorey of the disturbance is temporary, although species abundance and cover vary significantly with time since fire following a normal successional process (Christensen *et al.* 1975). In a study of site classification and karri regeneration, Inions *et al* (Inions *et al.* 1990) found that the community type as defined by indicator species remained unchanged in 143 out of 144 sites that were clearfelled and burnt. In a study of areas that had been subjected to clearfelling and slash burning over the last 60 years, Wardell-Johnson *et al* (2004) found that karri forest understorey was highly resilient to this level of disturbance. Vascular species richness was highest for the first few years following the disturbance and declined over time in the absence of further disturbance. The principal exception is blackberry which may occur on some moister sites. The fern *Asplenium aethiopicum* has been shown to recolonise clearfelled and burnt areas as far as 400m from a mature forest edge (McCaw 2006). Introduced plant species that occur immediately after disturbance are generally ephemeral and are soon suppressed by the developing native understorey.

In areas of karri forest subject to timber harvesting, the most common second-storey species is karri oak (*Allocasuarina decussata*). Wherever practicable, groves of karri oak are retained without disturbance.

Overstorey species composition

Two thirds of the karri forest occurs in mixture with marri and to a lesser extent with jarrah. Following the harvesting of marri/karri stands, only karri needs to be deliberately regenerated. Marri regeneration occurs naturally from lignotubers, supplemented by seedling regeneration from retained marri in some situations. Marri continues to be represented in the dominant strata of the regeneration following clearfelling and regeneration with karri (Hewett 1993). Where jarrah or blackbutt (*E. patens*) occurs in mixture with karri and marri in the overstorey, it may be necessary to plant seedlings of these species to ensure that they continue to be represented in the regeneration and thus future stand. Regeneration survival monitoring records species mixtures.

To ensure that secondary species (marri, jarrah, blackbutt) continue to be maintained in the stands (where they previously existed), thinning prescriptions provide for their continued representation even where they may not qualify under the normal criterion for retention as 'crop' trees.

4.1.4 Genetic conservation

Since the cessation of the seed tree regeneration method (see Appendix 1), all seed used in regeneration is collected for the production of nursery raised seedlings. Given the low level of genetic differentiation within the main karri belt, with the exception of the lower Warren area (Coates *et al.* 1989), Millar *et al.* (2007) have recommended that seed collection zones for regeneration within this area be based on climatic and eco-geographical factors, rather than a narrow definition of 'local' or on the river valleys that have been used in the past. It may be prudent to give consideration to the selection of seed that is expected to be more suited to the predicted climate for a site than for its present climate (Aitken *et al.* 2008) and the current FMP allows for this.

Where knowledge of the population genetic structure of a species exists or can be reasonably inferred, this should guide seed collection areas. Factors considered to be important for any seed collection strategy include: matching topographic and edaphic features; allowing for expected changes in climatic conditions between seed collection sites and regeneration sites; and the need to use good quality seed with sufficient genetic variability to help enhance the resilience of regeneration (Millar *et al.* 2007).

Seed collected for regeneration is usually collected from the same LMU as the area to be regenerated. Flexibility is required to facilitate desired outcomes – for example where disease is present, or rainfall has declined, it may be appropriate to consider the use of disease, and/or drought, resistant varieties of those same species. In this case, the best source of seed or seedlings may be from another area. Alternatively, if disease or drought resistant varieties are unavailable or unknown, then using mixed seed sources to maximise genetic diversity might be an appropriate alternative strategy. This would provide a broader source of variation, allowing for greater potential to adapt to new perturbations such as disease or environmental change.

4.1.5 Predator control

The fox (*Vulpes vulpes*) has had a major impact on the populations of small native mammals and together with land clearing and the feral cat, remains the most important threat to native wildlife in southern Australia. It was implicated in the rapid demise of quokka (*Setonix brachyurus*) in the karri forest in the 1930s (Stewart 1936). 'Western Shield' is a broad scale baiting program used throughout the south-west of WA to control foxes (De Tores *et al.* 1998) and also feral cats in some areas. Additional baiting before, during and after timber harvesting operations is also carried out. The natural tolerance of south-west mammals to sodium fluoroacetate makes it possible to use 1080 poison to control introduced predators without risk to native species. The program has resulted in significant increases and/or maintenance of the population of many vulnerable native species.

4.1.6 Fire and biodiversity

Understorey species composition and structure change with time since fire (Christensen 1972; Burrows *et al.* 2003; Wardell-Johnson *et al.* 2004). One of the goals of the FMP is to *"protect and maintain...ecosystem health and vitality, through maintaining biodiversity, including structural complexity and heterogeneity"*. Prescribed burning is used to create a diversity of understorey age at the landscape scale as represented by LMUs (Mattiske *et al.* 2002), by the use of appropriate prescribed fire frequency and season; and at the local level by ensuring patchiness and spatial distribution of prescribed burns (CALM 2000; Adams *et al.* 2003).

In planning for its prescribed burning program, the department takes into account a hypothetical negative exponential distribution of time-since-fire for the forest as a whole and for each of the major LMUs (Burrows 2008; Conservation Commission 2008). The most common pattern and stable pattern of vegetation ages in natural fire-prone environments is one where there are small patches of long unburnt vegetation within a matrix of younger vegetation (Weir *et al.* 2000). The degree to which the fuel age distribution conforms to a theoretical negative exponential curve has been used as an indicator of how well biodiversity objectives are being met by the fire management program (Conservation Commission 2008: KPI 16). Time since last burn (a surrogate for understorey age) illustrated in Figure 36 shows a marked skew towards older ages, especially in areas available for harvesting. The reason for this difference is that regrowth areas are protected from fire for up to 25 years until the regrowth is old enough to withstand mild fire. Figure 26 highlights the higher proportion of understorey aged greater than 20 years old in juvenile and immature growth stages. The range of understorey age within each overstorey development stage is a further measure of structural and floristic complexity which changes over time. To provide for the maintenance of late succession species, a range of sites in reserved areas should be protected from fire for extended periods. Relatively few such sites have been identified at present. The risk of accidental burning of long-unburnt sites is reduced if a larger number of small areas are selected rather than a few large areas.

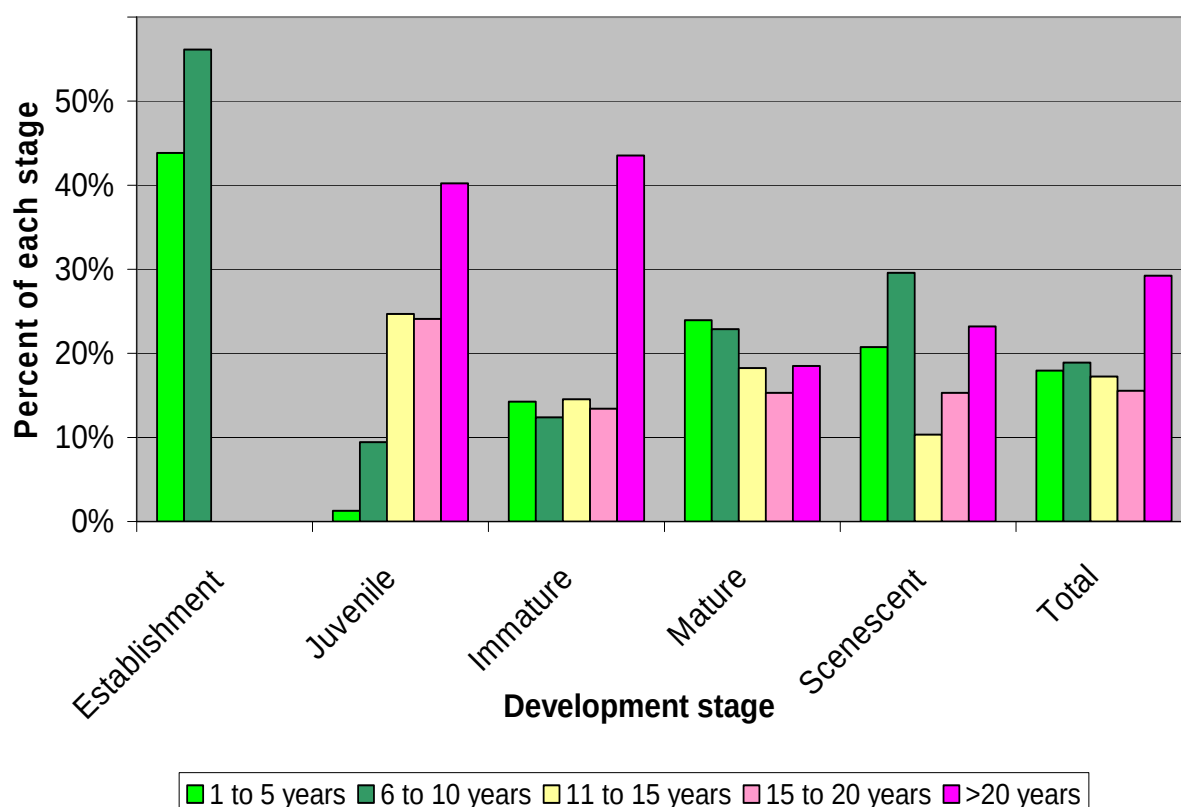


Figure 26: Nominal understorey age within each overstorey development stage in the karri forest in 2008. The nominal overstorey ages are: Establishment 0-8 years, Juvenile 9-30, Immature 31-120, Mature 121-250, Senescent >250 years. (Source: FMIS database).

4.1.7 Climate change

Further reductions in rainfall will increase moisture stress and may adversely affect vegetation health and species composition in some environments, with effects on overstorey more likely in areas of lower rainfall and shallower soils.

Analysis using a CSIRO high-severity climate scenario identified the five LMUs associated with the karri forest and two to the tingle/karri forest most at risk. These are LMUs that would be outside the rainfall threshold for karri or the tingle forest (Gentilli 1989; CSIRO 2007; Maher *et al.* 2010). Under the CSIRO high-severity scenario, six per cent of the karri forest and 75 per cent of the tingle forest would be outside the required rainfall threshold by 2030. While a general shift in biodiversity reflective of a new rainfall regime could be expected, the most immediate effect is likely to be on stream biota and streamside vegetation. Sites with shallow soils are also likely to be affected. There is already evidence that some perennial streams have become seasonal in recent years and the period of flow has decreased in many (Hughes *et al.* 2012; Kinal *et al.* 2012). Reduced abundance of some species associated with these changes have been observed in jarrah forest areas and similar changes might be expected in karri forest streams (Storey *et al.* 2010; Penniford *et al.* 2011).

Croton *et al.* (Croton *et al.* 2015) have shown that (under the high-severity model (CSIRO Mk 3.5)) predicted rainfall-deficit (rainfall minus potential evaporation) will significantly reduce

the maximum leaf area index (LAI) that can be supported in the karri forest by 2070, especially in the drier parts of the forest. The most effective means of alleviating the impact is to reduce water use by reducing overstorey vegetation density. This practice could only be applied to one third of the FMP area (i.e. the area available for timber harvesting) and there is a practical limit to the rate that this could be done. If overstorey vegetation density were to be reduced by a proposed treatment to levels which were outside the scope of the current silvicultural guidelines, the FMP requires the approval of a catchment management plan and may require development of specific silvicultural guidance.

4.2 Maintenance of productive capacity

Under the FMP, the principal timber production objective for the karri forest is the sustained yield of sawlogs, defined as the yield of sawlogs that the forest can produce for an extended period (to at least the year 2070) at a given intensity of management (Conservation Commission 2013b; Ferguson *et al.* 2013). It is based on the yield that is achievable from the resource base available for harvest at the time of preparation of the plan. Sustaining the yield in an absolute sense is seen as secondary to the achievement of other conservation objectives as expressed through increased reservation in recent years. The yield of non-sawlog material is based on that produced in the course of producing sawlogs. The modelled sustained yield is modified by applying allowance for risk factors, to give a lower allowable cut.

The principal source of sawlog yield for the previous FMP period (2004-2013) was mature trees in stands that were selectively cut in the past. During the current FMP, there will be a transition from selectively cut mature stands to regrowth originating from the 1930s to the 1940s. Regrowth stands will become the major source of sawlogs for toward the end of this FMP period, transitioning into areas regenerated from the 1970s in the next FMP period (2024-2033). The sustained yield estimates assume that regrowth stands will be thinned, thereby reducing the time for crop trees to reach sawlog size compared to the time that it would take if they were not thinned. The allowable cut of karri sawlogs proposed for the FMP (59,000m³/yr) is projected to progressively increase in subsequent decades, with the potential to more than double by 2045 due to the assumed ongoing thinning of the areas of regrowth regenerated since the 1970s (Conservation Commission 2013b), which are currently available for timber harvesting.

4.2.1 Factors influencing silvicultural practice

Influence of stand density on growth

The growth response of karri at different densities following a first thinning is illustrated in Figure 27. This data shows that: at low density stands are effectively 'free-growing' and growth (expressed as basal area) increases with density until the trees begin to compete with each other at 'critical density'; the point of 'optimum' and 'maximum density' increases with stand age; older stands maintain the same growth over a wide range of stand density; total stand growth reduces at maximum density; and stand growth reduces with stand age. These follow the general relationships common to most forest species (Assmann 1970; Smith 1986). Rayner (1992a) has expressed this relationship in the form a stand density management diagram.

The current thinning procedure for karri specifies a higher density for retention following a second thinning, given that typically growth response will be slower than that following a first thinning, thus requiring a less intense thinning to maintain stand growth rate. This is based on the response reported by Assmann (1970) for other species, but is not supported by the modelling of Rayner (1992a). No second thinning trials have yet been done to resolve this issue. However, eventually as the trees reach maturity they will reach the stage when they can no longer continue extending their influence to occupy more of the site. At this point, further thinning would leave the stand understocked and may no longer be appropriate, and the development of regeneration would then be more appropriate.

Thinning response and stand growth are also affected by site quality. Figure 28 illustrates the lower basal area increment and the earlier onset of competition (lower critical density) observed in lower site index stands at the same top height. The data indicates that stand basal area (and volume) growth can be maintained over a wide range of stand density, but the range varies according to age and site. Figure 28 indicates that top height alone is not suitable as an indicator of optimal thinning intensity across all site indices. In the absence of growth data, the density of dominant and co-dominant trees is the most reliable indicator of 'critical' density.

While Moller's general hypothesis (Moller 1954) that 'critical' density is approximately 50 per cent of maximum density seems to hold true for high quality stands, it appears to be less so for lower site quality karri stands. The presence of some marri in these stands may have contributed to the higher maximum basal area (Figure 28).

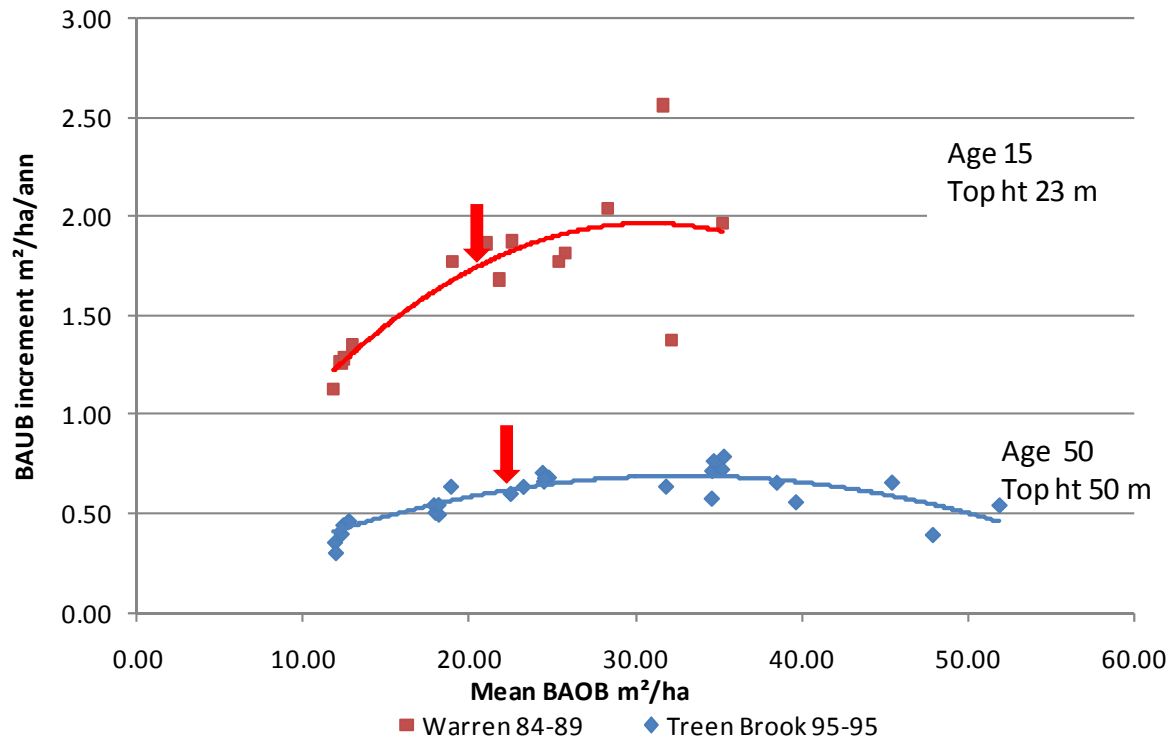


Figure 27: Basal area growth at different age and density in high quality (SI 50) stands, based on data derived from the first thinning of even-aged stands at 15 year old (Warren plots) and 50 year old (Treen Brook plots). Pemberton annual rainfall during the period was 1130mm and 1127mm respectively. The 'critical' density at each age is highlighted with an arrow. BAUB is the mean for the period of measurement. Mean BAOB is the mean standing basal area during the period between measurements.

