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Reference material for jarrah forest silviculture

F.J. 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 was prepared by 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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Cover image: Immature jarrah forest, courtesy F.J. Bradshaw

Department of
Parks and Wildlife

Reference material for jarrah forest silviculture

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1.1	11 March 2015	Danielle Wiseman	Updated definitions of old-growth forest and update of information relating to climate change and modelling.
1.2	9 April 2015	Danielle Wiseman	Figure 15 re-inserted, Figure 18 updated, corrected section 4.1.5 to reference goal of FMP, Additional headings and captions added to figure 26, additional clarification of what was modelled with 3PG in section 4.2.5 para 6, Section 4.3.2 reviewed by Roger Armstrong and some changes made, Section 4.3.3 updated to reflect Vulnerability of Forests in the south west to timber harvesting report, Section 4.5 changed to reflect fact sheet Estimating native forest carbon stocks, glossary terms amended to ensure consistent with FMP and silvics guidelines, duplicate references removed.

1.3	2 June 2015	Danielle Wiseman	Minor edits to version 1.3 of the document are accepted, the document underwent a final editorial check to update relevant dates and remove draft watermarks; the document number was inserted in the title page and header, checked that there are colons after Figure X: Table Y:, footers corrected to have the same font throughout, checked page numbering, all headings/sub-headings being kept with following text, check that passages of text/the glossary match (are a copy & paste of) the DFMP (eg, App 14 thereof)/FMP where the text/terms are common to both, Updated ToC

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Appendices

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Introduction

The jarrah (*Eucalyptus marginata*) forest of Western Australia forms the largest medium or tall forest type in the south-west of Western Australia. Most of the forest is on public land and is important for its conservation values, water and wood production, recreation and for bauxite mining. Most of the public forest is vested in the Conservation Commission of Western Australia and managed by the Department of Parks and Wildlife, hereafter referred to as the department. Over half of the forest is reserved for conservation purposes and the remaining forest is managed for multiple uses.

Management of the multiple use forest is guided by the Forest Management Plan 2014-2023 (FMP). Subsidiary guidance documents provide the next level of detail for the implementation of the FMP. Included are silviculture guidelines which provide for the maintenance of forest stand characteristics and forest regeneration associated with timber harvesting operations.

This document has been prepared to provide a summary of the scientific and observational information that underpins the jarrah silvicultural guideline and related documents. It is intended to provide reference material for policy makers, foresters and interested members of the public. This document is not intended to provide detailed prescriptions, but rather to assist in understanding why particular practices may be employed. Silviculture guidelines and related procedures and manuals are provided in separate documents.

Silvicultural practice in WA has been developed from specific and more general scientific studies, observation of the results of natural or deliberate events, and on an 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 government policy.

Over the past 130 years, a wide variety of practices have been used in the forests of the south-west of Western Australia. For the last 80 years, practices have been generally well documented. This period 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 jarrah forest, information relating to silvicultural practice applies only to the multiple use forest. The discussion relates both to the jarrah forest type and to jarrah as a species.

To avoid excessive referencing in the text, secondary publications are quoted if they provide a suitable summary or an extensive reference list for a particular topic.

Silvicultural practices used in the rehabilitation of bauxite (and other) mining areas are not covered in this document.

1 The jarrah forest

1.1 Forest types and communities

The dry sclerophyll jarrah forest occurs throughout the south-west forest region of Western Australia within a wide range of climate and soil types (Figure 1). It occurs in mixtures with marri (*Corymbia calophylla*) throughout its range and with wandoo (*E. wandoo*) and powderbark wandoo (*E. accedens*) on the drier eastern edge; with blackbutt (*E. patens*) in moister sites; and with karri (*E. diversicolor*) and yellow tingle (*E. guilfoylei*) in the cooler southern forests. It may occur in intimate mixture with these species or as a mosaic of different forest types.

The area of jarrah forest at the time of European settlement is estimated to have been 2.8 million hectares, although the species occurred over an area double that size. Sixty five per cent of the original forest area remains, with approximately 1.6 million hectares on public land (RFA 1998a). Several maps of the distribution of the jarrah forest have been prepared since that of Surveyor General Fraser in 1882 (Abbott *et al.* 1986). These maps differ in their definition, varying from the area within which the species occurs to the area where jarrah is the dominant forest type. Comprehensive and detailed aerial photograph interpretation (API) maps were produced by the Forests Department in the 1950s and 1960s and covered all land tenures within an area from Mundaring to Albany (excluding the coastal plain), and included virtually all of the then-existing jarrah forest (Bradshaw *et al.* 1997a).

The jarrah forest vegetation has been described in several different ways and levels of detail. The API mapping described forest type (the dominant and secondary canopy species), canopy density, structure and mature height at a resolution of two hectares. A simplified version of this was published as “Forest associations and height of tallest native vegetation” (Bradshaw *et al.* 1997a Map 1). Smith (1974) amalgamated the API data to produce maps of structural formations (after Specht (1970). This was part of a larger series of maps of vegetation formations and associations for the south-west (Beard 1972-80). This series has since been updated (Shepherd *et al.* 2001).

The relative uniformity of the tree species composition belies the wide variation that exists in the understorey vegetation of the jarrah forest over a wide range of landforms and climate. Mapping of the jarrah forest with an emphasis on the understorey composition has been undertaken at several levels.

Systems of site classification based mainly on understorey species associations (site types) have been developed separately for the northern jarrah forest (Havel 1975b, 1975a), the southern jarrah forest (Strelein 1988) and the Blackwood Plateau (McCutcheon 1980). Because of the detailed vegetation survey required for mapping, their application has been confined to project areas.

However, these classifications have been used as the basis for the vegetation complex classification, which has been mapped at 1:250,000 scale for most of the jarrah forest except for the Swan Coastal Plain. Vegetation complexes are primarily based on the attribution of site types 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 identified in the field with any degree of precision.