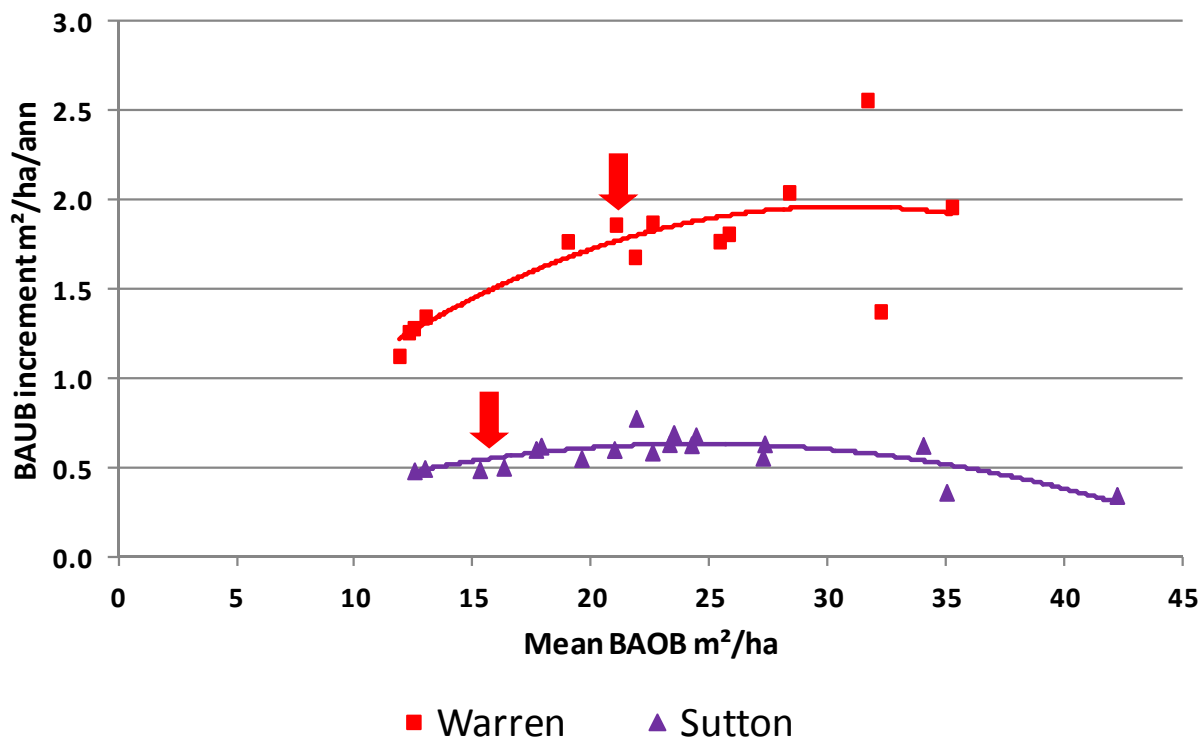
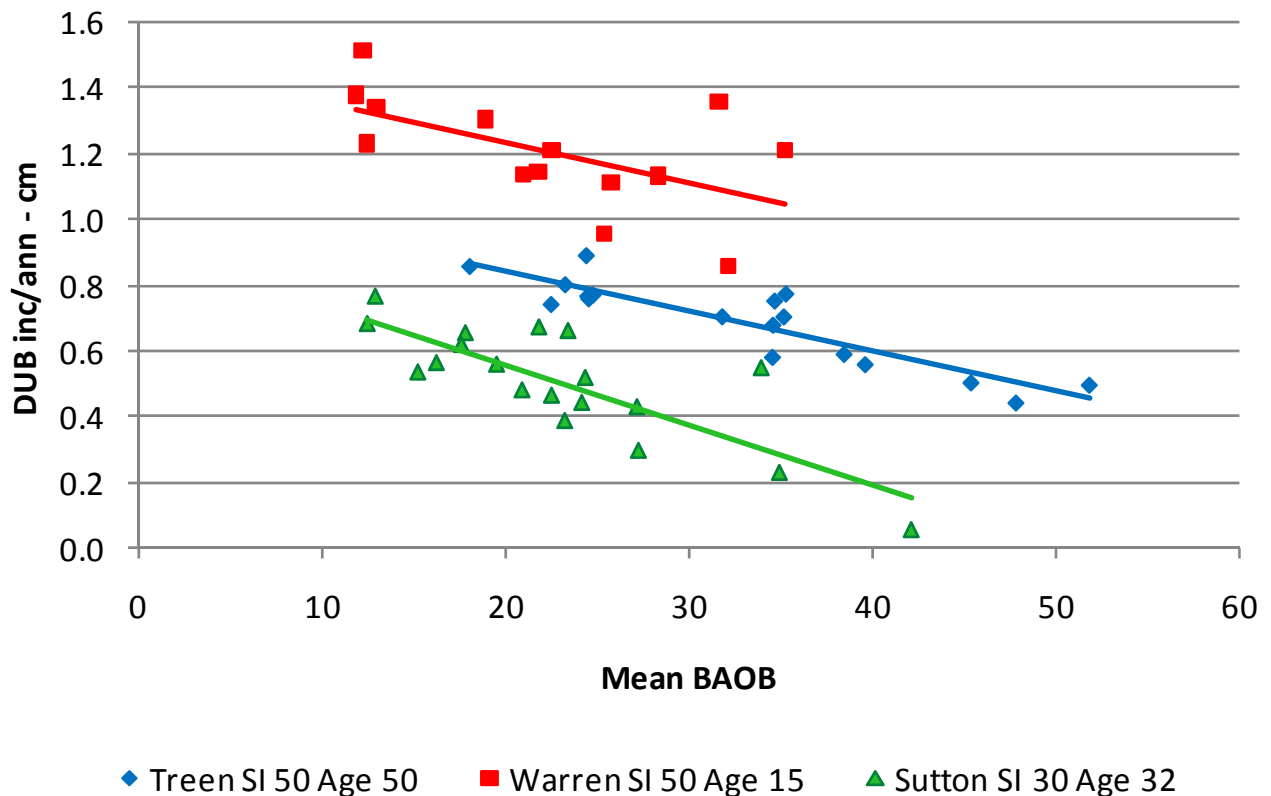


Figure 28: Basal area increment of two stands at the same top height (24m) but of different site index showing the onset of competition (critical density) at a lower stand density in lower quality stands. Warren is SI 50, average age 15 years, critical density 21 m²/ha. Sutton is SI 30, average age 32 years, critical density 16 m²/ha.

Influence of tree size on product yield

Tree diameter is important in the production of sawlogs. While a tree of 40cm dbhob can typically produce a minimum sized Grade 1 sawlog, log length (length to minimum top diameter), tolerance to defect and sawn recovery increase with diameter (Figure 29



). A 60cm dbhob tree will produce a log of Grade 1 sawlog size over the full length of the 'clean zone' (Figure 33). The maximum number of 60cm trees (crop trees) that the stand can support before suppression occurs (approximately 35m²/ha) is 125spha. Maximising the diameter and volume growth on the best 125spha (or less as the stand ages) is therefore the primary aim. Based on similar logic, 280spha at 40cm dbhob could be supported, where optimum production is the objective. Trees of this size are an important source of sawlogs from intermediate thinning and in this sense can be considered as 'secondary' crop trees. Stems in excess of 280spha will never produce significant volumes of sawlogs and are surplus to the requirements of sawlog production, but are nevertheless valuable in the initial stages of stand development to maintain small branches on the crop trees and later as a source of chipwood.

Thinning is the most important tool available to increase the rate of diameter growth. The maintenance of stand growth on fewer stems results in an increase in the diameter growth of the trees retained, thereby reducing the time for them to reach any nominated size. Figure 30 illustrates the response to thinning of the largest 100 trees in the stand. The trade-off between maximising diameter growth of crop trees and maximising stand growth (Figure 28) needs to be evaluated against the requirements of yield flow. While the crop trees in all stands respond to thinning, the relative response is greatest in the lower quality stands. This data shows a 350 per cent response in Sutton forest block, relative to the 35 per cent response for the younger, higher quality Warren stand for a thinning to 15 m² /ha', (both stands are the same top height).

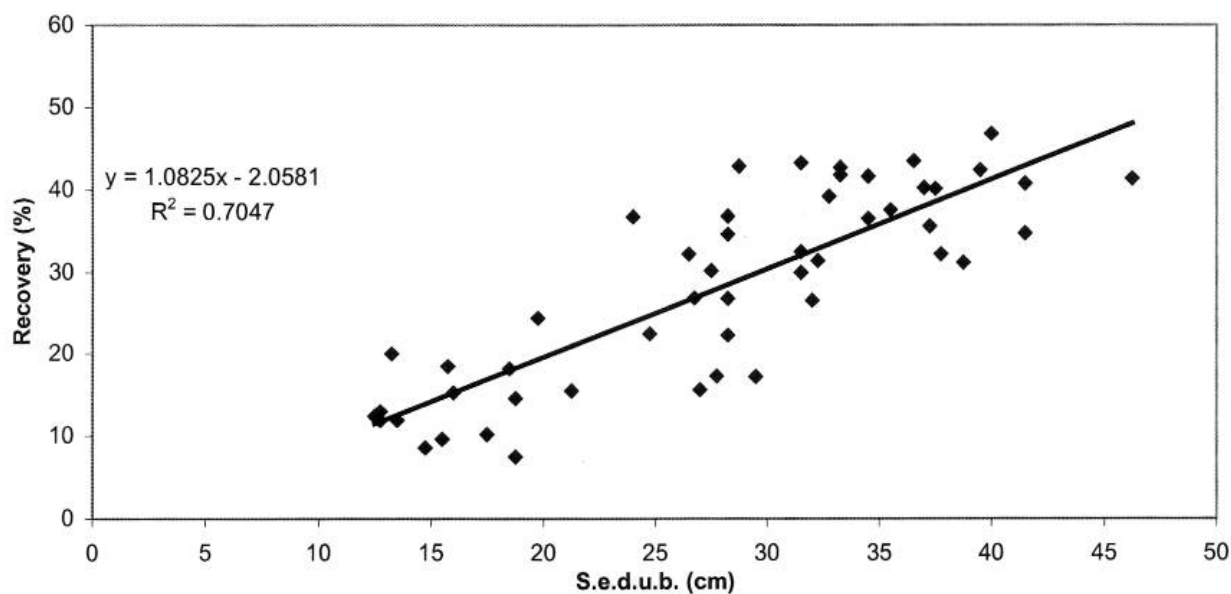
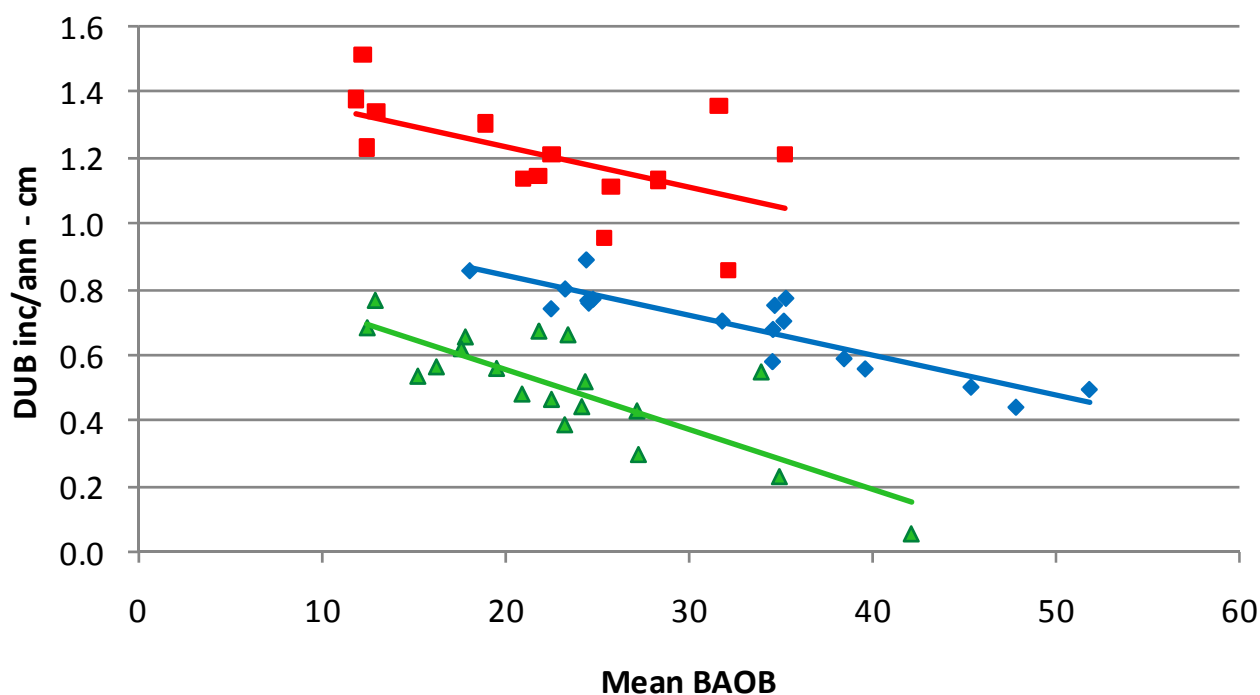


Figure 29: The recovery of structural grade karri in relation to log size from 30 year old karri regrowth (Brennan et al. 1999).



◆ Treen SI 50 Age 50 ■ Warren SI 50 Age 15 ▲ Sutton SI 30 Age 32

Figure 30: Diameter increment on the largest 100spha in karri stands of different age and site index at different stand density.

The influence of rotation length

The rotation length that has been used in the planning of karri yield in forest available for timber harvesting is 100 years, when many trees would have reached 1m in diameter. This is close to the age of maturity, provides logs of a size that industry is accustomed to and also maintains a large proportion of the forest of an age that is aesthetically attractive. Variation to this rotation length has been considered from several standpoints: to lengthen the rotation to increase the number of trees of hollow bearing age, to shorten the rotation to reduce the size of the trees in final felling to more closely suit the requirements of modern sawmilling, and to increase the yield of sawlogs in the short term to alleviate the shortfall in sawlog yield created by the protection of all old-growth forest.

The following considerations influence the choice of rotation length: the impact of a change on the structural stability of the forest (Bradshaw *et al.* 1997c; Bradshaw 2002); the relatively modest increase in hollow bearing habitat that would result from increasing rotation length (e.g. increasing rotation from 100 to 150 years in forest available for timber harvesting increases hollow bearing habitat by six per cent in the whole forest); that very large trees may not be usable for sawmilling in the future; that smaller trees are likely to have lower sawn recovery; and that shorter rotations in combination with legacy element retention (in the forest available for timber harvesting and formal and informal reserves) may provide greater benefits to economics, productive capacity and biodiversity than the adoption of a uniform change to rotation length.

The influence of market conditions

The existence of a market for karri chiplogs makes it possible to undertake commercial thinning as soon as the desirable bole length has been achieved and before the thinned trees are large enough to produce sawlogs. Early commercial thinning is important for several reasons: it provides an early return on investment that makes long rotation management more financially viable; it reduces the time for trees to reach sawlog size and utilises material that would otherwise die in the normal process of competition (Figure 31). Such utilisation also improves the economics and resource use from later thinning where it constitutes about 25-30 per cent of the thinning yield. Importantly, a market for chiplogs provides opportunities to recover costs (at least partially) associated with rehabilitation of younger, bushfire-affected stands.

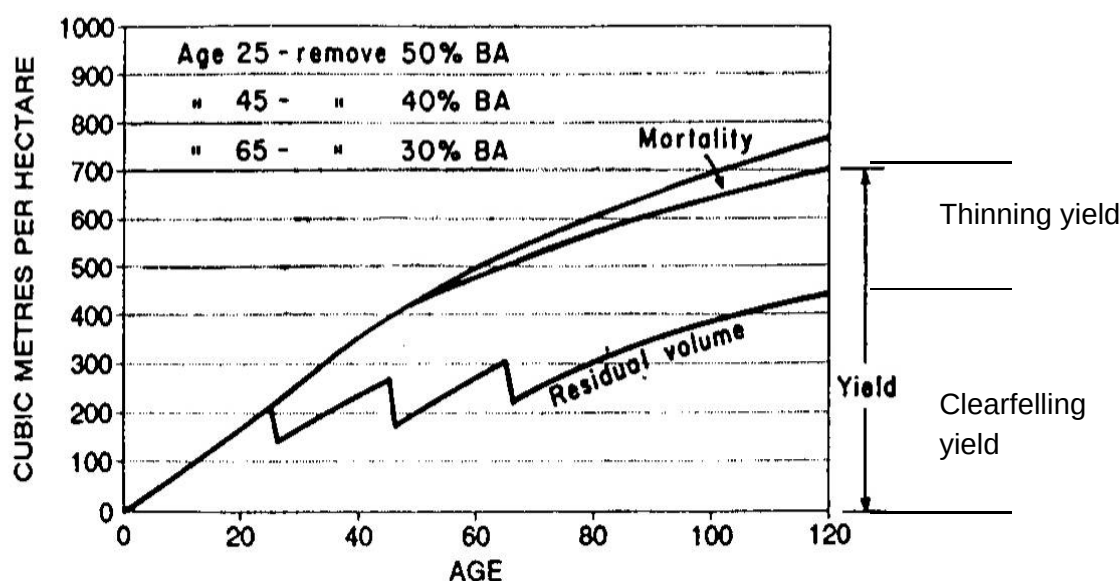


Figure 31: An illustration of the intermediate and final yield from a managed average quality karri stand. Comparison with Figure 8 shows the utilisation by thinning of volume on trees that would otherwise die during the rotation in an unthinned stand.

Influence of legacy tree competition and edge effects

In multi-aged stands, competition from legacy trees continues throughout the life of the regrowth stand. The influence zone from retained legacy trees increases as the size of the retained tree increases and as the age of the regrowth increases (Bradshaw 1985; Ellis *et al.* 1987). Rotherham (1983) showed that the competitive influence extended to twice the crown radius of veteran trees in 50 year old karri regrowth. On this basis, a 25 per cent crown cover of evenly spaced legacy trees would reduce the volume growth of the karri regrowth in that stand by 40 per cent. A similar suppressive effect has been shown for a number of other eucalypts, though the magnitude and the extent of the effect varies with species and site (Opie 1968; Bassett *et al.* 2001; van der Meer *et al.* 2007).

Desirable gap size for the establishment and management of regrowth is influenced by a number of factors, including the impact of various edge effects, by the practicalities and economics of regeneration burning, safety considerations and aesthetics. Groups of trees of different age growing alongside one another are influenced by a variety of edge effects. One tree height is sometimes used as a rule of thumb to represent the zone of influence of the surrounding forest (Forestry Tasmania 2009), but this needs to be qualified depending on the influence being considered (Bradshaw 1992; Wardlaw *et al.* 2012; Baker *et al.* 2013). Figure 32 illustrates a range of potential edge effects, the extent of which varies with species, site and the age of the adjoining stands.

Table 2 indicates the relationship between canopy gap size and an influence zone of one tree height.



Figure 32: A diagrammatic illustration of some common edge effects. In this hypothetical example the black bars indicate the potential extent of the influence zone. 1. Establishment of regeneration inhibited. 2. Regrowth suppressed by the overstorey. 3. Foraging zone (species 'x'). 4. Zone of seed shed from surrounding trees. 5. Zone of browsing damage. 6. Damage zone from the felling of surrounding trees. 7. Damage zone from intense regeneration burn. (Bradshaw 1992).

Table 2: The percentage of a gap of different size surrounded by trees 60m high within one tree height of the edge.

Gap Diameter (tree heights)	Diameter (m)	Area (ha)	Perimeter (km/100 ha)	% of gap within 1 tree height of edge ^a
2	120	1	38	100 per cent
4	240	4.5	17	75 per cent
6	360	10	11	56 per cent
8	480	18	8	44 per cent
10	600	28	7	36 per cent
12	720	41	6	31 per cent

^aBased on a circular gap. Other shapes will give rise to higher influence percentages.

Edge effects can be seen as being of positive or negative value, depending on the issue and the object of management. This is a major factor in the selection of gap or group size that is created for regeneration purposes.

Regeneration is suppressed and killed beneath the crown of a mature karri tree within two years of establishment. This influence extends as the regrowth develops and by 50 years the

suppressing influence extends to two crown radii (Rotheram 1983). That is, in most areas, all 50 year old regrowth trees within a 50m gap would be suppressed to some degree by surrounding mature trees (Bradshaw 1985).

Where natural seed is required for regeneration, a gap size of 120m (two tree heights) is the maximum gap size to ensure adequate seed coverage.

Felling damage has a significant edge effect, especially when mature trees may be 60-70m in height with a 25 metre diameter crown. While careful directional falling minimises the impact, much of the regrowth within one tree height of a mature edge is at real risk of damage. This is especially so when felling large veteran trees.

Edge effect also influences the use of fire for site preparation for regeneration. The removal of heavy fuel loads (slash) with fire can cause unacceptable damage to surrounding retained trees, especially where gaps are small. This can be minimised by the use of larger gaps which reduces the length of edge per ha of gap or area to be regenerated. The cost and difficulty of slash burning is also related to the number of burns and their total perimeter, accordingly a large number of smaller burns are more difficult to achieve than would fewer larger burns. In very small coupes it is generally necessary to heap the harvesting debris in order to burn safely (Ferguson *et al.* 1999).

Coupes where there is a larger proportion of edge effects are generally considered preferable for colonisation and recovery of the disturbed site (Forestry Tasmania 2009; Baker *et al.* 2011; Wardlaw *et al.* 2012; Baker *et al.* 2013). While the need for this has not been demonstrated with respect to most karri forest flora which regenerates from soil-stored seed, it is a factor in the re-colonisation of the fern *Asplenium aethiopicum*, though it has been shown to establish in a clearfelled site at least 400m from the edge (Wardell-Johnson *et al.* 2004; McCaw 2006). Proximity of edges have also been shown to assist in the dispersal of birds that prefer mature habitat (Wardell-Johnson *et al.* 2000; Atkinson 2003).

Although small coupe sizes are generally considered to be more aesthetically acceptable at a single coupe scale, this is not necessarily true at the landscape scale, where more, smaller coupes impact on a larger proportion of the landscape in any period of time.

While small gaps can be harvested and regenerated, the key determinant is the ability to manage those regrowth patches into the future. For example, 60 per cent of the regrowth in a two ha patch will be suppressed to some degree by surrounding mature trees and while some of these patches survive the future harvesting and burning of the surrounding forest, most small patches of regrowth are damaged or reduced in size to some degree. Attempts to manage regrowth in very small patches have been unsuccessful in the past. The reasons are more fully discussed in Appendix 1.

Gap size is ultimately chosen with all of these factors in mind.

The maximum gap size that may be used in clearfelling in the karri forest under the FMP is 40 hectares in selectively cut (uneven-aged) forest and 20 hectares in even-aged forest, limited primarily for biodiversity reasons (Ferguson *et al.* 1999; Conservation Commission 2004). In practice, gap size has been much smaller than that in recent operations (See Table 3). Smaller gap sizes to a large extent reflect the patch size available for harvest.

Table 3: Size of coupes harvested for regeneration in 2011

Coupe size (ha)	Number of coupes	Total area of coupes (ha)	Percentage of total area (per cent)
2- 5	15	51	12
5-10	13	89	22
10-20	11	149	37
20-30	2	44	11
30-50	2	72	18

(Source: SILREC)

Role of coppice

Karri has the capacity to coppice from stumps that have been harvested or in response to fire that kills the above-ground stem. The capacity for karri to coppice from large stumps is lower than for some other species such as jarrah.

Karri coppice develops from the stumps of most small trees that are removed in thinning, but it does not generally develop beyond a spindly whip because of suppression from vigorous retained trees, which expand into the spaces vacated by the thinned trees. If the stand was thinned very heavily, then coppice would continue to develop. In a thinning of high quality 12 year old regrowth, coppice development resulted in about a 10 per cent reduction in growth of retained trees for the first five years, but the effect reduced to zero thereafter (unpublished data from Warren thinning trials, courtesy J. Bradshaw). Coppice control with herbicides has not been considered necessary following thinning. However, it may be required following thinning in lower site quality stands where the growth rate of retained stems is less. It may be required in mixed stands where marri coppice will result from thinning, depending upon the management objectives.

Influence of branching and tree form

The majority of mature karri trees are single stemmed, straight trees with a long branch-free bole. However, this is something of an illusion in that as much as 60 per cent of the bole of a mature tree may contain occluded dead branches with little external evidence.

In the early stages of development, eucalypts shed their lower branches as they become less efficient and moribund. Branch death occurs following the development of a brittle zone and an abscission layer at the base. Dead branches that are less than about 25-30mm diameter at the base are shed cleanly by the pressure of the increasing stem diameter pushing against the branch stub (Jacobs 1955). Marks *et al* (1986) found that *E. regnans* branches that died before they had formed heartwood were shed cleanly. Branch size increases with height above ground. As the tree reaches pole size, apical dominance

reduces, the lower branches persist and increase in size. With increasing shading they eventually die and break off, to overgrow and become occluded within the stem. At still greater height the lower branches remain alive to shape the crown of the early mature tree (Figure 33).

In dense natural karri regrowth, the height of the bole below which branches are shed cleanly (the 'clean zone') is about 14 - 18m when the total tree height is about 30m. Branch size increases with decreasing stand density so that stands that are less than fully stocked will have larger branches and lower bole height. There is no increase in the 'clean zone' length at densities above about 3,000 spha in stands originating from seed (Bradshaw *et al.* 1991). Trees developing from nursery raised seedlings appear to have larger branches than those developing from transplanted wildlings or from seed at the same stocking rate. The cause has not been determined. Planted stands at 3000spha had an estimated 'clean zone' length⁸ of 13m, reducing to 11m at 1250spha (Wheatley spacing trial). Planted stands at lower SI (Nairn spacing trial) appeared to have a longer clean zone at the same spacing⁹.

Branches are an important cause of defect in sawn timber because of their effect on physical strength, appearance and the incidence of decay and insect attack (Donnelly *et al.* 2008). Dry and occluded branches were associated with more borer galleries of bullseye borer (*Phorocantha* (previously *Tryphocaria) acanthocera*) and brown-wood and rot than were green branches in karri regrowth (Brennan *et al.* 1999). Larger branch stubs also serve as initiation points for tree hollows (see Figure 25).

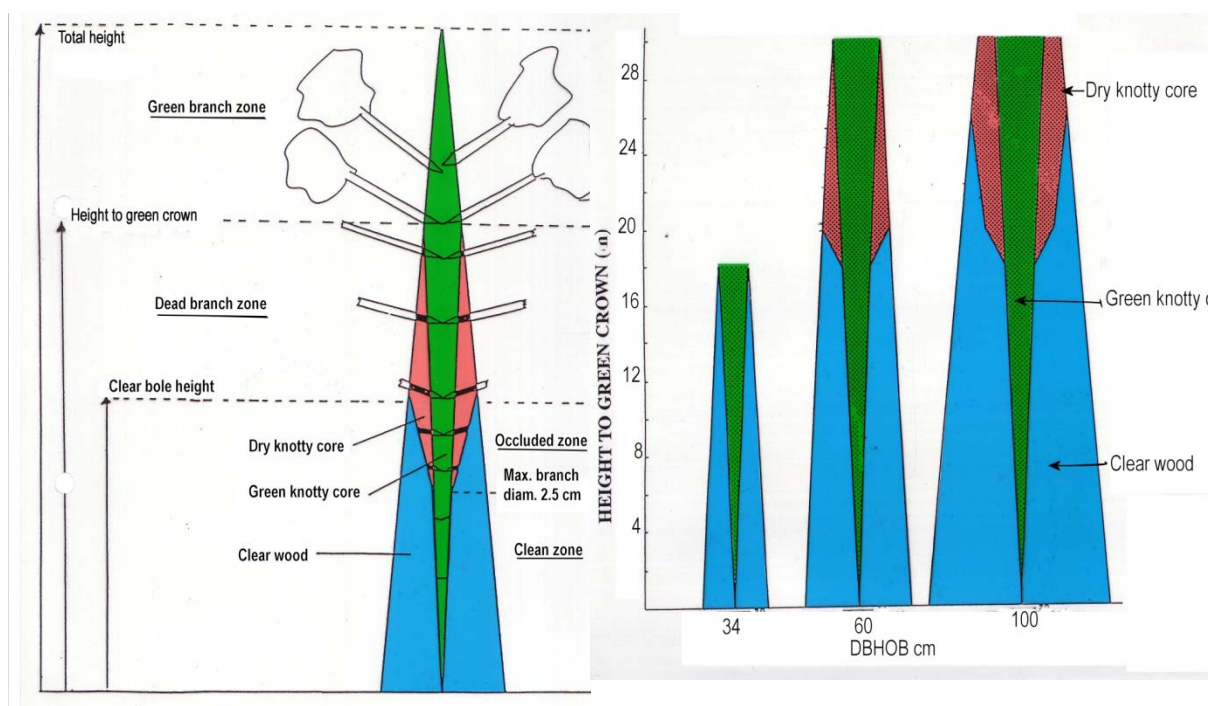


Figure 33: Diagrammatic illustration of the branching structure of karri.

⁸ Bole length to lowest dead branch larger than 25mm diameter.

⁹ Source: Wheatley spacing trials. Julian Geisel pers. comm.

As the tree grows and diameter increases, and as dead branches decay or fall, the occluded zone will increase upwards, reducing the length of, or eliminating the dead branch zone. The right hand figure illustrates the relative size of clear wood, green knotty core and dead knotty core with increasing tree size (Bradshaw *et al.* 1991).

Influence of insects and disease

Several wood rots affect karri, including those known as straw rot, white rot, white pocket rot and brown rot (Clarke *et al.* 1989). A condition known as 'brown wood' has recently been confirmed as incipient rot that will eventually develop into white rot or white pocket rot. More than one species of fungi may be associated with each of these rots (Davison *et al.* 2008). These do not affect the health of the forest, but are significant because of their impact on wood quality.

The most common entry points for wood rotting fungi are branch stubs, borer galleries and scars caused by fire, *Armillaria* or mechanical damage. Brown wood has become a source of concern in recent years, especially because it is found to be more common in regrowth than in mature trees (Donnelly *et al.* 2008). This is likely to be more a function of tree age than of current growing conditions. For example, many old trees have a rotten heart or pipe which represents the wood it laid down at the 'regrowth' stage that has since rotted away, while wood laid down when the tree was older is now sound. However, brown wood was raised as a concern in mature trees in particular areas (Boorara and Walpole) in the 1960s. The issue is now of greater economic importance because future sawlog supplies are required to come from younger and smaller regrowth trees than in the past. This is a common issue with the utilisation of young eucalypts throughout Australia.

The three main insect pests that affect the wood of karri are the bullseye borer *Phoracantha* (formerly *Tryphocaria*) *acanthocera*, the witchetty grub (*Xyleutes* spp.) and the pin hole borer (*Atractocerus kreuslerae*) (Clarke *et al.* 1989). The latter most commonly attacks trees about two years after they have been damaged (dry-sided) by severe fire. While it does not affect strength, it affects appearance and thus wood end-use.

Bullseye borer causes significant degrade in regrowth logs as a direct result of the galleries and as a source of fungal infection. A survey of fully stocked regrowth plots across the range of karri indicated incidence was associated with proximity to mature forest (small coupe size) and with marginal sites. No incidence of borer damage was found in regrowth younger than 14 years (Abbott *et al.* 1991). It is likely that the onset at this age is a symptom of the severe inter-tree competition and moisture stress that occurs from this stage of stand development.

Farr *et al.* (2000) found a weak association between the incidence of bullseye borer and lower rainfall sites, small coupes and proximity to jarrah/marri forest. Dominants and co-dominants were more severely affected. They found that 15-25 per cent of small sawlogs were rejected due to borer galleries and associated rot or incipient rot.

The presence of *Armillaria* (Section 4.3.1) is relevant to silvicultural practice. Timber harvesting, in particular thinning which produces a large number of dead stumps for the fungus to colonise, has the potential to increase the effect of the disease. For example, fifteen years after thinning in an area with a 50 per cent infection rate, the volume loss due to defect accounted for 50 per cent of that gained by the thinning response (Robinson 2004). Removal of stumps from infected areas at the time of thinning can be reduce the effect of the

disease, but results in high levels of soil disturbance. Delaying thinning, thereby delaying the creation of many dead stumps may be an alternative (R. Robinson, pers. comm.).