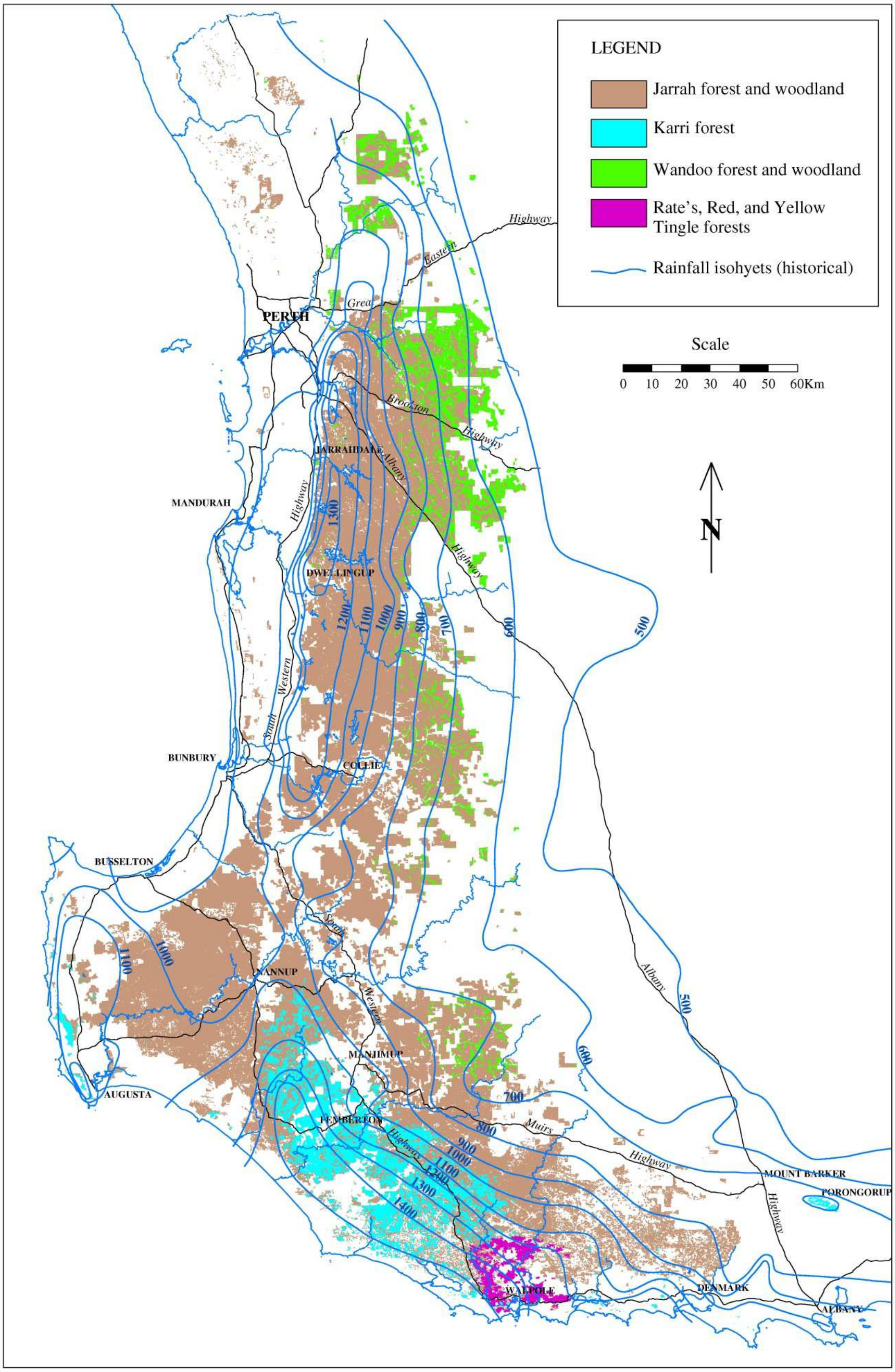


*Figure 1: Distribution of major forest types on public land in the south-west. Isohyets are based on data to 1979 (Hayes et al. 1981)***.**

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 (Heddle *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 312 vegetation complexes recognised within the RFA area.

Vegetation Complexes have been amalgamated to create 120 ecological vegetation systems (EVS), 70 of which apply to jarrah or marri forest (Havel *et al.* 1999).

Vegetation complexes are also used as the basis for the mapping (Mattiske *et al.* 2002) of 30 Landscape Management Units (LMUs) (Conservation Commission 2013a).

Christensen *et al*. (2005) used groupings of vegetation complexes to define fauna habitat types. Thirty-three fauna habitat types were recognised in the jarrah forest of the 54 types identified across the south-west.

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, 12 of which relate to jarrah forest (Bradshaw *et al.* 1997b; RFA 1998b Map 12). Additional information gathered during the previous FMP (2004-2013) supported the recognition of a further forest ecosystem, the Whicher Scarp, which was added in 2013 (Keighery *et al.* 2008).

1.2 Climate

The south-west of Western Australia has a Mediterranean climate, and the occurrence of jarrah and marri is influenced by both annual and seasonal rainfall.

The jarrah forest occurs over a rainfall zone of 600-1300mm / annum with summer evaporation ranging from 400-800mm / annum.

Jarrah (as a species) occurs in areas with rainfall as low as 375mm / annum. It is limited on the eastern fringe by annual rainfall and on the northern boundary by summer rainfall. Marri is limited to an annual rainfall of 450mm / annum but is more tolerant of low summer rainfall and extends 90km further north than jarrah (Churchill 1968; Crombie *et al.* 1988). Jarrah occurs as a forest when the median annual rainfall exceeds 600mm and summer evaporation is less than 750mm (Gentilli 1989) (Figure 2).

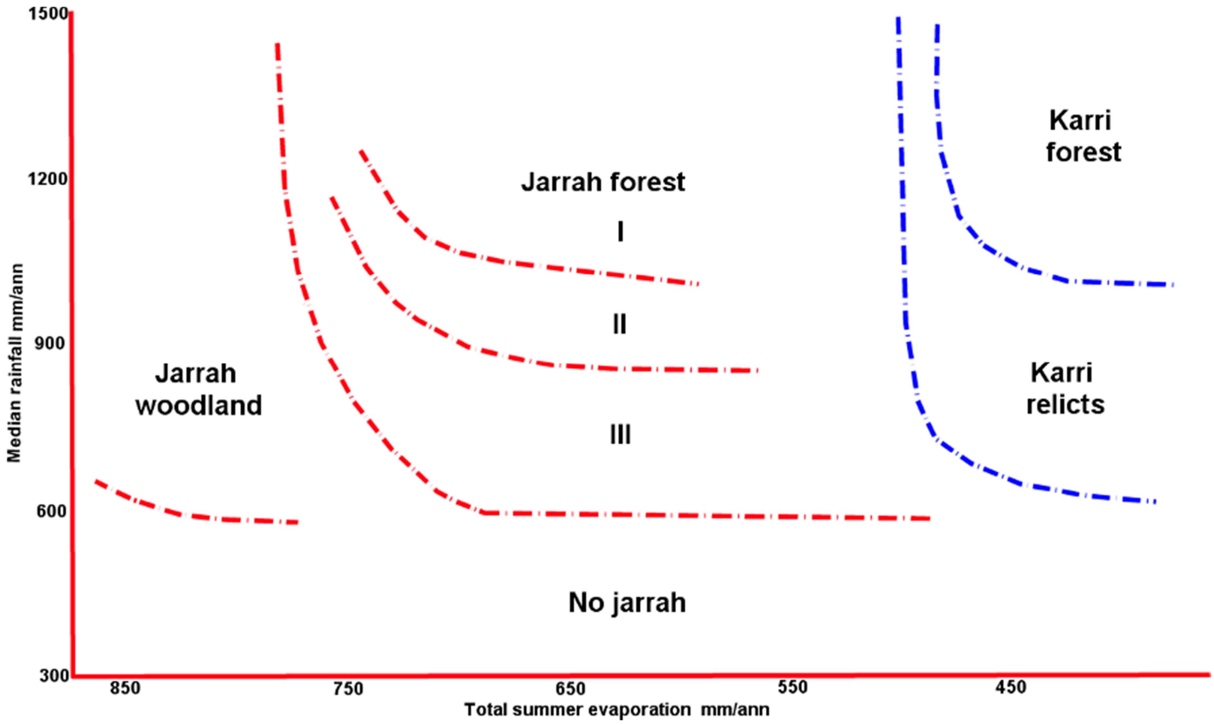


Figure 2: The distribution of jarrah in relation to median rainfall and summer evaporation. Derived from Gentilli (1989). Based on rainfall data to 1975.