Options for regeneration

There are several options available for the regeneration of karri forest.

Seed trees

While seed tree regeneration has not been used for several years, the following points are key to its success.

Seed-crop assessment is required to determine the stage of the seed cycle and the potential seed crop, to decide whether or not seed tree regeneration is feasible in a particular year. Where seed tree operations are planned, seed crop assessment is also required prior to the regeneration burn to confirm that the required seed crop is still in the capsules and to determine their viability (CALM 1997). Areas planned for seed tree regeneration should not be burned in the five years prior to regeneration burning, to avoid disrupting the seed cycle and removal of understorey that is desirable for a more even regeneration burn. Harvesting to seed trees prior to or during flowering will reduce the potential for cross pollination and may result in excessive loss of flowers due to exposure. Seed requirements are indicated in Table 4.

Table 4: Seed requirements for seed tree regeneration at the time of the regeneration burn

Soil type	Spring burn – seeds per hectare	Autumn burn – seed per hectare
Red 'karri loams' (generally pure karri)	120,000	90,000
Podsols (generally mixed stands)	180,000	135,000

Seed trees should be wind-firm dominant or co-dominant trees with a healthy spreading crown, free from hereditary defect such as forking or excessive grain deviation. They should be as evenly spaced as possible at the rate of 3 - 4spha, increasing to 6spha in stands of below average height or in stands with past fire damage (White 1974b, 1974a). Marri seed trees are not required in karri/marri stands since adequate marri regeneration will develop from existing lignotubers (White 1971).

The ideal seed tree regeneration burn is one that scorches but does not defoliate the seed trees (Christensen 1971b). The regeneration burn has multiple objectives; to create the maximum receptive seedbed and ashbed; to remove harvesting debris; to temporarily remove understorey competition to provide short term advantage to karri seedlings; to stimulate understorey regeneration, particularly nitrogen fixers, from the soil stored seed-bank; and to induce rapid seed fall from the seed trees.

Seed trees should be removed within two years of regeneration to avoid suppression of the regeneration beneath their crowns and while seedlings are flexible and less liable to felling and snagging damage. Seed trees that may have been damaged in the regeneration burn will not have degraded substantially within that time.

Seed tree regeneration is typically not satisfactory where there has been excessive soil compaction or where broadcast burning is not possible. Broadcast burning creates more even ash bed and more even heat to stimulate seed fall.

Planting

Planting with nursery raised seedlings is now the most common method of regenerating clearfelled karri areas. Preparation for planting requires the harvesting debris to be removed by burning or physical removal and the ripping or scarifying of any soil that has been excessively compacted. The objectives of the burn are the same as for the seed tree method, except that seed fall is not relevant and the surface soil conditions (ashbed and surface tilth) are less demanding than for seed regeneration. While ripping is important on any excessively compacted soil, it also increases the growth rate of seedlings on undisturbed soil.

In more recent years, smaller coupes and concerns from vignerons about smoke taint of their grapes has led to a change of practice in the regeneration burning technique. Rather than broadcast burning in spring/summer or autumn, harvest debris is now pushed into heaps and the heaps are burnt after the first rains in autumn or early winter. This means that small scattered coupes can be burnt more safely and smoke is generated after the grapes have been harvested. The disadvantage of this practice is that more fertiliser is needed at the time of planting, and unless CWD is specifically excluded from heaps, it will be burnt. The impact on understorey regeneration of removing fire and increasing mechanical soil disturbance on a significant proportion of the coupe has not been investigated.

Where a clearfelled coupe is not surrounded by recently burnt forest, pushing in the harvest debris from the edge by about 20m increases the ability to contain the regeneration burn (Anon 2004).

The planting rate and mix may be varied according to site and the species composition, with the object of minimising branch size and maintaining the species that existed on the site. There is no intention of maintaining the same mix in the regeneration as was in the mature stand. Species mix is a naturally dynamic process where species representation changes over time and it is expected that stands will adjust over time, provided that the species are adequately represented. It is not usually necessary to plant marri in karri/marri stands since it will regenerate adequately from lignotubers (White 1971). Regeneration monitoring indicates that ripping has not adversely affected marri lignotuber stocking (A. Seymour, pers. comm.).

Planting at 3000spha will minimise branch size and maximise the length of the clean bole (Bradshaw *et al.* 1991), but this can be adjusted downward where marri lignotubers are present because of their additional contribution to seedling density. Lesser density may be appropriate where site index is lower. Where 'enrichment' planting is carried out in mixed stands to ensure a representation of karri, it is preferable to plant karri in small groups at

normal spacing rather than planting evenly over the site. This will improve the form of the karri and more closely reflect natural conditions.

Retained marri will suppress seedlings within their influence zone, so there is no advantage in planting within 7m of a retained veteran marri.

Nursery raised seedlings which have received fertiliser in the nursery prior to dispatch typically do not require fertiliser to be applied in the field at planting. However, seedlings planted within 'pushed in edges' require additional fertiliser at the time of planting for them to match the growth of the broadcast burnt area. Additional fertiliser at time of planting is routinely applied where heaping of debris is practiced to offset the lack of ashbed.

Where the stocking is less than the acceptable level, supplementary planting or replanting may be required. Experience has shown that planting in the following winter into already established understorey has a poor record of survival and growth. It is preferable to rip the site before replanting to alleviate compaction (a common cause of the original failure) and temporarily reduce understorey competition.

Broadcast seeding

Broadcast seeding can be used successfully on a broadcast burnt seedbed. Unpelleted seed applied at the rate of 45,000 seeds per ha in March/April has been shown in the past to be effective (Annels 1980). The high usage (15 times that of planting) of a limited seed supply has restricted its application.

4.2.2 Fire management

Fire has an important role in various aspects of silviculture. Different silvicultural objectives have different fire requirements.

Regeneration burns

The objective for broadcast regeneration burning is to burn with sufficient intensity to create a satisfactory seed bed and remove excessive harvesting debris. Where seed tree regeneration is practiced, fire is used to promote rapid seed fall onto a receptive seedbed. For broadcast burning, a more even burn is achieved where 'scrub-rolling' has been done. The success of the regeneration burn is influenced by the SDI and the fine fuel moisture content, as shown in Table 5 (Anon 2002).

As explained above, debris is now pushed into heaps for burning to prepare for regeneration. This practice is more manageable for burning very small coupes that are surrounded by heavy fuel and affords protection to retained trees during the burn.

Table 5: Fuel moisture conditions required for regeneration burning

SDI ^a	Success rating of the burn				Rating scale	Minimum required for:
1500-2000	G	VG	EX	EX	EX - excellent	Seed trees
1000-1500	M	G	VG	EX	VG - very good	Seed trees
500-1000	P	M	G	G	G - good	Planting
250-500	VP	VP	P	M	M - moderate	Heaped areas where re-burning is planned
0					P - poor	
Moisture content of fine fuel	20-25 per cent	15-20 per cent	10-15 per cent	5-10 per cent	VP - very poor	

^a 1 unit = 0.1mm of rainfall

Protective burning of regrowth

For the reasons described in section 2.6, karri regrowth is protected from fire for at least 20 years after establishment, by which time it may have accumulated about 30 tonnes per ha of dead fuel. The opportunity for burning these stands is limited because by the time the fuels dry out, summer is well advanced and the elevated fuels make burning without damaging regrowth (McCaw 1986, undated) or the burn escaping difficult. Opportunities for burning are improved after thinning when elevated fuels have been reduced and reduced canopy density causes drying to occur as early as October. However, with the addition of thinning debris, total fuel loads may be as high as 75 tonnes per ha (McCaw *et al.* 1996).

Stem damage in post thinning burning is related to heat release rather than fire intensity and McCaw *et al.* (1997b) found fire damage was acceptable in fires that consumed less than 50 tonnes per hectare. Most damage was caused by the burning of CWD within 1m of retained stems. Autumn burning caused unacceptable damage.

Table 6: Parameters for post thinning burns in > 30m top height regrowth in spring (Source: McCaw et al 1997).

SDI ^a	Weather			Fuel moisture content				
	Maximum Temp °C	Min RH per cent	Wind speed ^b m / sec	Elevated leaves per cent	Profile litter per cent	FROS ^c m/hr	Max. flame height m	Scorch height m
<40 0	20-25	60-45	<3.0	12-18	>90	15-25	1.5	<10

^a 1 unit = 0.1mm of rainfall

^b Wind speed at 10m height in the open.

^c Forward rate of spread.

Operational application of post thinning burning remains problematic. Thinned regrowth and adjacent unthinned stream (and other informal) reserves are a complex mosaic of different fuel conditions, and the weather conditions under which both can be safely burnt are quite limited. Fuel conditions are more uniform prior to thinning but the potential for stem damage is greater. If significant fuel reduced buffers are to be created, greater damage to regrowth may need to be traded off against strategic protection objectives. In either situation, opportunities for burning are limited and made more so by opposition to burning by vigneron in early summer.

The risks associated with leaving thinned areas unburnt are mixed. Relative to unthinned stands, thinned stands have less elevated but higher fuel loads, easier access and visibility, earlier opportunities for burning in spring, a longer burning-conductive season and higher internal wind speeds. The risk of introducing fire under difficult conditions must be weighed against the risk of bushfire and associated damage in more extreme conditions.

4.2.3 Summary of silvicultural methods in the karri forest

Determining the silvicultural objective

Areas available for timber production consist of stands that have been previously harvested either under a clearfelling or a group selection system. They are either even-aged or uneven-aged stands varying from pure karri to mixed marri/karri/jarrah stands. Stands are defined as even-aged regrowth if they have less than 15 per cent canopy cover of mature trees. The regrowth in the even-aged stands generally dates from 1930 to 1945 and is mainly pure karri, while the uneven-aged regrowth dates from about 1935 to 1967 with a greater species mixture.

The objectives of silviculture in the karri forest used for timber production are to:

- thin stands of regrowth and promote the growth of the retained trees, or to
- clearfell and regenerate stands of mature trees.

Promoting growth (thinning)

The primary objective of thinning in the karri forest is to concentrate stand growth onto fewer trees, thereby increasing their diameter growth, reducing their time to reach sawlog size and increasing the proportion of the total production in the form of sawlogs. However, it also achieves a number of other outcomes including; an intermediate yield of wood suitable for chipwood from trees that would otherwise be unused; provides an early income to improve the economics of management; and improves the health of the forest by reducing stand density and moisture stress.

Thinning objectives

The present thinning regime focuses on maintaining high stand density until the maximum length of the 'clean zone' (Figure 34) is achieved (i.e. until the co-dominant height reaches 30m) and then thinning to dominants and co-dominants. Stands reach a top height of 30m from 15 to 50 years of age depending on the site (Figure 9). The second thinning is envisaged when the stands have grown to the point where an economical thinning can be achieved, but before suppression density is reached, typically about 15 years after the first thinning. A third thinning is possible, but the point will eventually be reached when all but the most conservative thinning will result in a reduction in potential growth. While the first thinning produces mostly chipwood, subsequent thinning will produce higher proportions of sawlogs.

While thinning schedules generally aim to maintain stand density between the 'critical' density and the suppression density, this may be varied depending on the overall management objectives, the balance needed for different objectives and the stand age. The present thinning intensity is aimed at maximising total volume production, including both sawlogs and chipwood.

Figure 31 illustrates the relationship between thinning and final yield. In this management regime, about 40 per cent of the total yield comes from intermediate thinning.

Variation to thinning schedules should be considered to cater for different sites and changes in wood flow requirements. Stands could be thinned more heavily at first thinning to increase the rate of diameter growth of the crop trees so that they reach sawlog size sooner. While

this would be done at the expense of total volume growth, it would provide a higher sawlog yield in the critical shorter term when regrowth stands are too young to produce large sawlog volumes. Lower site quality stands that are in greater need of thinning could be thinned before they reach 30m top height, as could those planted stands where the 'clean zone' has already become established because of heavier branching. The uniform application of the relationship between top height and thinning intensity across all sites should be reviewed as increasing areas of lower site index stands become available for thinning.

Thinning practice

Stands of regrowth are mapped as even-aged if they are more than two hectares in size and if they contain less than 15 per cent crown cover of veteran trees (Figure 34). At present these are scheduled for commercial thinning (primarily for chipwood) when they reach a top height of 30m. Coppice control is generally considered unnecessary. The past rate of thinning is illustrated in *Figure 38* *Figure 40* *Figure* .



Figure 34: A diagrammatic illustration of an even-aged stand that is subject to thinning. Patches of regrowth larger than two hectares with less than 15 per cent canopy of veteran trees are regarded as even-aged stands. Large veteran trees are generally retained in thinning to provide diversity and to avoid damage to regrowth by their falling.

Harvesting for regeneration

Stands of mature karri trees that are not suitable for thinning are clearfelled and regenerated. While mature even-aged stands will develop from existing regrowth stands in time, all of the stands that presently contain sufficient mature trees are uneven-aged and have been previously cutover under some form of selective timber harvesting, and contain patches of regrowth in gaps that vary from less than 0.5x tree heights to 2x tree heights.

For the reasons given in Appendix 1, selective harvesting of stands with small patches has not been successful in the past and these stands are clearfelled and regenerated as even-aged stands (Figure 35). Patches of regrowth less than two hectares in size that occur within these stands can rarely be protected from felling or subsequent regeneration burning. Larger

patches of regrowth that can be protected from felling and from the effects of the regeneration burn are excluded from clearfelling and thinned.

The inability to sell marri chipwood since 2000 has had a severe impact on the ability to regenerate some stands. To maximise regeneration, these trees could be removed by culling but compromise is required between the cost and waste of culling, the loss of species and structural complexity and the requirement for regeneration. Where the canopy cover of marri exceeds 30 per cent (about 12 m²/ha), excessive culling would be required to achieve satisfactory regeneration and culling is usually considered inappropriate. Where the canopy cover is less than 30 per cent, culling to reduce it 15 per cent (6 m²/ha) will enable a regrowth stand of acceptable quality to be created. Consideration could also be given to felling some of these marri after the regeneration burn to create CWD, if required.

Retention of marri in varying proportions will have a considerable impact on future yield, and the existence of partially clearfelled stands is mapped and provided for in yield estimates.

A diagrammatic illustration of timber harvesting and regeneration in relation to different stand structure is shown in Figure 35.

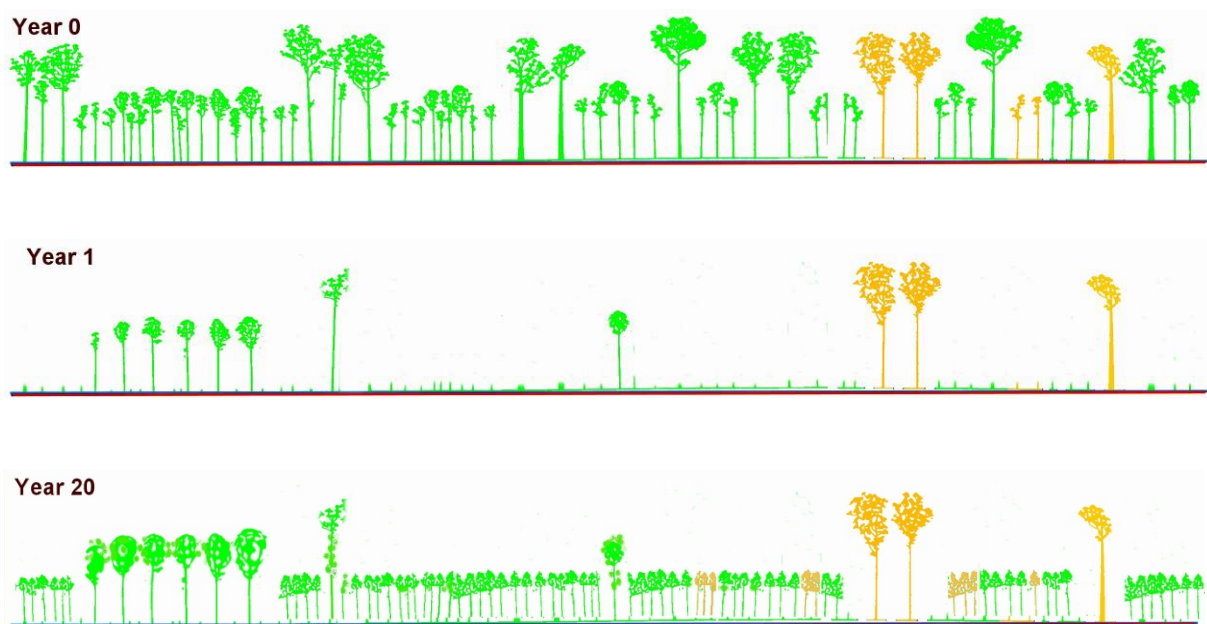


Figure 35: Year 0 is a representation of the range of structure that may occur in stands that have been selectively cut in the past. Year 1 shows the situation after harvest where protectable patches of regrowth are thinned (left side of figure) and remaining areas are clearfelled with exception of potential habitat trees. Year 20 illustrates the development of the retained and the new regrowth and the suppressing effect of the retained marri. Regrowth is due for first thinning at this stage. Karri is represented as green, marri as brown.