Churchill (1968) suggests that the distribution of jarrah has both expanded and contracted over the past 8,000 years, the last contraction beginning in about 1,400 AD. He attributes these changes to changing rainfall rather than fire. Marri appears to be more responsive to changes in rainfall. 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 over that period.

There is a strong west to east and south-west to north-east gradient of reducing rainfall and increasing summer evaporation that is reflected in reduced height, total density and productivity of the forest from about the 800mm rainfall isohyet. However, the 800-1000mm rainfall zone also coincides with a change in landform and associated soil types, described below. The relative effects of soils and rainfall on productivity is difficult to determine.

1.3 Hydrology

The hydrology of the jarrah forest is highly variable. Evapotranspiration ranges from 68-100 per cent of rainfall with streamflow from 0-32 per cent. Forested catchments with a rainfall of 700 mm or less can be expected to have a negligible streamflow while a substantial proportion of catchments within the highest rainfall areas have a streamflow of less than five per cent of rainfall. There are few perennial first and second order streams.

Various estimates of the components of evapotranspiration have been made in different parts of the jarrah forest. These range from: interception (10-21 per cent of rainfall); transpiration from trees (18-50 per cent), second-storey (16 per cent); and the ground layer, including soil (37 per cent) (Schofield *et al.* 1989; Bari *et al.* 2003; Grigg *et al.* 2008).

In high rainfall areas (>1,100 mm / annum), up to 90 per cent of streamflow is from shallow throughflow with a negligible contribution from overland flow. Groundwater discharge contributes to winter streamflow. Groundwater recharge has been measured at between one and nine per cent of rainfall in forested catchments depending on position in the landscape. Forested low rainfall areas (<900mm / annum) may have a streamflow as low as one per cent of rainfall, with no contribution from groundwater (Schofield *et al.* 1989). These observations are in general based on long term rainfall data prior to the trend towards reduced rainfall that has occurred since the mid-1970s.

A reduction in rainfall of 23 per cent since 1975 (as recorded at Jarrahdale) has resulted in a 50 per cent reduction in streamflow of catchments supplying the Perth metropolitan area (Water Corporation 2005). A more severe reduction since 2000 has been attributed to a disconnection between groundwater and surface systems. The ratio of streamflow to rainfall has reduced significantly since that time (Kinal *et al.* 2012).

Hydrological aspects of the jarrah forest are described in detail in Schofield *et al*. (1989) and (Bari *et al.* 2003).

The strong west to east trend of decreasing rainfall is accompanied by a decreasing atmospheric deposition of salt (Schofield *et al.* 1989). Despite the reduced salt deposition, the almost complete evapotranspiration of rainfall in inland areas results in high soil salinity and saline groundwater. Salinity levels are low in higher rainfall areas due to the regular groundwater discharge of deposited salt into the streams. However, soil salinity is spatially variable and some moderate salinity sub-catchments do occur in higher rainfall areas (Johnson *et al.* 1980; Stokes *et al.* 1980). Stream salinity is influenced by groundwater salinity and the relative discharge of groundwater and throughflow. In forested inland catchments, the deep water table and low streamflow results in low total salt discharge, but permanent clearing of vegetation in these areas will result in high salt discharge into streams. Salt storage and groundwater salinity are related to historical rainfall patterns rather than current rainfall patterns.

1.4 Geology and soils

The soils of the jarrah forest are primarily determined by the degree of dissection of the Darling Plateau. The northern forest is dominated by lateritic uplands with soils consisting of sandy gravels overlaying the lateritic duricrust, with varying depths of laterite. The soils of the deeply incised valleys consist of younger red and yellow earths with some gravel colluvium. On the eastern side of the forest, more of the lateritic duricrust has been stripped and the landform changes to one dominated by broad valleys containing yellow duplex soils or orange earths (Churchwood *et al.* 1980; Churchwood *et al.* 1989).

In the southern part of the forest, the lateritic duricrust decreases in elevation and has been more extensively eroded. Jarrah occurs on the lateritic ridges with increasing marri occurrence on the red and yellow podsols on the slopes, giving way to karri on the younger soils where erosion has been more extensive (McArthur *et al.* 1975; Bradshaw *et al.* 1981). At a landscape scale, the influence of soil type affects the proportion of jarrah to marri with a higher proportion of marri found in southern jarrah forests (e.g. Deanmill) as opposed to northern jarrah forests (e.g. Harvey / Collie)(Figure 3).

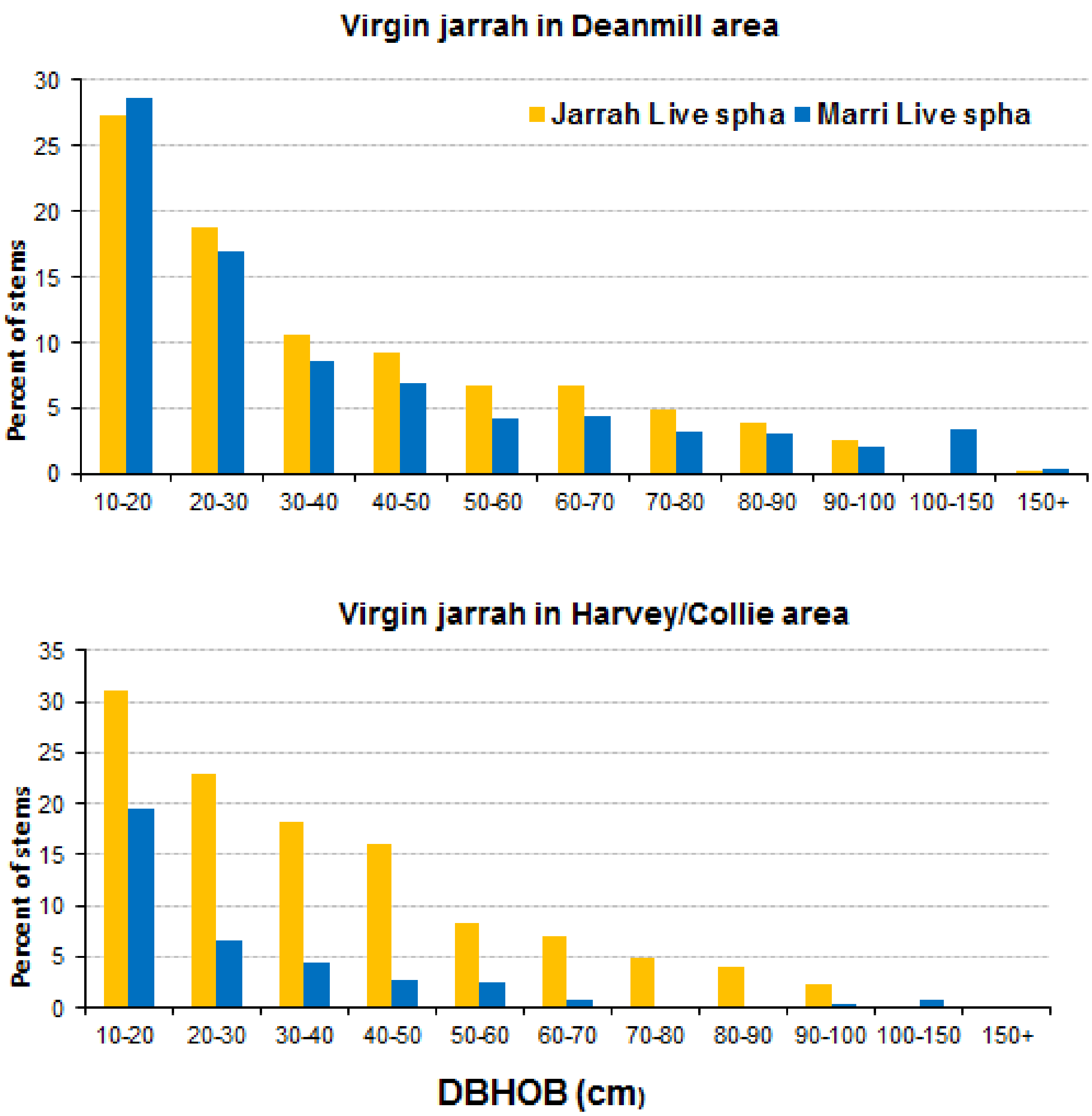


Figure 3: An illustration of the variation in the proportion of marri to jarrah in virgin forest at the landscape scale reflecting the influence of soil type. Based on data from 128 (0.8 ha) plots in the Deanmill area (southern forest) and 80 plots in the Harvey / Collie area (northern forest) established in virgin forest that existed in the 1960s. At the stand scale species composition may vary from pure jarrah to pure marri.

In the Blackwood Plateau, the landform is a mosaic of laterite ridges and seasonally moist sandy flats with yellow duplex soils. In the lower south and south-east of its range, jarrah occurs as a mosaic with shrubby flats and sedge-lands on a previously inundated landscape.

The jarrah forest reaches its best development on the deep lateritic soils of the Darling Plateau with an annual rainfall exceeding 1,100 mm, where much of it occurs in relatively pure stands.

1.5 Influence of fire

The jarrah forest occurs in a fire prone environment. Left unburnt, the jarrah forest will accumulate up to 18 tonnes of flammable litter per hectare (Bell *et al.* 1989). The dry summers make it possible for fire to be sustained and spread on about 120 days during the spring / summer / autumn period and on half of those days at high intensity. Lightning is a common source of ignition. For example, 250 fires were caused by lightning in forest areas in the six years between 1995 and 2001 (McCaw *et al.* 2003). All of the elements for the maintenance of fire are therefore present every summer without any human influence. Fire frequency was believed to have been higher in areas with high aboriginal usage in pre-European times. A study of grass tree scars indicates that the jarrah forest may have been burnt as frequently as one to four times per decade prior to European settlement (Hallam 1975; Hassell *et al.* 2003; Lamont *et al.* 2003). There has been a decrease in frequency and a corresponding increase in intensity of fires since that time (Burrows *et al.* 1995). Apart from small experimental plots, there are few if any examples remaining of areas that have been burnt at a frequency of twice per decade or more (for an extended period). Variation in fire frequency since the 1920s has been described in a report for the RFA (Gill *et al.* 1997). Bushfire and prescribed burning has been systematically recorded in forest areas since 1937 (McCaw *et al.* 2005; Boer *et al.* 2009).

The vegetation of the jarrah forest varies in its adaptation to fire. Nearly 70 per cent of the northern jarrah understorey species and all of the major overstorey and second-storey species are capable of ‘re-sprouting’ after fire. The remainder require seed to regenerate and the germination or seeding of many of the major ‘re-seeders’ are stimulated by fire (Bell *et al.* 1989). The proportion of ‘sprouters’ is higher in the drier uplands. While particular species are favoured by either short or long intervals between fire, maximum species richness has been found to occur at about 2-7 years after a fire (Bell *et al.* 1980; Burrows *et al.* 2003). This corresponds to the time of maximum contribution of re-seeders that regenerate after fire, but which progressively reduce over time since disturbance. A recent study of areas of southern jarrah forest and associated shrub-land showed that fire intervals of less than five years to more than 10 years had no persistent effect on the species richness and composition of plants, vertebrates, invertebrates and fungi (Wittkuhn *et al.* 2010).

1.6 Flora and fauna

The south-west botanical province of Western Australia is recognised as one of the world's 25 global biodiversity hotspots, with approximately 7,400 species of vascular plants, half of which are endemic. However, within this area the high rainfall forest areas (800-1,500mm / annum rainfall) are relatively species poor (Myers *et al.* 2000; Hopper *et al.* 2004).

Within the general forest area[1], the most species-rich types are the sedge-lands and fringes of rock outcrops associated with the forest, rather than in the forest itself.

There are about 245 vertebrate species within the forest area comprising 29 mammals, 150 birds, 44 reptiles, 11 amphibians and 11 fish. Details of flora and fauna species and the conservation measures they may require are discussed in detail in Section 4.1.2 and Appendix 2.

[1] This includes jarrah and karri forests and the associated vegetation within the forested landscape.

2 The silvics of jarrah

2.1 Taxonomy

Jarrah is a member of the subgenus Monocalyptus, a subgenus it shares with four other forest species: blackbutt (*E. patens),* Rates tingle *(E. brevistylus),* red tingle *(E. jacksonii)* and bullich (*E. megacarpa)*.

The jarrah forest occurs in the south-west corner of Western Australia, east of the Darling Scarp where historical rainfall exceeds 600mm / year.

There are three recognized subspecies of jarrah, these are:

- *Eucalyptus marginata* subsp. *marginata* is the main sub-species of jarrah. It ranges from a small to medium tree up to heights of 40m in parts of the south-west forests. Occurring as a mallee in its most northern outlier near Mount Lesueur, and on the southern coastal heaths, it also has two outliers at Jilakin rock east of Kulin and at Tutanning reserve south-east of Pingelly.
- *Eucalyptus marginata* subsp. *thalassica* (Blue-leaved jarrah) differs from *Eucalyptus marginata* subsp. *marginata* in its bluish leaves and often drooping branchlets. This subspecies occurs as a small to medium tree and is the most common form of jarrah in the northern Darling Range north and east of the Brookton Highway.
- *Eucalyptus marginata* subsp. *elegantella* differs from the other subspecies by the smaller, narrower adult leaves and smaller compact habit. It is confined to the foot of the Darling Scarp between Perth and Serpentine and is conspicuous along the South Western Highway southwards from Byford.

Differentiation into sub-species is not supported by recent genetic analysis. There is moderate genetic diversity but low differentiation across the distribution of jarrah (including the outliers). Some isolation by distance is observed (Wheeler *et al.* 2003; Wheeler *et al.* 2006).

There are three recognised hybrids of jarrah. These are:

- *Eucalyptus bupestrium X marginata* - a mallee that occurs north east of Albany.
- *Eucalyptus marginata X megacarpa* - a small tree or mallee that occurs on the Leeuwin-Naturaliste ridge.
- *Eucalyptus marginata X pachyloma* - a mallee that occurs north of Albany and at Tutanning.

A tree previously described as a variety of *E. marginata* is *E. staeri,* a small tree on the sub coastal sand-plains of the south coast. It differs in that it has consistently poor form, thicker bark and thick adult leaves.