Although it is not common, there are some stands consisting of a mixture of karri, marri and jarrah. Mixed stands are defined as those with 5-20 per cent canopy cover of karri within a

predominantly marri/jarrah stand. Karri and jarrah predominantly occur in a fine mosaic rather than being evenly distributed. These stands are harvested in the same way as karri/marri stands, but steps are taken in the regeneration process to ensure that jarrah is maintained in the mixture of the future stand.

Site preparation and regeneration

Current regeneration practice consists of rough heaping the harvesting debris, followed by burning and planting with karri. Where excessive soil compaction is evident, shallow ripping is undertaken between the heaps prior to burning. On some larger coupes, broadcast burning may be used instead of heaping. In these cases, it is common practice to 'push in' the harvesting debris around the boundaries of the coupe to assist in containing the regeneration burn and protecting adjacent areas from fire damage.

Planting density is normally 2200spha, aiming at a stocking rate exceeding 85 per cent at 1666spha. Seedlings are fertilised prior to nursery despatch (Hewett 1991) and given additional fertiliser at time of planting on pushed-in sites.

Where regrowth stands have been damaged by fire, stems between 15 and 25m may be deliberately coppiced, but most will develop multiple stems from the stump (Figure), right photograph). Clearing and replanting these stands will produce a better outcome at slightly higher cost where salvage is not possible.

Monitoring regeneration success

Regeneration success is monitored after the first summer following establishment (CALM 1990). Seedlings, whether from seed or nursery stock, are most vulnerable in the first summer.

There are a variety of methods that can be used to monitor regeneration density and stocking (Lutze 2001). The system that provides the most information and which is used in the karri forest uses a grid sample framework and measures the density of seedlings at each sample point (seedlings per hectare). These are then analysed to determine the percentage of points that contain a density greater than a nominated density (per cent stocked at or above a specified level). By measuring the actual density of seedlings at a point rather than using a stocked quadrat method, it is possible to determine the percentage of the area stocked to any given level. For example, a stand may be 80 per cent stocked at >3000spha or 95 per cent stocked at >1000spha. Average density is not relevant because it provides no indication of variability or the proportion of the area that reaches a desirable level of site occupancy. Point density is estimated using the tessellated triangle method (CALM 1990; Ward 1991).

The following definitions are currently used in the karri forest:

- *Optimal density*: the density at which crop trees will develop the maximum possible length of 'clean zone'. Optimal stocking at a point is achieved with a density of 3000spha or more at age 1
- *Adequate density*: the density at which crop trees will develop less than the maximum possible 'clear zone' length but still provide a full stocking of acceptable

crop trees at first thinning. Adequate stocking at a point is achieved with a density of 1666-3000spha at age 1

- *Understocked*: that at which the bole length of crop trees at first thinning will be unacceptable. An understocked point is achieved with a density of less than 1666spha at age 1.

The percentage of the area in each category provides the basis for predicting future yield and for deciding on remedial action.

4.2.4 Climate change

The predicted reduction in and changes to seasonality and reliability of rainfall has the potential to affect growth rate and sustained yield in the very long term. Most impact is likely to be felt in the drier north and east margins of the karri forest (Maher *et al.* 2010). The allowable cut for karri sawlogs under FMP has not altered substantially from the FMP 2004-2013, despite the calculations for this plan explicitly incorporating the projected impact of 'high severity' climate change scenario (see(Maher *et al.* 2010) based on CSIRO (2007)) in modelling sustained yield. The relative impact of declining growth rates on the level of sawlog sustained yield is considered higher for stands which have a high proportion of smaller trees yet to attain minimum sawlog size, than for stands which have a higher proportion of standing trees already in sawlog size classes. Therefore, climate change will have less effect on yield for the next several decades compared to other factors that affect yield, such as utilisation standards and the timing of successful regeneration and thinning.

The long term effect of projected changes in rainfall, atmospheric CO₂ and temperature is uncertain. While it is possible that increasing CO₂ concentration will increase water use efficiency and growth rate, it is uncertain which factors will ultimately become limiting since the interactions between water use, CO₂ and water availability are complex (Keenan *et al.* 2013). Physiological modelling has been used to assist in the understanding of the relationships between rainfall, atmospheric CO₂ and temperature in the jarrah forest.

The use of a physiological growth model which uses rainfall and temperature as inputs, when combined with future climate projections of rainfall and temperature, is an approach used to quantify the possible decline in growth rate for the sustained yield calculations. The physiological growth model 3-PG¹⁰ (an acronym for Physiological Processes Predicting Growth) has been widely used around the world to simulate the growth of even-aged forests, and the department has calibrated the model for use in both jarrah and karri forests. Predictions of the growth in standing volume over time, made using the calibrated 3-PG model (using the historical rainfall and temperature recorded at nearby meteorological stations), have been consistent with the volumes measured in plots.

This modelling did not take into account the potential effect of increased CO₂ concentration leading to increased growth, or explore how varying the timing and intensity of future

¹⁰ The 3-PG model is a canopy carbon balance model which simulates tree and stand growth on a monthly time step. It comprises a number of sub-models for net primary production, biomass allocation, stem population dynamics, and soil water balance. It is described in Landsberg and Waring (1997) and Landsberg and Sands (2010).

thinning events might be able to modify soil water demands and potentially mitigate some of the adverse impacts on tree or stand growth (Rayner 2013). Other broad-scale modelling for the south-west forests (ABARES 2011) predicts that while growth could be expected to reduce with the predicted higher temperatures and lower rainfall, this may be offset by the increased 'fertiliser' effect of increased CO₂.

Climate change has implications for site potential. Reducing rainfall and increasing potential evaporation will cause increased competition for moisture, increased moisture stress and a lower maximum LAI (Croton *et al.* 2015). A new equilibrium stand density will eventually be reached through mortality. The pattern of this adjustment and the time it will take to adjust to these new conditions is presently unknown.

Thinning reduces stress, improves the health of the remaining trees and has the potential to increase the growth rate of future sawlogs (Stoneman *et al.* 1989b). Thinning is therefore the highest silvicultural priority in forests under water stress. Thinning can achieve a sustained yield of sawlogs and alleviate any long-term reduction in growth rate or increase in mortality that may result from reducing rainfall.

4.3 Maintenance of ecosystem health and vitality

4.3.1 Pests and disease

There are few organisms that impact adversely on the health of large areas of the karri forest. The most significant is the naturally occurring root disease caused by *Armillaria luteobubalina* (Pearce *et al.* 1986). This pathogen infects the roots of trees by root-to-root contact and gradually spreads to the lower bole, forming a scar that is often mistaken for a fire scar, and leading to more extensive internal rot. If the growth of the organism extends around the bole at ground level the tree will die rapidly and some trees may fall due to lack of root support. Dominant and subdominant trees are equally affected and trees of all ages may be affected, though older and larger trees generally take longer to be killed. *Armillaria* grows in the dead material of stumps as well as in living trees. Infected trees may not show symptoms for many years. Fruiting bodies are distinctive but ephemeral, making disease mapping unreliable (Robinson *et al.* 2003). Although widespread in the forest, it first came to prominence in the 1970s when some areas were observed to be more heavily affected than others. The reason for the difference is unclear. While *Armillaria* affects the health of individual trees, it is mostly an economic problem.

Phytophthora cinnamomi, the cause of jarrah dieback, is an introduced root rot fungus that also occurs in the karri forest. While karri does not show effects of the disease in the field, some of the understorey species are susceptible. While the effect on the understorey is low, the principal concern is that karri sites can be infected without showing obvious symptoms and are therefore a potential source of inoculum and disease spread.

(*Uredo rangelii*) is a rust fungus that was recently introduced to eastern Australia and has spread rapidly on the east coast. It is a serious pathogen that affects a wide range of Myrtaceae, a major component of Australian flora. The disease results in malformation in the shoots of young plants and new growth causing a reduction in growth and possibly the death of seedlings. Myrtle Rust also affects flowering parts of some species. The host list and effect on susceptible species under Australian conditions has yet to be fully determined. A

list of Western Australian species considered susceptible to the fungus can be found in the *Myrtle Rust Incursion, Preparedness and Response Plan* (Department of Parks and Wildlife 2015). The principal mode of spread appears to be via infected nursery stock, on animals and human clothing and equipment. The disease has not been detected in Western Australia (as at early 2015), but it has the potential to cause serious damage to eucalypts and a wide range of understorey species should it be introduced here (DAFF 2011). Notable, *Agonis* species (such as Warren River Cedar and WA Peppermint) are known to be susceptible from laboratory trials.

4.3.2 Fire management

Prevention of serious damage by intense bushfire is a major consideration in the maintenance of forest health and vitality. Karri may be considered to be moderately sensitive to fire, and bushfires may vary from minor, to severe stand replacing events, wherein almost all of the existing trees are killed. Very High and Extreme fire danger conditions are experienced every summer in the karri forest, though less frequently than in the jarrah forest. Ignitions from both human and natural sources occur every summer. Fires that occur in severe weather conditions where heavy litter fuel loads exist cannot be effectively suppressed once they become established, regardless of the suppression effort, a fact that has been repeatedly demonstrated in Australia and elsewhere.

The principal means by which the success of fire suppression may be improved and fire damage minimised is by strategic fuel reduction (planned, prescribed) burning.

Prescribed burning is carried out in the karri forest to:

- reduce fuel loads in order to improve the ability to control bushfires and reduce their damage
- regenerate the forest after timber harvesting
- maintain nutrient cycling
- regenerate understorey for biodiversity reasons.

Fires can be generally be controlled when the fuel load is less than 15 - 20 tonnes/ha, a figure that is reached at about five years after a previous fire in karri forest depending on forest density and site (Sneeuwjagt *et al.* 1985). Much of this fuel is in the form of bark and twigs known as trash fuel. Unplanned fire in fuels of 25 tonnes per ha (about 15 years after previous fire) can cause tree mortality. Prescribed burning aimed at improving the opportunity for bushfire control would be at a frequency in the order of 5 to 10 years, but for various reasons, this is not consistently achieved.

Prescribed burning in the karri forest is more difficult than in the jarrah forest because of higher fuel loads, fewer days when conditions are suitable for burning, and a more complex mosaic of vegetation types, each with different burning characteristics. The issues involved in the first prescribed burn under regrowth karri forest have been discussed in Section 4.2.2.

The frequency of burning for biodiversity is more complex since no particular regime is suitable for the all of the organisms involved (Burrows *et al.* 2003; Friend *et al.* 2003; van Heurick *et al.* 2003). Variation in fire regimes is the broad objective for this purpose, although more specific regimes may be used in some circumstances. The issues involved and the

resolution of the sometimes conflicting objectives are discussed in the *Special Fire Edition of Landscape* (CALM 2000).

Figure 36 indicates the time since fire for different proportions of the forest. This suggests that in about 25 per cent of the karri forest, a direct attack on a bushfire in severe weather conditions would have a good prospect of success. Bushfire under severe weather conditions is likely to cause severe damage and mortality in about 50 per cent of the forest. The proportion of the forest that is available for timber harvesting that has not been burnt for a long period represents those areas of young regrowth that are protected from fire until they are old enough to withstand their first prescribed burn (Figure 26).

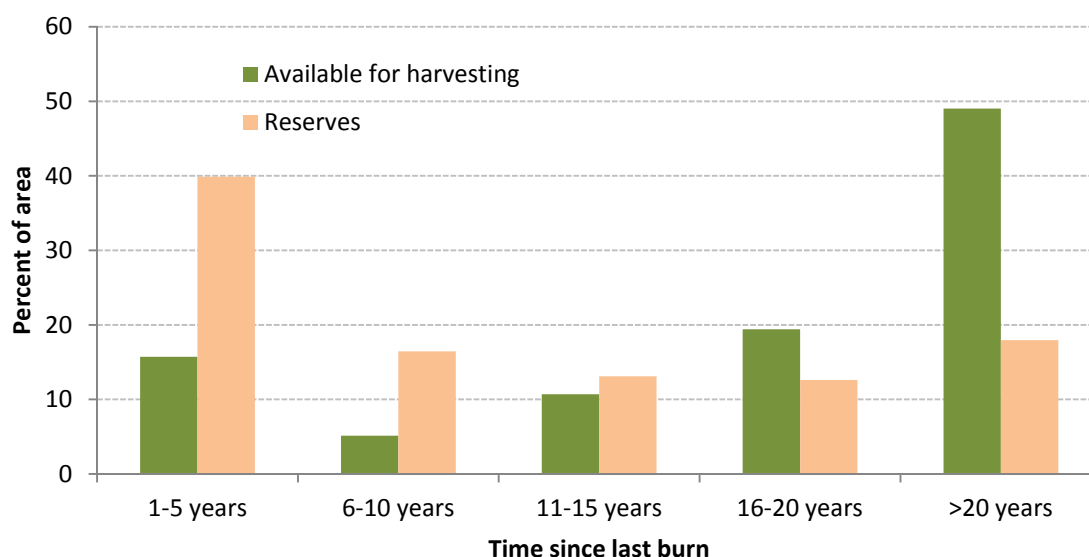


Figure 36: The time since the last burn in the karri forest at 2009. The percentages sum to 100 per cent for each of 'reserves' and 'available for timber harvesting'. (Source: FMIS database).

4.3.3 Climate change

Should the current reduction in rainfall in south-west Western Australia be sustained long term there will be a widespread effect on forest areas through increased drought stress and reduced carrying capacity of stands in lower rainfall areas. A combination of reduced rainfall and increasing evapotranspiration is predicted to result in a reduction in the maximum Leaf Area Index that can be supported throughout the forest (Croton *et al.* 2015). Declining rainfall since the 1970s (Table 1) has reduced stream flows and lowered water tables (Kinal *et al.* 2012). Changes to stream flow are felt by stream biota (Penniford 2013), followed by the understorey which is subject to moisture stress for 1-2 months longer than the deeper rooted trees (Crombie 1992).

Under a medium severity scenario for climate change (CSIRO 2007), an estimated 4.4 per cent (7,000 ha) of the publicly managed karri forest would be in areas receiving less than 900 mm annual rainfall, less than that generally needed to support karri forest (Figure 2). The tingle forest is likely to be the most severely affected (Maher *et al.* 2010).

Moisture stress can be reduced and forest health improved by thinning to maintain stand density closer to the critical density. This is more important in the lower SI stands on the drier fringe of the karri forest (Figure 28). The continuing availability of a chipwood market for small thinning is an essential means of facilitating such a program.

The effect of climate change on pests and pathogens in the karri forest is uncertain. It is likely that the incidence and severity of bullseye borer (*Phorocantha acanthocera*) will increase on marginal sites, (Abbott *et al.* 1991; Farr *et al.* 2000) but this will affect forest productivity rather than being a threat to the health of the forest.

Climate change will also affect fire weather conditions. Most modelled scenarios of the effect of climate change on fire weather in Australia predict an increase in the severity of burning conditions and the risk of extreme fire danger events (Williams *et al.* 2001; Lucas *et al.* 2007; Pitman *et al.* 2007). While no detailed investigations of potential effects have been carried out for karri forest, issues of particular relevance are any increase in the incidence of lightning ignitions and any further incursion of tropical cyclones down the west coast, south of latitude 30°S (McCaw *et al.* 2003; Maher *et al.* 2010). Other factors that may be of greater significance include changes to weather patterns that affect the ability to carry out prescribed burning. A mitigating factor is the predicted long-term reduction in LAI which has the effect of reducing fuel accumulation rates and fire behaviour.

4.4 Conservation and maintenance of soil and water

4.4.1 Soil

The maintenance of soil structure is one of the most important elements of sustainability. Soil structure can be adversely affected by soil compaction and the mixing of soil profiles during timber harvesting and other disturbance operations when the soil is moist (Bradshaw 1978; Rab *et al.* 2005). This type of damage adversely affects germination, seedling survival and growth of both overstorey and understorey. While compaction may be alleviated by ripping (Schuster 1979), profile mixing or topsoil removal cannot. Although the red earths of the pure karri forest are much less vulnerable than the podsols of the karri/marri forest, soil damage may still occur. While different machinery and snagging techniques have varying effects, damage is minimised by restricting machine activity during moist soil conditions (Rab *et al.* 2005; Department of Environment and Conservation 2010; Whitford *et al.* 2012).

4.4.2 Water

There are several aspects to the relationship between silviculture and water. These relate to sedimentation, salinity, and stream flow.

Sedimentation

The principal source of sedimentation in forest operations is from unsealed roads (Croke *et al.* 1999). Careful siting of roads, minimisation of stream crossings and appropriate drainage can reduce this to acceptable levels. Sedimentation from the general harvest area is an order of magnitude less than that from roads and is more dispersed. Nevertheless, it can result in increased sedimentation for a period of 2 to 3 years if disturbance through the stream zone occurs, particularly in winter. Stream sedimentation can be effectively prevented

by the retention of vegetated stream buffers (Hordacre *et al.* 1987; Borg *et al.* 1988a; Croke *et al.* 1999; Water and Rivers Commission 2001).

Water quality

The karri forest is confined to areas with annual rainfall exceeding 1000mm. There is a relatively small area where the historical rainfall is less than 1100mm where groundwater is generally saline. While a small rise in stream salinity has been recorded about two years after clearfelling in these areas, it reduced thereafter and always remained below the 200 mg/L TSS, well within the acceptable limits of 500 mg/L TSS (Borg *et al.* 1987). Salinity is not an issue in the karri forest whilst the area is retained and managed as forest.

Water availability

Streamflow in the karri forest has typically been about 5-10 per cent of rainfall. Reducing stand density reduces leaf area, reduces interception, reduces evapotranspiration and increases streamflow (Stoneman *et al.* 1989c).

Paired catchment experiments conducted in the 1970s and 1980s in the karri forest showed that water yield doubled at two years after clearfelling and regeneration and then declined to pre-harvest levels after ten to fifteen years when the regrowth had reached a density of about 20-25 m²/ha (Borg *et al.* 1987; Bari *et al.* 1994; Bari *et al.* 2003). Water yield beyond 10 years has not been monitored.

Groundwater recharge followed the clearfelling, raising the groundwater level by up to 6m and increasing the groundwater discharge area. Base flow (subsurface and deep groundwater flow) was responsible for two thirds of the streamflow increase (Bari *et al.* 1994). Groundwater discharge resulted in a small increase in salinity, which subsequently declined as the forest cover was restored. Groundwater began to decline after about five years as the vegetation density increased and continued to decline. Groundwater levels continued to fall but the contribution of declining rainfall during this period remains unknown.

While clearfelling can be expected to cause an increase in the recharge of groundwater, the increase in streamflow may take longer to occur, depending on how long it takes for the groundwater to rise sufficiently to intercept the surface (Kinal *et al.* 2012). The response period may also be shorter.

Research examining the recovery of vegetation following timber harvesting found canopy cover reached pre harvest levels in 5 - 10 years, and canopy cover of regrowth forest exceeded that of mature forest by about 15 per cent for at least the next 70 years (Stoneman *et al.* 1989a). It is anticipated that unthinned karri regrowth will use more water than mature forest after about 10-15 years, but studies comparing water use of regrowth and mature forest have not been undertaken. Sapwood area has been shown to be a key determinant in transpiration by trees. A stand of small trees has a larger sapwood area than a stand of larger trees at the same stand density (Macfarlane *et al.* 2010). The relative contribution of the regrowth and the understorey to evapotranspiration is not known.

Reduction in forest density to increase water yield for domestic consumption has been undertaken with successful results in two catchments supplying Manjimup, on two occasions in mixed karri/jarra forest (Bradshaw 2010). It has resulted in an increase in annual streamflow and an increase in the period of flow, but subsequently declined as regeneration

and coppice developed. The extent to which the decline is due to coppice and regrowth development or to a disconnection of the groundwater in response to a long period of reduced rainfall is not known. A more enduring/consistent program of vegetation density management is required to maximise the efficiency of this approach and 'even out' streamflow over time.

The use of stream buffers to reduce the risk of sedimentation conflicts with the objective of maintaining stream flow and its associated biota during dry seasons. Evapotranspiration from stream buffers also reduces groundwater recharge and consequently delays or reduces the streamflow response (Bari *et al.* 1993). An evaluation of the effect of harvesting streamside buffers in an agricultural landscape found harvesting could be carried out without having substantial detrimental effects on water quality if best practice management was used (Neary *et al.* 2011). The risk to water quality of harvesting or selectively harvesting stream buffers needs to be trialled and changes made where necessary, as part of an adaptive management approach to maintaining stream flows while not affecting water quality.

4.4.3 Nutrient recycling

The maintenance of soil fertility is an important element of sustainability. Timber harvesting and burning have the potential to adversely affect long term fertility.

The understorey of the karri forest contains a significant proportion of the nutrients in the forest (Hingston *et al.* 1979; Grove *et al.* 1985; O'Connell 1988). Fire has an important role in nutrient cycling by mineralising nutrients such as calcium (Ca), magnesium (Mg), sulphur (S) and phosphorus (P), making them more available for plant uptake. While some S is volatilised by fire, it is replaced in rainfall. Nitrogen (N) is also volatilised, but the regeneration of leguminous species replaces it over time. The frequency and intensity of burning is therefore important in achieving a balance between the input and output of N, particularly on less fertile sites. A 12 - 14 year burning cycle, together with the retention of about 25 per cent of the forest floor litter, was estimated by O'Connell (1989) to achieve a balance between volatilisation and input of N.

The level of nutrients in the bole of a tree is small relative to other parts of the system with bark representing about half of the total in the bole (Hingston *et al.* 1979). One of the most significant elements is N, where a significant proportion may be volatilised by burning and is replaced from the atmosphere and from N fixing legumes which are regenerated by fire. The losses of N from routine timber harvesting and occasional slash burning are much less than losses of N from regular prescribed burning over the rotation (McMurtrie *et al.* 1997).

The burning of thinning slash has the potential to volatilise more N because freshly felled leaves and small branches are being burnt. Loss of N is closely correlated with litter consumed and O'Connell and McCaw (1997) have concluded that much of this N can be retained by burning in spring under conditions that do not burn the lower N-rich component of the litter. A two year delay in burning after thinning will allow time for much of the leaf material in thinning slash to decompose and release its nutrients (O'Connell 1997).

Current prescribed fire regimes and rotation lengths are expected to maintain a long term N balance, but this should be reviewed periodically as more information becomes available and if practices change.

4.4.4 Climate change

The maintenance of 'normal' streamflow is important for sustaining the existing stream biota and streamside and swamp vegetation as well as maintaining water supplies for domestic and industrial consumption. Trends in streamflow in the karri forest over a number of years have been variable (Figure 3) (DOW 2009).

There has been a disconnection between the groundwater and surface water systems in the northern jarrah forest since the year 2000 as a result of reduced rainfall since 1975 (Kinal *et al.* 2012). This disconnection appears to be occurring more slowly in southern forest but it is likely to become evident in the next few years.

If rainfall continues to decline or does not return to past levels, forest density will eventually reach a new equilibrium and streamflow and water table decline should stabilise at lower levels. However this will be a long term process and it is likely that stream biota and streamside vegetation will be severely affected before this occurs. The effect can be reduced by reducing forest density by thinning to maintain a lower forest density.

4.5 Maintenance of forests contribution to the global carbon cycle

The general findings of the Australian Greenhouse Office in respect to managed (multiple use) native forests in Australia has been described in the Australia's State of the Forests Report (Montreal Process Implementation Group for Australia 2013). The report considers carbon in relation to forest management and silviculture, fire, product storage, bio-energy and forest soils. This concludes that managed native forests in Australia sequester more greenhouse gas than they emit. Managed native forests offset 32M tonnes of carbon dioxide, about five per cent of Australia's greenhouse gases (2005 data); and wood products storage offsets a further one per cent. The report also concluded that there is considerable scope for using biomass energy for the production of electricity to offset the emissions created by the use of fossil fuels.

Forest fire causes the release of carbon and other greenhouse gases, but the effect of carbon release is generally considered to be balanced by growth in the long term.

Moroni (2011) considers that rather than simply seeing forests as primarily carbon storage, the greatest contribution of forests towards the reduction in carbon emissions is the substitution of wood products for more energy intensive products. This view is consistent with that of the Intergovernmental Panel on Climate Change and the Climate Commission in Australia (IPCC 2007; Steffen *et al.* 2011).

Within the south-west forests, the approach adopted for the FMP for estimating native forest carbon stocks was influenced by the type of data and information available for the karri forests. Carbon is stored in forest ecosystems in the soil, the below-ground biomass (roots), and the above-ground biomass (comprising trees, understorey, surface litter and coarse woody debris such as fallen trees). An extensive review confirmed there was limited data available to estimate the soil carbon and below-ground biomass components of the total carbon pool, and that data available for the above ground biomass components varied markedly in quality, comprehensiveness and geographic representation.

The scope of the Department's work focussed on estimating that component of the total carbon stock for which reliable, consistent data was available and for which sampling had covered most of the forests – the above-ground component of live standing trees. This generally comprises the largest proportion of the total carbon pool in the forest, and is the component for which quite robust mathematical relationships have been developed to indirectly estimate the below ground components.

Changes to the carbon stock during the period of the FMP will arise from a number of factors, including variations to the extent of forests, varying rates of growth and mortality across the forest, changes initiated by disturbances such as mining and timber harvesting, and natural disturbance events such as forest disease, bushfire and drought. Overall, the quantity of carbon stored in the live trees on karri forest in the FMP area is projected to increase over the period of the FMP (Department of Environment and Conservation 2013).

4.6 Integrated management

A number of factors need to be evaluated and weighed in the development of management procedures appropriate to different forest values and in different parts of the forest. In native forest management, silviculture is seldom applied for the enhancement of a single objective. An understanding of the impact of silvicultural practice on each forest value is important to be able to balance competing goals and values if required. There is also a temporal element in that forest management objectives do not remain static over time, nor does the forest condition remain the same. Maintaining options for the future needs to be a consideration in the development of any silvicultural practice.

In its simplest sense, the practical implementation of silviculture typically involves the protection of key elements (e.g. habitat), the manipulation of growing space and the regeneration of stands. Examples of the impact of retaining different stand density (growing space) in satisfying the requirements of wildlife, wood and water are given below to illustrate the synergies and possible conflicts that can be involved.

Reduced stand density is an advantage for forest health and resilience, water production, and timber production. While very low stand density is most suited to water production, a stand density much below 'critical' density will reduce timber productivity.

Water production is enhanced at lower vegetation/stand density, but low density may reduce water quality if it increases the risk of stream salinity in some parts of the catchment. The risk of water table rise leading to increased salinity has become somewhat less relevant in a drier climate (Kinal *et al.* 2011).

Water production would be most advantaged by the retention of mature trees and the minimisation of regeneration. However, this would disadvantage timber production and would be unsustainable if taken to the extreme.

While wildlife may benefit from the retention of more habitat trees, timber production is disadvantaged due to competition with retained crop trees (Rotheram 1983).

Undisturbed stream buffers minimise sedimentation in disturbed landscapes, but they also reduce groundwater and streamflow, which may adversely impact of stream biodiversity.

Small coupes increase the cost of regeneration burns, reduce the intensity of landscape disturbance, but this results in an increase in the frequency of landscape disturbance.

Thinning to produce bioenergy has the potential to reduce greenhouse gas emissions and is compatible with improving growth rates and forest health, streamflow and groundwater, and can assist in maintaining biodiversity under conditions of a drying climate.

Forest management sometimes requires balancing competing values that forests are able to provide. To provide this balance, we require a clear enunciation of the objectives for each of the values in question and a good understanding of the impacts of silvicultural practices on each.

Appendices

Appendix 1 Changes in silvicultural practice

Silvicultural systems

Classical silvicultural systems are generally divided into systems that are designed to create even-aged stands (clearfelling systems) or uneven-aged stands (selection systems) (Florence 2004). The essential difference between these systems is the size of the gap created in the canopy and the size of the patches of regeneration that result. While often seen as distinctly different, they are more appropriately seen as part of a continuum of canopy gap sizes; with varying extent and intensity of the associated edge effects. Where most of the gap is affected by the edge influence, the stands are considered to be uneven-aged. However since there are many edge influences (Figure 32), there can be no definitive distinction (Bradshaw 1992). Because the mature trees in the karri forest are large, many of the edge effects are also more extensive and the appropriate gap size is larger than that for smaller trees.

Within any of these systems, various sources of regeneration may be employed (seeding, planting, lignotuberous advance growth or coppice) and commercial timber harvesting may be supplemented with the non-commercial removal of unsaleable trees to facilitate regeneration and thinning. Thinning, the process of removing some of the trees to improve the growth or health of the remainder may be undertaken at intervals throughout the life of the stand. Thinning differs from selection cutting in that it does not seek to establish regeneration.

Any of these systems may also be supplemented with various forms of 'legacy tree retention', that is, the retention of individuals or groups of trees or patches of older forest designed to provide older elements of the forest during the period before the regenerating forest again provides these values (Florence 2004; Forestry Tasmania 2009; Baker *et al.* 2011). The elements of diversity in a forest of even-aged and uneven-aged stands may be the same but at different scales or resolution.

The design of the most appropriate silvicultural system depends on a range of factors, including the management objectives, scientific and observational knowledge, ecological and silvicultural characteristics, existing forest structure, markets and economics, fire management issues, and policy and public opinion (Florence 1977). Many of these factors will change over time and so too will the practices considered most appropriate.

A wide variety of silvicultural practices has been employed in the karri forest since the 1880s and are summarised here. Examples of all of these practices still exist and provide invaluable insights into the expected outcome of current practice. The structure that has resulted from those practices in the past is a determining factor in the choice of subsequent silvicultural practice.

Past silvicultural practice

Silvicultural practices employed in the karri forest fall into several distinct periods and are described in detail by Bradshaw (Bradshaw 1999; Bradshaw 2007). The location of earlier timber harvesting has been summarised by Heberle (1997). Past practices are summarised below to provide an insight into the causes of the present range of stand structure.

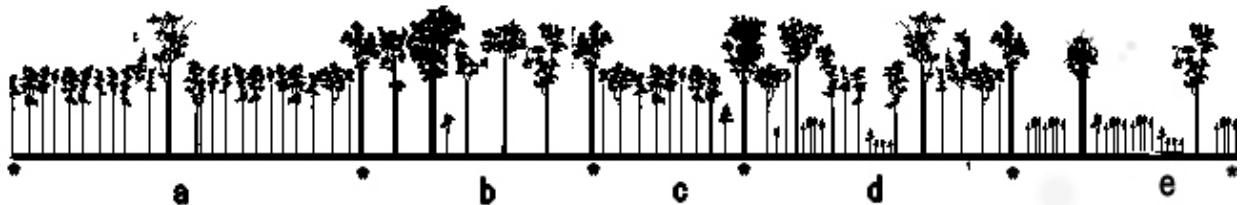


Figure 37: The range of stand structure found in the karri forest.

a) large groups of pole size trees resulting from the removal of large groups of mature trees, with occasional mature trees remaining. b) mature trees in virgin forest of varying size or those trees retained during the previous harvest, some of which were immature at the time. c) smaller groups of pole sized trees resulting from the removal of several mature trees. d) multi-aged stands resulting from the removal of single trees. e) large groups of saplings with occasional large trees remaining. All of these structures may be present in the virgin forest as a result of bushfire or in the cutover forest as a result of timber harvesting.

Harvesting for regeneration

1880 to 1925

This was a period of largely uncontrolled timber harvesting for sawlogs, starting in the coastal areas near Augusta (Karridale) and later at Denmark where there was ready access to shipping. Regeneration followed commercial exploitation, mostly by clearfelling followed by burning and seed fall from trees that were left because they were not economical to harvest at that time. Much of the regeneration was burnt and coppiced after bushfires, perhaps more than once. As a consequence much of the existing regrowth at Karridale (also known as the Boranup forest) appears to be younger than the original regeneration date. The large mills that operated in these areas ceased about the turn of the century and smaller operations then continued for several years. Most of the surviving regrowth in the Karridale area is now in National Park while most of the Denmark regrowth is privately owned.

Following the harvest the Karridale area would have had a structure similar to e) in Figure . Over the last 100 years the structure has progressed through a) with older areas approaching extensive areas of b) (Figure 37), virtually all of the original veteran trees have died of old age and fire damage.

The first major incursion into the main karri belt began with building of large mills at Jardee, Deanmill and Pemberton in 1911, 1912 and 1914 with the coming of the railway to these

areas. All of the karri forest harvested till 1925 was converted to agriculture and majority of the forest cut in that period is permanently cleared.

1925 to 1940

The passing of the Forests Act in 1918 resulted in the dedication of State forest and the start of active forest management. Big Brook was the first area of karri forest to be dedicated and managed as State forest. The system used from 1925 in State forest was nominally a girth limit but in practice was closer to clearfelling but with the retention of cull trees unsuited to sawmilling. Burning in a seed year resulted in wheat-field regeneration after which the cull trees were ringbarked. A concern for the fire hazard associated with the holding of felling debris till the next seed crop led to annual regeneration burning with supplementary seeding. Towards the end of this period there was a trend towards the retention of more immature trees. Most of this are now has structure a) with later areas approaching the c), d) (Figure 37).