2.2 Flowering and seed production

Jarrah is capable of flowering each year, but significant flowering events occur on a four to six year cycle, with most trees within any region flowering at the same time (Abbott *et al.* 1986). Since the floral cycle takes three years to complete, cycles often overlap so that more than one stage may exist on the tree at any one time. A diagrammatic illustration of a single floral cycle is shown in Figure 4.

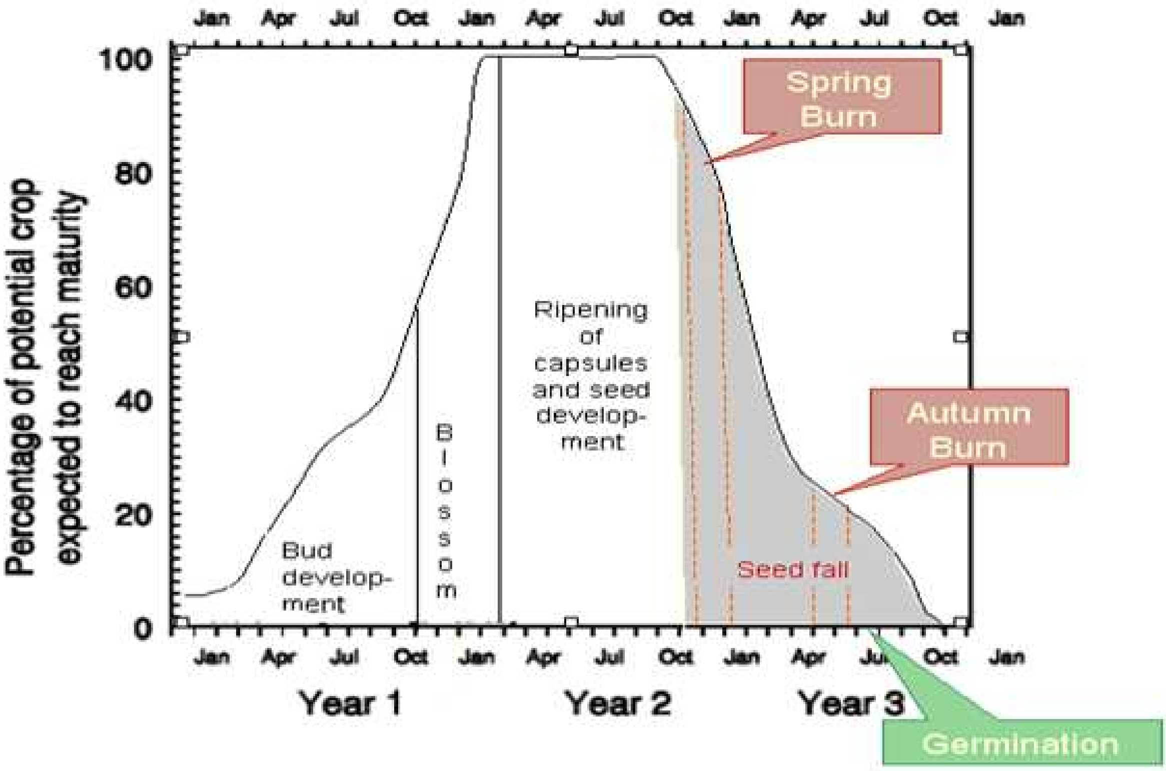


Figure 4: The floral cycle of jarrah (after Whitford and Maher).

Jarrah capsules contain, on average 1.6 seeds per capsule (Abbott 1984b; Whitford *et al.* in review). Laboratory germination is typically about 75 per cent (Monica Dalton, pers. comm.[2]), but has been reported as high as 97 per cent. In nursery conditions jarrah germination is spread from two to four weeks after sowing, compared to a more consistent two weeks for marri and karri. About 20 per cent of jarrah seed is self-pollinated, within the range of most eucalypts (Millar *et al.* 2000).

Leaf density and diameter growth may be reduced by up to 30 per cent during the period of capsule development (Kimber 1978; Abbott *et al.* 1986).

While flowering and seeding may occur on very young jarrah trees, trees in the order of 45cm diameter at breast height over bark (dbhob) are believed to be necessary for the production of reliable seed crops under normal stand conditions. In a study of a 75 year old stand, seed fall increased with stand density up to critical density, after which it plateaued at over 400,000 seeds / ha / annum when measured over a 41

[2] Forest Products Commission Seed Store

month period. Fertiliser application increased flowering but had no effect on final seed production. Only seven per cent of flower buds that were initiated became seed bearing capsules. Seed forecasting can be done with increasing reliability after the later stages of flowering (February) (Whitford *et al.* 2004).

Most of the seed is shed from the capsules in the first summer after maturity (Figure 4) and the presence of seed in the capsules that are still on the tree needs to be confirmed prior to regeneration burning. Virtually all seed from a particular cycle falls within 12 months. Fire will accelerate seed shed under the right conditions, causing most seed to fall within a month of the fire.

Dispersal distance depends on tree height, wind direction and strength. Seed is generally shed to a distance of 1.5 times tree height, with most falling within one tree height.

Seed predation by vertebrates and invertebrates can account for the loss of more than 90 per cent of jarrah seed. Invertebrates (mainly ants) account for relatively more losses in summer, while vertebrates are more important in cooler periods (Stoneman (1992). Seed harvesting may be increased on burnt seedbed relative to unburnt (Koch 1992). Seed that is covered by soil is protected from predation, but germination is reduced if the covering is more than 5mm thick (Stoneman *et al.* 1994a).

2.3 Plant development stages

Jarrah regeneration progresses through several distinct stages before it has the capacity to develop into a sapling (Abbott *et al.* 1986). These stages are illustrated in Figure 5. The rate at which plants develop is influenced by moisture availability (soil, rainfall and competition) and nutrition (site fertility and ash-bed).

2.3.1 Germination

Germination begins after the first winter rains and most occurs in the early part of winter when the soil is still warm. Germination, emergence and survival of seedlings are improved on disturbed seedbed, and germination and emergence are better under canopy than in the open, probably due to more uniform temperature. However, survival is lower under a canopy.

2.3.2 Seedling

Following germination, seedlings develop rapidly with both growth rate and survival improved by cultivation and fertility (Abbott *et al.* 1986). Survival during the first summer drought is dependent on the rapid development of a root system. Seedlings are vulnerable to grazing by both vertebrates and invertebrates. Grazing pressure may be higher after patchy fires than after larger fires, where grazers are more likely to be satiated (Stoneman *et al.* 1994c).

Seedling survival under canopy is reduced due to increased competition for moisture (Stoneman 1992). These effects are exacerbated in drier parts of the forest where year-to-year survival is much more variable than in more favourable climates. The survival rate of marri seedlings is considerably higher than for jarrah (Abbott 1984b).

Survival can be adversely affected by heavy rain that washes out germinants and establishing seedlings. However, moisture stress and high temperatures in the first summer are responsible for most seedling mortality.

The lower regrowth stocking found in the eastern jarrah forest is a reflection of the effect of lower rainfall and summer moisture stress. Similarly the low stocking of jarrah in stands with a dense understorey of root stock species (e.g. *Taxandria* spp.) is a reflection of the effect of understorey competition on the survival and growth of seedlings.

Overstorey competition reduces the growth rate of seedlings to as much as one eighth of the growth of seedlings in the open (Stoneman *et al.* 1994b).

2.3.3 Lignotuberous seedling

Within one to three years, the seedling begins to form a lignotuber, a small swelling at the base of the stem. The development of the lignotuber reduces, but does not eliminate its vulnerability to physical damage by grazing or fire (Abbott *et al.* 1986). An ash-bed and regular burning enhances the growth of the lignotuber (van Noort 1960; Abbott *et al.* 1984).

2.3.4 Seedling coppice

Following damage to the shoot of the lignotuberous seedling, new (multiple) shoots emerge from the lignotuber to form what is described as ‘seedling coppice’. The above ground shoots form a small multi-stemmed bush growing little in height. However, the lignotuber continues to grow and the taproot extends. The rate of growth of the lignotuber is affected by nutrition (soil type), competition (overstorey and understorey) and rainfall. Mild fire will stimulate the rate of development of the lignotuber (van Noort 1960; Kimber 1971). While a reduction in overstorey density will increase the growth rate of the lignotuber, under forest conditions the above ground shoot will not develop until the lignotuber has reached a critical size.

Seedling coppice remains vulnerable to drought. While post-fire survival is enhanced as the lignotuber develops, it is not immune to fire effects. In heavy fuel the lignotuber may develop above ground within the litter where it is vulnerable to fire. Below ground lignotubers are also vulnerable when soil moisture is low or the fire intense (Harris 1956). Marri appears to be more vulnerable in these circumstances.

2.3.5 Ground coppice

When the lignotuber reaches a critical size, if the overstorey competition is reduced, jarrah is capable of rapid development into a sapling. The assumption is that this stage is reached when the root system is sufficiently deep to provide the necessary water supply to sustain growth (Harris 1956; Crombie 1997). At this stage it is described as ground coppice.

In the northern jarrah forest, the critical size of the lignotuber before it can develop into a sapling is considered to be 10cm in diameter. The critical size in the southern jarrah forest, where summer drought is less severe, is about 5cm, but up to 8cm on poor sites (Howesmith 1983).

The time to reach the critical size is influenced by site, burning history and competition (Abbott *et al.* 1984). Typically it may take 20 years to develop or it may take as little as two years with cultivation and freedom from competition (Harris 1956). In bauxite rehabilitation, 67 per cent of seedlings became saplings within 13 years, regardless of whether they were seeded or planted, but this proportion was reduced in the absence of fertiliser or the presence of dense understorey (Koch *et al.* 2005).

Ground coppice develops as a shrub to 1.5m in height. While it is multi-stemmed it is known as ‘incipient’ ground coppice and becomes ‘dynamic’ ground coppice when one shoot begins to dominate.

Seedling | Lignotuberous seedling | Seedling coppice | Incipient ground coppice | Dynamic ground coppice

Figure 5: Stages of jarrah regeneration.

For management purposes it is important to understand the stage of development of the regeneration that exists in each stand. Only regeneration that has reached the ground coppice stage will develop rapidly into saplings when overstorey competition is reduced.

2.3.6 Stump coppice

When felled, jarrah stumps coppice strongly. Coppice stems grow rapidly, taking advantage of the water available to the plant from the established root system (Crombie 1997). Stump coppice can be an important source of regeneration, especially in drier forest areas where the establishment of new seedlings and lignotubers is less reliable. However, it has the disadvantage that multiple stems are common and coppice which develops high on a stump is more likely to split from the stump. On the other hand, the deliberate coppicing or ‘mullinising’ of malformed or fire damaged saplings can result in excellent single stemmed regrowth. This was the source of much of the present day pole stands that were treated in the 1930s (Kessell 1931; van Noort 1960; Bradshaw 2005).

2.3.7 Saplings, poles and mature trees

Following a reduction in overstorey density by timber harvesting or death of overstorey trees, the 'released' ground coppice develops rapidly into saplings, reaching 5-6m in five years (Campbell 1956). Saplings (arbitrarily defined as >1.5m tall and <15cm dbhob) have strong apical dominance, abscising and shedding their side branches as they become shaded and less efficient (Jacobs 1955). The lignotuber becomes ill-defined at the base of the stem.

The pole stage (nominally 15-45cm dbhob in high quality stands) is reached when apical dominance begins to lessen, side branches become more persistent and the crown changes to a broader, more rounded shape. Height growth rate slows, but high quality stands achieve about 50 per cent of their final height at about 30 years from the time of lignotuber release. Maximum bole height is achieved by about 12 years. Bole height is about two-thirds of the total height at the sapling stage and about one-third to a half at maturity.

Final mature height varies from 15-40m, depending on site quality, corresponding to a diameter at breast height of 40-100cm respectively (Abbott *et al.* 1986).

Individual jarrah trees are not particularly long lived and the oldest tree measured to date (based on ring counting) was 377 years old, with relatively few living beyond 250 years (Burrows *et al.* 1995). Mortality of dominant and co-dominant trees in a mixed aged low rainfall site has been measured at 1.1 per cent per decade, similar to the overall mortality of 1.24 per cent per decade (Burrows *et al.* 2010).

Natural tree fall is relatively low. A study to determine the survival of habitat trees found that trees over 70cm dbhob fell at the rate of 2.4 per cent per decade. The rate of natural tree fall of jarrah and marri are similar. This compares with 36 per cent per decade for ash-type forest in Victoria (Whitford *et al.* 2001b).

2.4 Stand development

2.4.1 Regeneration dynamics

Regeneration of a natural jarrah stand occurs over an extended period of time. Little of the seed that falls on a regular basis will survive predation and little that germinates will survive the first summer. The chances of survival are substantially increased if the area has been burnt but even then, most that have survived will succumb to competition from existing vegetation. Those germinants that do survive add to the lignotuber pool, remaining in this state until the opportunity arises for their further development. Because of the continuous recruitment process, there is usually a full range of lignotuber stages present at any one time, collectively known as the 'lignotuber pool' (Stoate *et al.* 1938). Virgin forest will generally contain a smaller lignotuber pool than partially harvested forest because of the greater competition exerted by the overstorey (Abbott *et al.* 1984).

Although most of the jarrah forest has a well-developed lignotuber pool of more than a thousand ground coppice per hectare, this is not always the case. Lower survival rates and smaller lignotuber pools are generally found in the drier eastern forest, on poorer sandy soils, in very dense mature forest and in sites with a high proportion of root stock understorey. The density of the lignotuber pool is highly variable at the local scale (Figure 6), and while some general trends exist, there is no close correlation between the density of lignotubers and vegetation complex, soil type or rainfall at the ‘patch’ scale.

The opportunity for ground coppice to develop into a sapling arises when a gap in the canopy occurs as a consequence of a falling tree or a tree that is killed or even temporarily defoliated by fire. Overstorey competition is also reduced by the decline of a senescent overstorey tree.