1940 to 1967

Concern for the waste of smaller trees not utilised by sawmills; a desire to cover more area to salvage what was seen as diminishing resource in old trees and to open up the forest for fire control; and increasing pressure to alienate clearfelled areas for agriculture resulted in the retention of more immature trees and the adoption of the Australian Group Selection system (Jacobs 1955; Meacham 1962). The gap sizes created were limited to a maximum diameter of 120 m, the limit of seed shed from surrounding trees but in practice the gaps were closer to 30m, about half tree height. Burning with sufficient intensity to burn the debris in the gaps without damaging the retained trees was a major inhibitor to regeneration burning. Annual burning of harvesting debris ceased in the 1950s and thereafter was confined to seed years. In the mid-1960s scrub-rolling was introduced to create a more even fuel for burning, wildlings were used to infill understocked regeneration and karri and marri were culled to prevent the inhibition of regrowth.

These areas are characterised by c), d) in Figure 37 with most regrowth in gaps of half a mature tree height.

1967 to 1975

In 1967 clearfelling was re-introduced (White *et al.* 1974). The reasons for the change were: the difficulties in undertaking successful regeneration burns in selection cut forest; the observation that the crowns of retained trees frequently degraded with exposure, retained trees significantly reduced the growth rate of regeneration in small gaps; the larger gross area of the regeneration program; greater gross area requiring fire protection of the regrowth; and the damage caused by felling large retained trees into the regrowth patches. Although the latter issue was foreshadowed by Ednie-Brown (1896) from his observations at Karridale, the significance was not appreciated until the second cut in selection stands was imminent.

The method used was clearfelling with seed trees. Following timber harvesting the understorey was scrub-rolled and the area burnt when sufficient seed was present in the crowns. The better trees in the stand were used as seed trees and were removed within two years of the regeneration (White 1974a, 1974b). Regeneration surveys in the first summer

monitored the results and open rooted nursery seedlings or wildlings were used to infill plant where necessary.

Harvesting was concentrated in pure karri stands where most trees could be removed for sawmilling. Stands with a high proportion of marri which could not be removed commercially and which if retained would cause severe suppression of regrowth, were avoided.

Stands from this period result in the structure depicted in a) (Figure 37) except that there are no veteran trees remaining.

1975 to 2000

The advent of an export wood-chipping industry provided the opportunity to harvest mixed karri/marri stands by providing a market for marri and the opportunity to remove it to facilitate regeneration. Trials of the regeneration of these types had demonstrated that adequate regeneration stocking of marri could be achieved from lignotubers and coppice without the need for a marri seed source (White 1971). The seed trees system continued using only karri seed trees. The resulting structure is similar to that described in the previous era for both pure karri and karri/marri types.

Clearfelling and planting complemented the seed tree program to provide for a more even annual regeneration program. Planting was initially done with open-rooted seedlings and later with potted stock. Potted stock was raised in jiffy pots to be replaced by copper-coated then moulded root-trainer pots. Limited regeneration was done with broadcast seeding.

Stocking standards were more rigorously monitored from 1990. The planting rate for seedlings was initially 1250spha but increased to 2000spha in 1996 because of concerns about branch size. Optimum stocking was defined as 85 per cent stocked at greater than 3000spha and adequate stocking as 85 per cent stocked at 1666 - 3000spha (CALM 1990; Lutze 2001).

Concerns about soil compaction due to timber harvesting in moist soil conditions led to restrictions on winter (moist soil) operations and from 1990 the ripping of compacted sites (Bradshaw 1978; Breidahl *et al.* 1995).

Seed tree operations require long lead times for planning to integrate seed cycles, nursery programs and multi-phase timber harvesting. As planning uncertainties increased during the 1990s with unresolved issues of reservation it became increasingly difficult to manage seed tree regeneration and the method was last used in 1997.

2000 to 2013

A number of changes in practice have occurred since 2000.

The loss of markets for marri chipwood in 2000 resulted in a change to clearfelling practices. Limited culling of marri was undertaken to reduce competition for regrowth in mixed stands but many stands are 'partially' cut and consequently only partially regenerated with karri.

The reservation of all old-growth forest in 2001 reduced the level of harvest and restricted timber harvesting (for regeneration) to forest that was previously cutover under the group selection system. Coupe size is restricted to less than 40 ha, partly by prescription and partly because of the patchy nature of the forests available for harvest. Coupe size in regrowth that is harvested and regenerated in the future is restricted to 20 ha. Harvesting for regeneration

is restricted to stands of the type c) and d) (Figure 37). Where marri is present many stands will have a structural outcome similar to d) (Figure 37) with a marri overstorey (CALM 2005).

Pressure from the newly established wine industry in the region to limit smoke haze during the period when grapes are maturing has resulted in the restriction of most regeneration burns till after the grapes are harvested in March / April. To enable burning to be done after the first rains it is necessary to heap the debris and burn heaps rather than broadcast burning. The site is usually shallow-rippled between the heaps. This method is also used to safely burn small coupes. This process necessitates the use of planting and the additional application of Phosphate fertiliser between heaps.

2014 to 2023

From the introduction of the FMP (Conservation Commission 2013b)) in 2014, the changes to silvicultural are as follows:

- Exclude some large logs from heaps that are to be burnt in the course of preparation for regeneration to provide Coarse Woody Debris habitat
- Retain some senescent and dead trees in regenerating areas within the limits of safe practice for present and future operations to enhance structural complexity.

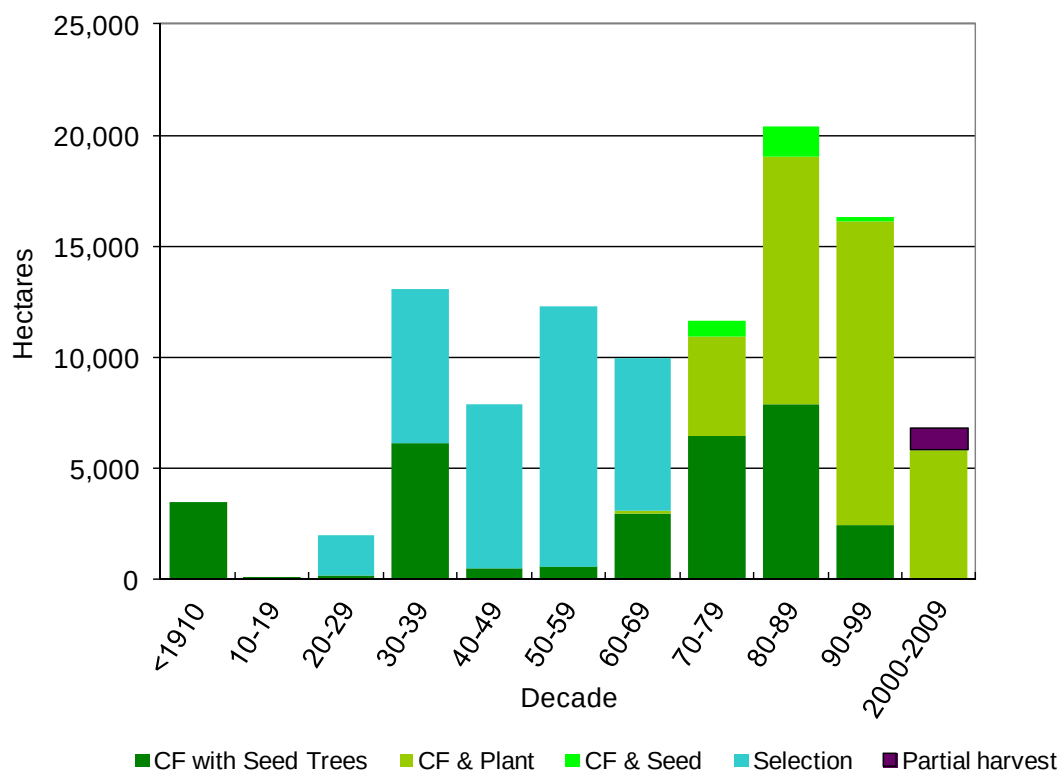


Figure 38: Changes in the method of timber harvesting over time.

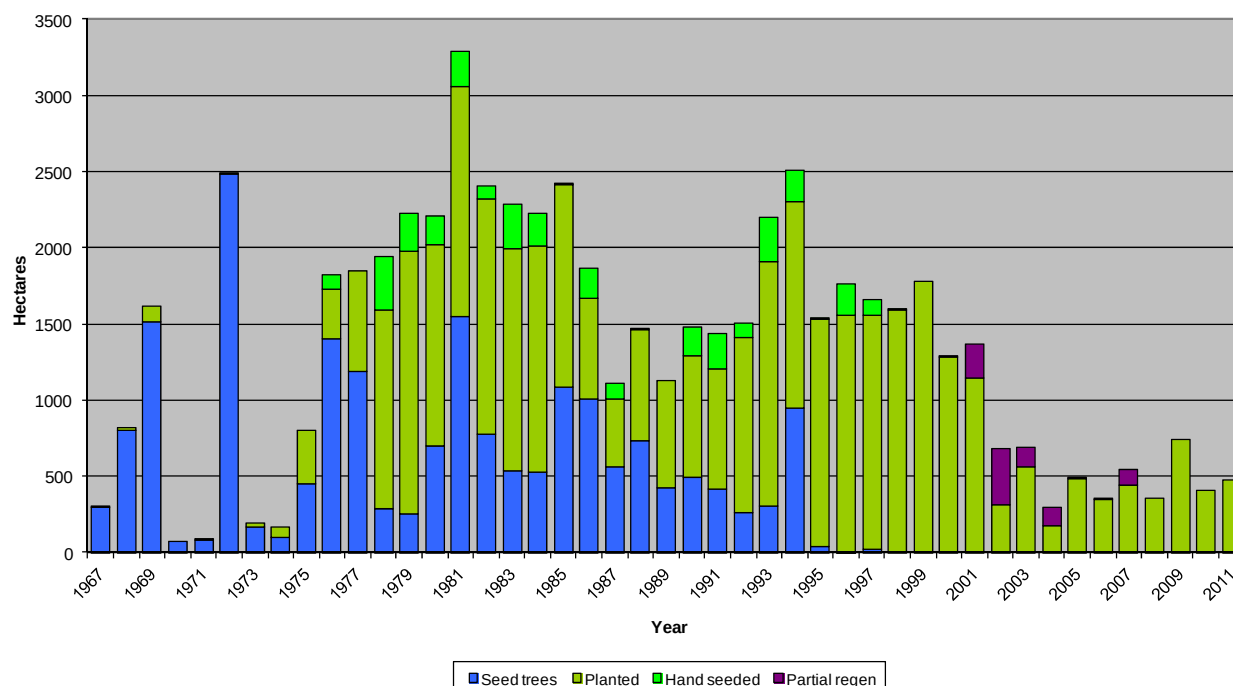


Figure 39: The method of regeneration showing the impact of the karri seed cycle the increasing use of planting and the increase in areas partially regenerated due to the loss of the marri chipwood market.

Thinning

The thinning of even-aged regrowth stands has occurred in parallel with regeneration operations.

1960 to 1980

During this period areas of even-aged regrowth at Karridale were thinned for tile-battens. A small area of Big Brook regrowth (then about 40 years old) was also thinned. However the specifications for tile battens were such that it required thinning from above and the operation in Big Brook was short-lived.

1980 to 1993

The first thinning of regrowth stands established from 1930 to about 1950 was carried out. This was a commercial thinning for chipwood with a small proportion of logs suitable for tile battens and electricity transmission poles.

1996 to 2013

The commercial thinning of even-aged regrowth established since 1967 commenced in 1996. Most of the product is used for chipwood. First thinning is delayed until the stand is 30m in height by which time the maximum 'clean zone' has developed. See Figure 40

Small areas of regrowth that were established in the 1930s have been thinned for the second time.

Proposed Changes from 2014

Changes proposed in the FMP (Conservation Commission 2013b) are as follows:

- Apply variable density thinning to promote structural complexity in large areas of uniform regeneration.
- Undertake thinning or other silvicultural management to protect ecological values (especially aquatic ecosystems) threatened by climate change.

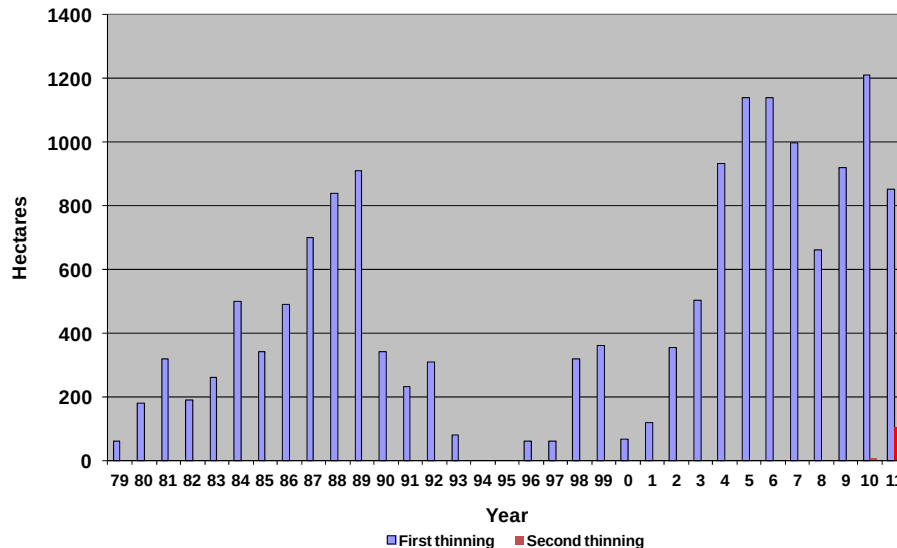


Figure 40: Thinning of even-aged karri regrowth. Stands thinned up to 1993 were mostly established in the 1930s. Later thinnings were in stands established after 1967. Second thinnings are in areas established in the 1930s.

Appendix 2 Vertebrate fauna that may be affected by timber harvesting

The species listed in the following tables occur in karri forest areas which are subject to timber harvesting and are considered to have at least one risk factor or have the potential to be adversely affected by timber harvesting. The risk factors considered are:

- listed on the W.A. Wildlife Conservation Specially Protected Fauna list
- have a restricted distribution of three or less habitat types
- have a habitat factor such as a requirement for tree or log hollows for breeding or have special habitat requirements
- indirect predicted impact of timber harvesting e.g. increases likely in fox predation.

After consideration of these factors, the FDIS report (Christensen *et al*, 2005) only highlighted six species where species specific measures were recommended in addition to those required by the FMP or the karri silvicultural guidelines. These six species are indicated with an asterisk in the following tables. The information in the table was current as of December, 2014 except for the population status of all species, which is as listed on the WA Wildlife Conservation Threatened and Protected Fauna list published on 6 November 2012.

Arboreal marsupials and terrestrial mammals.

Fauna group	Common Name	Scientific Name	Mean Body Mass (g)	Diet	Breeding	Population Status
Arboreal mammals	*brush-tailed phascogale	<i>Phascogale tapoatafa</i>	230 males 160 females	Invertebrate's small birds and small mammals.	Tree hollows	Vulnerable
	common brushtail possum	<i>Trichosurus vulpecula</i>	1 300-4 500 male 1 200-3 500 females	Herbivore and tolerant to poisonous native plants.	Tree hollows	Not listed
	honey possum	<i>Tarsipes rostratus</i>	Up to 12 males Up to 22 females	Nectar and pollen.	Abandoned nests or hollow stems of grass-trees.	Not listed.
	*Western ringtail possum	<i>Pseudocheirus occidentalis</i>	800 -1 130	<i>Agonis flexuosa</i> leaves and, leaves of myrtaceous species.	Tree hollows, ground hollow or balga	Vulnerable
Terrestrial mammals	chuditch	<i>Dasyurus geoffroii</i>	700 - 2000 males 600 - 1120 females	Opportunistic feeders, invertebrates to small mammals and birds	Hollow logs	Vulnerable
	quenda or Southern brown bandicoot	<i>Isodon obesulus</i>	550 - 1 850 males 400 - 1 200 females	Omnivore.		Priority 5

	*quokka	<i>Setonix brachyurus</i>	2 700 - 4 200 males 1 600 - 3 500 females	Herbivore.	Dense understorey or swamp thickets	Vulnerable
	water-rat	<i>Hydromys chrysogaster</i>	400 - 1 275 males 340 - 992 females	Omnivore.		Priority 4
	Western brush wallaby	<i>Macropus irma</i>	7 000 - 9 000	Herbivore.		Priority 4

Bats of the karri forest.

Fauna group	Common Name	Scientific Name	Mean Weight (g)	Foraging	Roosting	Breeding	Population Status
Bats	chocolate wattled bat	<i>Chalinolobus morio</i>	8 - 11	Between understory and canopy.	Hollows, caves, and logs.	They give birth in October	Not listed
	Gould's long-eared bat	<i>Nyctophilus gouldi</i>	9 - 13	Will capture insects on the wing but will also drop onto them from a resting place.	Tree hollow, peeling bark.	Young fly in January.	Not listed
	Gould's wattled bat	<i>Chalinolobus gouldii</i>	8 - 18	Above tree canopy and in open areas within 1 metre of the ground.	Trees with hollows, diameter 10cm.	Mating in autumn and winter, sperm is stored until fertilisation in spring and young are born late spring early summer.	Not listed
	greater long-eared bat	<i>Nyctophilus timoriensis</i>	11 - 20	Amongst understorey stratum.	Tree crevices, foliage and under bark.	Reproductive biology has not been studied	Priority 4

Fauna group	Common Name	Scientific Name	Mean Weight (g)	Foraging	Roosting	Breeding	Population Status
	lesser long-eared bat	<i>Nyctophilus geoffreyi</i>	6 - 12	Close to the ground as close as 1 metre.	Dead trees, hollows and under bark.	Mating in autumn sperm stored and fertilisation in spring young born October to November.	Not listed
	Southern forest bat	<i>Vespadelus regulus</i>	3.6 - 7	Between the canopy and top of understory.	Tree hollows.	Mating occurs year round insemination autumn, fertilisation spring and young are born in summer.	Not listed
	Southern freetail-bat	<i>Mormopterus planiceps</i>	8 - 10.5	Above the canopy uncluttered habitat.	Tree hollows.	Young born early summer	Not listed
	Western false pipistrelle	<i>Falsistrellus mckenziei</i>	17 - 26	Between crown and understory canopy.	Tree hollows, or ground logs on ground where tree hollows not available.	Reproductive biology has not been studied.	Priority 4
	white-striped freetail-bat	<i>Tadarida australis</i>	32 - 48	Above the canopy, fast and direct.	Large trees live or dead with hollows.	Mating in late winter and young are born mid-December to late January.	Not listed

Hollow nesting birds.

Fauna group	Common name	Scientific name	Foraging	Size of hollow used	Population status
Canopy birds	Baudin's cockatoo	<i>Calyptorhynchus baudinii</i>	Eucalyptus fruit and flowers while occasionally feeding on insect larva.	Large	Endangered
	forest red-tailed black-cockatoo	<i>Calyptorhynchus banksii naso</i>	Feeds on fruit, seeds and flowers e.g. Eucalyptus and Banksia, occasionally feeds on insect larva in trees.	Large	Vulnerable
	Port Lincoln ringneck	<i>Barnardius zonarius</i>	Feeding on seeds nuts, fruit, flowers, nectar and insects, mainly on the ground but when food is abundant they feed in the trees.	Medium	Not listed
	red-capped parrot	<i>Purpureicephalus spurius</i>	Seeds from Eucalyptus trees as well as grasses and other understory species	Small	Not listed
	striated pardalote	<i>Pardalotus striatus</i>	Feed in the foliage in the canopy on insects and the insect larva.	Small	Not listed
Second-storey birds	Australian owl- nightjar	<i>Aegotheles cristatus</i>	Predominantly feed on insects either on the wing or by pouncing upon they prey from their current position.	Medium	Not listed
	barking owl	<i>Ninox connivens</i>	Feeds on small to medium sized mammals, birds, reptiles and insects and their diet varies depending on their breeding cycle.	Large	Priority 2

Fauna group	Common name	Scientific name	Foraging	Size of hollow used	Population status
	barn owl	<i>Tyto alba</i>	Feeds on small marsupials, rodents, birds and reptiles. House mouse is a common food source and as a result the owls can be found around human habitation which attracts mice.	Large	Not listed
	masked owl	<i>Tyto novaehollandiae novaehollandiae</i>	Feed mainly on small mammals, such as rodents, rabbits and bandicoots. Other prey animals include possums, reptiles, birds and insects, with hunting taking place in the early hours of night.	Large	Priority 3
	rufous treecreeper	<i>Climacteris rufa</i>	Insects, ants, snails, small reptiles and seeds.	Small	Not listed
	sacred kingfisher	<i>Halcyon sancta</i>	Mainly feed on land occasionally capturing prey on the water feeding on reptiles, insects and larva.	Small	Not listed
	Southern boobook	<i>Ninox novaeseelandiae</i>	Insects, small mammals, and reptiles make up the majority of the owls diet. Some prey is taken on the wing early morning or dusk, occasionally during the day if it is over cast and dull.	Large	Not listed
	Western rosella	<i>Platycercus icterotis</i>	Seeds, grass, fruit and flowers they feed on the ground in open areas or in trees.	Small	Not listed
Non-forest birds	elegant parrot	<i>Neophema elegans</i>	Seeds from grasses and herbaceous plants as well as fruit from fruit trees.	Small	Not listed

Fauna group	Common name	Scientific name	Foraging	Size of hollow used	Population status
	tree martin	<i>Cecropis nigricans</i>	Insects from open areas. Generally captures the insects on the wing.	Small	Not listed
	Australian kestrel	<i>Falco cenchroides</i>	Small mammals, birds, reptiles and insects make up the food source which is captured by hovering a short distance above the ground and dropping onto its prey.	Medium	Not listed
Aerial hawking birds	dusky woodswallow	<i>Artamus cyanopterus</i>	Insects and nectar from flowers.	Small	Not listed
	peregrine falcon	<i>Falco peregrinus</i>	Birds, rabbits, marsupials as well as other day active mammals.	Medium (occasionally)	Specially protected
Seasonal birds	purple-crowned lorikeet	<i>Glossopsitta porphyrocephala</i>	Lives solely in trees in which it feeds on the fruit.	Small	Not listed

Birds (not hollow using)

Fauna group	Common name	Scientific name	Foraging	Nesting	Population status
Non-hollow nesting birds	crested shrike-tit	<i>Falcunculus frontatus</i>	Feeds on insects in the canopy of trees. It uses its strong beak to lever off bark to get at insects. Prefers open wandoo woodlands or karri which has been burnt or cleared. There is variation between the foraging of the males and females of the same species with the females foraging on branches while the males forage on the trunks of the tree.	Nest in the fork of trees made from spider webs and fine twigs.	Priority 4
	*malleefowl	<i>Leipoa ocellata</i>	Opportunistic feeders on what food source is abundant at the time.	Builds a mound nest and they bury the eggs.	Vulnerable

Reptiles and amphibians

Fauna group	Common name	Scientific name	Activity pattern	Foraging	Common shelter site	Breeding	Population status
Reptiles	South-western mulch skink	<i>Glaphyromorphus "koontoolasi"</i>	Unknown.	Unknown.	Unknown.	Unknown.	Priority 1
	jewelled South-west ctenotus	<i>Ctenotus gemmula</i>	Unknown.	Unknown.	Burrows at the base of tussocks, understoreys or beneath leaf litter.	Unknown.	Priority 3
Amphibians	*orange-bellied frog	<i>Geocrinia vitellina</i>	Calls only at night although active both day and night.	Invertebrates.	Permanently moist sites in lateritic uplands and narrow valleys.	Lays eggs in shallow chamber.	Vulnerable
	*white-bellied frog	<i>Geocrinia alba</i>	Calls only at night although active both day and night.	Invertebrates.	Persists along creek lines within forested and agricultural landscape.	Eggs deposited in shallow depressions.	Critical

Fish

Fauna group	Common name	Scientific name	Life cycle	Size (mm)	Foraging	Breeding	Population status
Fish	Balston's pygmy perch	<i>Nannatherina balstoni</i>	Live for one year and die after spawning, breeds middle of winter in response to flooding.	90	Prefer shallow dark water with vegetation, insects and larvae primary food source.	500-1600 eggs laid.	Vulnerable
	black-stripe minnow	<i>Galaxiella nigrostriata</i>	Breeds following winter rains.	44	Found in slow running tannin stained water, even roadside ditches. Feeds on insects and larvae.	Eggs laid in vegetation.	Priority 3
	mud minnow	<i>Galaxiella munda</i>	Survives for 1 year and breeds in winter spring period.	Maximum of 58	In areas where water is fast flowing with submerged vegetation. Feeds on insects and larvae.	Eggs laid in flooded vegetation.	Vulnerable

	Pouched Lamprey	<i>Geotria australis</i>	Lampreys have several distinct stages in their life cycle. Pouched lampreys larvae remain in streams for up to 4 years. After a metamorphosis, the juveniles then migrate to the sea where they become parasitic feeders. After 1 - 2 years at sea the adult lampreys return to the streams to spawn.	500 - 700	The larvae are filter feeders, feeding on algae, detritus and micro-organisms. Adults during their marine stage are parasitic on marine fish. Returning adults do not feed when they enter freshwater.	After 1-2 years at sea the adult lampreys return to the streams to spawn. They require waters with sand, gravel or pebble substrates for spawning. They die after spawning.	Priority 1
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Glossary

Basal area	The sum of the cross-sectional areas of trees in a given stand measured at 1.3m above the ground. It is usually expressed as square metres per hectare.
Bole	The tree trunk from the ground to the crown break. The bole does not include the major branches supporting the crown.
Breast height	A standard point of measurement at 1.3m above ground level.
Cambium	A layer of living, meristematic cells between the wood and the innermost bark of a tree. Division of these cells adds a new layer of cells on the wood already formed as well as a layer of inner bark on the outer surface of the cambium.
Canopy	The uppermost layer in a forest, formed by the crowns of the trees.
Clearfell	A silvicultural method in which all, or nearly all, trees in a defined area are removed at one time to allow regeneration to establish and develop (note legacy elements are marked for retention, and some non-commercial trees may still remain on site).
Coarse woody debris	Dead woody material such as boles and branches on the ground or in streams.
Cohort	Trees of the same age.
Coppice (noun)	A shoot (or shoots) arising from adventitious buds at the base of a woody plant that has been cut near the ground or burnt back.
Coppice (verb)	The act of cutting near the ground or burning back a woody plant to encourage a shoot (or shoots) to arise from dormant buds at the base of the plant. Often completed to encourage the development of a new vigorous coppice stem. Sometimes referred to as mullenising.
Crop tree	A tree selected to retain during a harvest operation, to be grown on for many years to become a component of a future commercial harvest
Crown cover	The area covered by the crowns of trees, assuming they are opaque and ignoring overlap. The parameter measured by air photo interpretation.
Cull	A tree that is not required in a stand but which has no saleable value.

Culling	The reduction in the density of unwanted vegetation, usually to reduce competition to retained crop trees or for establishing or releasing regeneration.
Current annual increment (CAI).	The increment measured over one year.
Dbhob	Stem diameter measured at breast height over bark.
Density - critical	The stand density at which competition between trees begins to occur and growth of stand volume or basal area begins to slow. For consistency it is often defined as 90 per cent of optimum density.
Density - optimum	The stand density at which maximum stand volume or basal area growth occurs.
Density - maximum	The maximum density that can be achieved by a stand of a particular age and site before competition-induced mortality occurs.
Density - suppression	The density at which growth in stand volume or basal area growth begins to slow as it approaches maximum density.
Diverse ecotype zone (DEZ).	Open jarrah forest (less than 30 per cent canopy cover), flats, sedgeland, rock outcrops and swamps excluded from timber harvesting.
Epicormic	A shoot that develops from dormant buds on the bole or in the crown when the original crown is removed or damaged. They may be suppressed and fall off if the original crown recovers quickly.
Evapotranspiration	Loss of water from an area of land through the transpiration of plants and evaporation from the soil.
Forest ecosystem	An indigenous ecosystem with an overstorey of trees of more than 20 per cent crown cover. These ecosystems should normally be discriminated at a resolution requiring a map-standard scale of 1:100,000. Preferably these units should be defined in terms of floristic composition in combination with substrate and position within the landscape.
Gap	A discrete opening in the overstorey canopy that reduces competition and allows seedlings to become established and/or develop.

Group selection	The removal or retention of trees in relatively small groups with the object of creating a gap or retaining a group of younger trees to grow on. While there is no specific size of the group, the size of the gap must be large enough to create a suitable microclimate for regeneration and/or growth of younger trees, and allow for later felling without causing undue damage to surrounding trees.
Habitat tree	A tree selected to be retained in a coupe because it has features attractive to wildlife, particularly for hollow nesting birds and animals.
Legacy tree (also referred to as Overwood)	Trees (usually mature to senescent, including habitat trees) retained from the previous stand within a regrowth.
Legacy elements Also Legacy habitat elements	Refers to existing key habitat features, such as hollow bearing trees and logs which may take many decades to replace and which are retained after harvesting or remain after natural disturbance, which provide refugia and enrich the structural complexity of the new stand.
Lignotuber	A woody swelling formed at the base of some eucalypts that has the ability to produce new shoots when the existing ones are destroyed.
Mean annual increment (MAI).	The average annual increment (growth) up to the stated age.
Old-growth forest	Ecologically mature forest where the effects of unnatural disturbance are now negligible. The definition focuses on forest in which the upper stratum or overstorey is in a late mature to senescent growth stage.
Patch	A group of trees resulting from a natural regeneration event or a past forest management activity such as gap creation and regeneration. May also refer to a particular, relatively small area of forest and/or other vegetation type(s).
Periodic annual increment (PAI or PMAI)	The average annual increment (growth rate) measured over a short period of time.
Rotation	The period between regeneration establishment and the final harvest.
Rotation - Physiological	The age to which most of the trees can live in the absence of a cataclysmic event (about 250 years in the case of karri).

Salinity	The mobilisation of stored salt to the surface soil groundwater leading to increased salt concentrations in water courses.
Sawlog	A log of a size and quality suitable for conversion into sawn timber.
Scrub-rolling	The knocking down of understorey with a bulldozer to provide access for fallers or to provide a more even distribution of dry fuel to facilitate the regeneration burn and seed bed preparation.
SDI (Soil Dryness Index).	An index of soil dryness on a scale (in WA) of 0 (field capacity) to 2000 (bone dry), calculated from daily records of temperature and rainfall. It is also related to the dryness of deep forest litter, logs and some living vegetation which influences the difficulty of fire suppression and mop-up and the extent of damage to trees by fire.
Second-storey	The structural layer between the shrub and herb storey and the overstorey (canopy). In the karri forest, this layer may include species such as <i>Agonis flexuosa</i> , <i>Allocasuarina decussata</i> , <i>Chorilaena quercifolia</i> , <i>Banksia grandis</i> , <i>Xylomelum occidentale</i> , and <i>Persoonia longifolia</i> .
Site Index (SI).	An index of site productivity based on the top height of the trees in a stand at age 50 years. The site index can be derived at other ages where the relationship between height and age is known.
Site potential.	The density of forest that can be supported before it becomes limited by nutrients and moisture. Site potential depends on site conditions such as climate, slope, landform, soils and geology. Site potential is also influenced by stand age in that young stands reach maximum density, and undergo suppression and mortality at a lower density than older stands. Young stands exploit a smaller volume of soil and access less soil moisture than older stands.
Site quality	Site quality is measure of relative productive capacity of a site for a particular species. It can be expressed as volume production or as a class.
Silvics	The study of the life history and general characteristics of forest trees and stands with particular reference to environmental factors, as a basis for the practice of silviculture.
Silviculture	The theory and practice (silvicultural practices) of managing the establishment, composition, health, quality and growth of forests and woodlands to achieve specified management objectives.
Snigging	The process of moving a log from the stump to the bush landing.

Stand	A group of trees or patch of forest that can be distinguished from other groups on the basis of size, age, species composition, and structural condition or other attribute.
Structure	When applied to a forest, is the horizontal and vertical distribution of the alive and dead vegetation.
Stocking rate	The proportion of an area, or the proportion of a sample, where the density of trees is equal to or exceeds a specified density standard e.g. 70 per cent stocking at greater than 1000 stems/ha.
Stream reserve	Areas of forest including watercourses and riparian vegetation that has been set aside as an informal reserve to provide forest undisturbed by timber harvesting; and to protect water quality, riparian vegetation and aesthetic values.
Suppression	The process whereby a tree or other vegetation loses vigour and may die when growing space is not sufficient to provide photosynthate or moisture to support adequate growth.
Thinning	A felling made to reduce the density of trees within a stand. Usually undertaken to improve the growth of trees that remain by reducing competition, without either permanently breaking the canopy or encouraging regeneration. May also be undertaken to enhance forest health, water production or achieve another objective.
Top height	For karri, the height of the tallest 25 stems per estimated by the mean of the two tallest trees within a radius of 16m. Top height is relatively unaffected by stand density.
Virgin forest	Forest that has never been harvested or cleared. It may be of any age.

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