The ‘released’ advance growth[3] rapidly develops to fill the gap with fast growing saplings. In the case where lignotubers are released by senescent overstorey, lignotubers will be gradually ‘released’ from competition with the outer edges of the senescent crown, slowly progressing towards the bole of the tree. In very small gaps or gaps created by temporary defoliation, the recovering canopy may arrest the development or suppress new sapling growth. While young regrowth is relatively intolerant of competition, it is very persistent and the suppressed stems continue to survive until the next opportunity for release. In these circumstances, differentiating between the dominants of one cohort and the suppressed members of an older cohort can be difficult. Stand age is an unsatisfactory concept under these circumstances. The result of this periodic release and suppression is the development of stands of relatively high density in a state of continuous competition and stress.

Figure 6: These photographs taken from the same position, but in the opposite direction, illustrate the variable lignotuber stocking that can occur at the patch scale.

Because of the periodic development and multiple stages of regeneration existing at any one time, the species composition of the next generation is determined by the composition of the existing ground coppice rather than the current year’s seed source. It is a common phenomenon, particularly in southern forests, for the lignotuber pool to consist predominantly of marri even in stands with a predominantly

[3] A term used to describe the established regeneration stages of jarrah, most often ground coppice.

jarrah overstorey. The progression towards jarrah over the ensuing rotation, particularly in the northern jarrah forest, is believed to be due to jarrah's relatively greater tolerance to fire (Figure 9). Periodic outbreaks of canker (*Quambalaria coryrecup*) may also reduce marri numbers over time.

2.4.2 Competition and site potential

Competition for resources in sapling stands intensifies from the time of crown closure at about eight years after release. Saplings compete strongly and rapidly sort themselves into dominance classes and self-thinning occurs, albeit at a modest rate (10 stems / ha / annum at 1,500 stems per hectare (spha)). The largest trees in the cohort are responsible for most of the stand growth. For a 12-year-old sapling stand, the fastest growing 300 spha (of 1,600 spha) accounted for 60 per cent of the total stand growth. As the stand continues to develop into a pole stand, competition for dominance continues, but self-thinning slows. Total stand growth declines at maximum density and the stand is said to have 'locked up'. At 40 - 65 years of age the fastest growing 300 stems (of 1,200 spha) account for 90 per cent of the total growth, while the slowest 600-700 spha account for only 3 per cent of the growth (Stoneman *et al.* 1989b).

While self-thinning continues throughout the life of the stand, it continues to be a slow and episodic process. An even-aged stand that starts life at 1,500 spha of saplings may reduce through competition to as few as 60 spha by the time the trees in the stand reach the end of their lives. From the time stands reach maximum density at maturity, growth merely replaces mortality, maintaining the stands in a state of relatively constant density.

The capacity to utilise the full site potential is also influenced by stand age in that young stands reach their 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 (Crombie 1997). The trajectory of basal area, shown in Figure 7 up to age 20 years, represents the maximum trees of that age the site can support. By the time the trees reach 20 years of age they can exploit most of the site potential. The achievement of maximum basal area will take longer where a full stocking of ground coppice does not exist at the time of release.

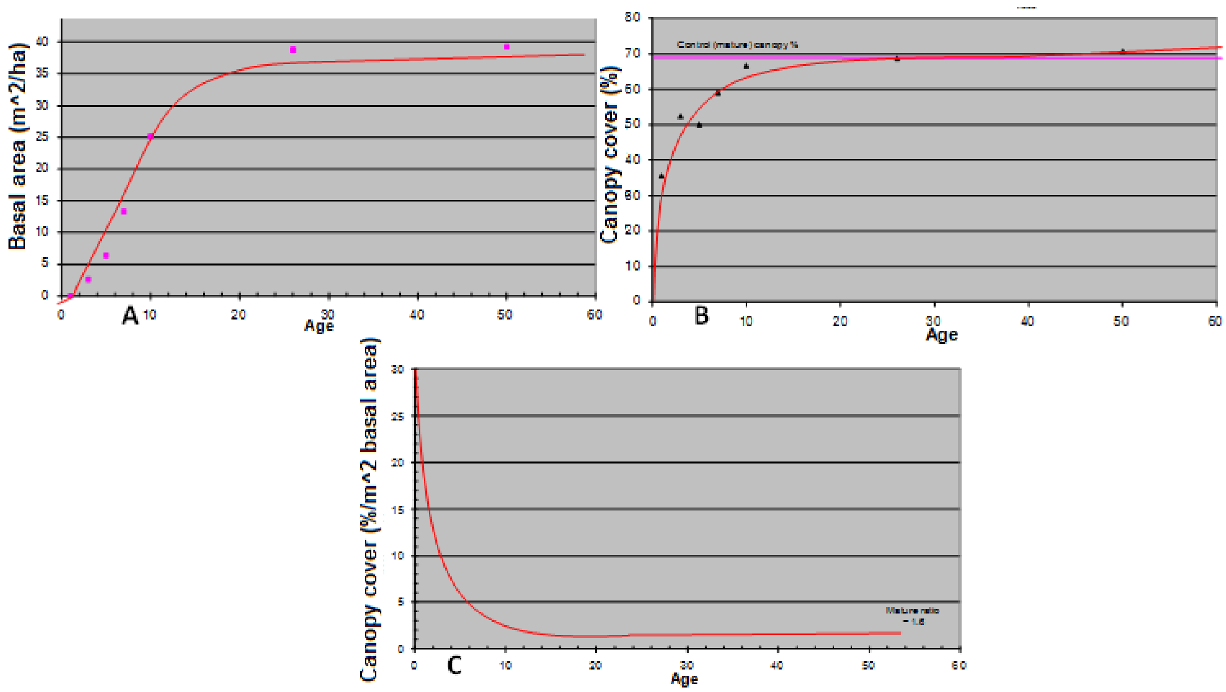


Figure 7: Changes in basal area and canopy cover with age in even-aged southern jarrah forest. Graphs A and B are from Stoneman et al. (1988). Graph C is derived from A and B.

2.5 Response to fire

Jarrah's tolerance to fire varies with age and development stage. Tolerance encompasses resistance and response to damage. Prior to the ground coppice stage of regeneration, the above-ground shoots have a low resistance to fire damage, but increasing ability to respond to that damage by re-sprouting from the lignotuber once it has become established (Bell *et al.* 1989). Lignotubers will survive most fires from about five years of age, though they may be killed by intense fire in dry soil conditions. Lignotubers that develop in the duff layer, as occurs in some southern forests, are also vulnerable to fire. High intensity regeneration burns have been shown to kill up to 20 per cent of marri lignotubers (White 1971).

Saplings are readily damaged by fire until they are 5-6m tall. Once the sapling stage is reached, the bark is thick enough to withstand a mild fire and the tree tall enough to escape damage to the leading shoot (Peet *et al.* 1971). Damage to the leading shoot can result in the development of a fork or a kink in the stem. However, it is also possible that the kink often observed after a fire may be due to the post-fire growth spurt from a stem affected by 'carrot topping' (Figure 8). The causes of carrot topping are yet to be fully understood.

Saplings that are killed by fire have a strong capacity to re-shoot (coppice) from the base of the stem or the lignotuber.

Bark thickness is an important element of fire resistance. The bark thickness increases with tree diameter at the rate of about 1mm / cm of diameter growth until about 30cm diameter, then more slowly until it reaches a maximum thickness of about 35mm. Bark thickness is reduced by burning.

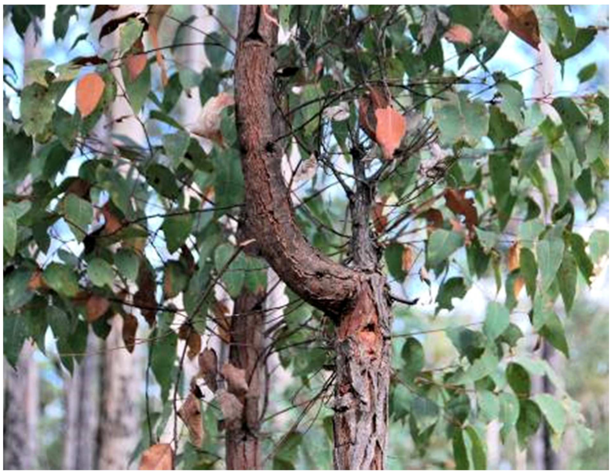


Figure 8: Kink caused by a rapid growth response following the development of 'carrot topping'.

Jarrah which has developed past the sapling stage will suffer little cambial damage if burnt at fire intensities of less than 600kW / m. A large proportion of trees that are burnt with sufficient intensity to cause complete crown scorch suffer no stem damage (Bell *et al.* 1989).

Trees are damaged by burning debris close the butt of the tree. This is particularly so in autumn fires when more of the heavy log debris is dry and the residence time of these fires is longer. Resultant cambial damage at the butt leads to 'hollow-butting' and, without bark covering, increasing vulnerability to burning down with each successive fire. Hollow-butting caused by fire and subsequent breakage at this point is the principal cause of natural tree fall in the jarrah forest, being implicated in more than 70 per cent of cases (Whitford *et al.* 2001b).

Trees that survive being scorched or defoliated respond by coppicing from the base or re-shooting from epicormic buds on the stem or within the crown. Plots established in an area of forest defoliated in the Mt Cooke bushfire of 2007, which had an estimated intensity of 20,000kW / m, illustrate different responses according to tree size. Approximately 90 per cent of small trees (dbhob < 20cm) survived by coppicing from the base. However, more than 50 per cent of trees larger than 30cm were killed; those that survived doing so with epicormic shoots from the bole or crown. Larger trees are apparently unable to coppice from the base if they are killed at ground level (Burrows *et al.* 2003). Marri was more sensitive to fire than jarrah at larger sizes (Figure 9). Damage assessment following the 1961 Dwellingup fire (intensity of 15,000kW / m) showed that 23 per cent of trees less than 30cm dbhob, and 62 per cent of trees greater than 30cm dbhob, fully replaced their crown with epicormic growth (Peet *et al.* 1968).

Damage to the stem by fire is a major cause of degrade in wood. Apart from the direct damage leading to the inclusion of gum (kino) rings and dry wood, it can lead to insect and fungal entry and pre-disposition to termite attack. There is a strong correlation between the extent of stem damage and the effect of fungal and termite attack 30 years later. This can lead to mechanical failure in the tree as well as damage to wood values (McCaw 1983). The incidence of fire scars in jarrah trees has been shown to be greater since the 1880s (Burrows *et al.* 1995), corresponding to the declining frequency and increasing intensity of fires (Lamont *et al.* 2003).

Kimber (1971) suggested that about a third of jarrah stem growth occurs in spring and about two thirds in autumn / mid-winter. However, data for pole sized trees over a three and a half year period from 1987 to 1990 at Inglehope showed that the season of lowest diameter growth was late summer, and growth rate in intervening seasons was more irregular (Stoneman *et al.* 1996). Crown scorch in autumn stops stem growth until leaf flush in the following January, representing the loss of almost a year's growth. Trees scorched in spring lose only a short period of growth (Kimber 1971).

The relationships between scorch height, fire intensity and flame height in spring and autumn in jarrah forest have been described by Burrows (1997). The effect of fire varies with species and tree size (Figure 9).

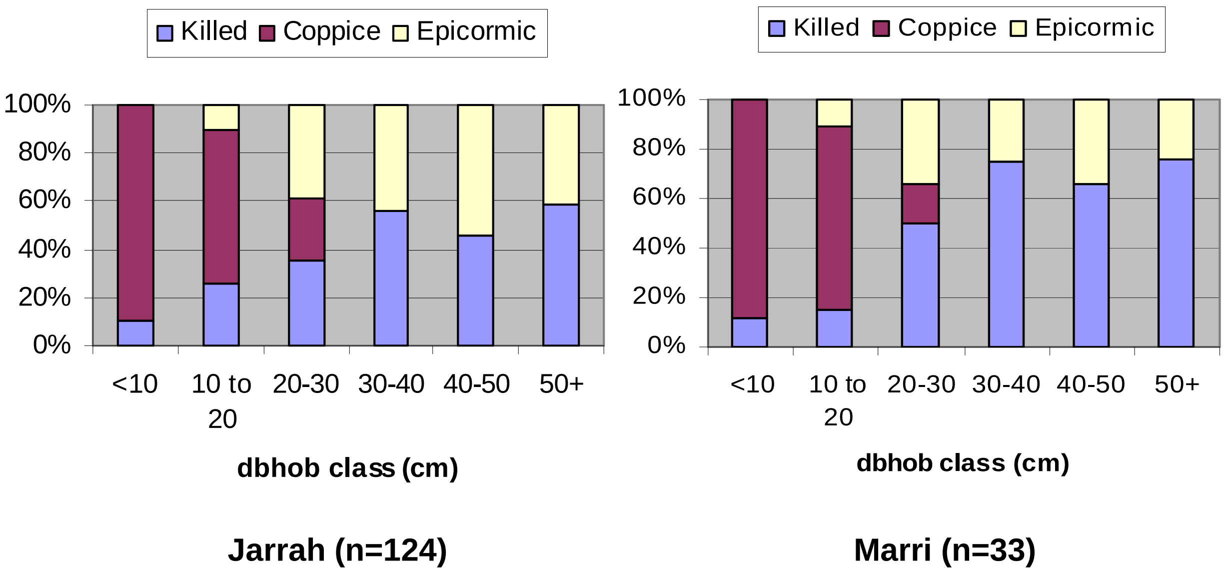


Figure 9: Effects of tree size on mortality for jarrah and marri plants. Note the higher resilience of jarrah compared to marri (Burrows unpublished data from trees assessed in the aftermath of the Mt Cooke bushfire of 2007).

2.6 Water relationships

Jarrah accesses water at two levels. Riser roots near the surface, where it requires moisture for nutrient uptake, and to depths of 20m or more through sinker roots, which access large stores of water throughout the soil profile. In some soil types the sinker roots follow the same channels that serve to recharge the groundwater (Abbott *et al.* 1989). Although jarrah has access to moisture at depth, it is subject to summer moisture stress, particularly in low rainfall areas (Crombie 1987). Stomatal closure and leaf shedding allow jarrah saplings to maintain normal water relations and avoid moisture stress with as much as 50 per cent of its surface roots removed (Crombie *et al.* 1987).

Annual rainfall deficit (the difference between annual potential evaporation and annual rainfall) varies from 0 to 800mm across the jarrah forest, with higher rainfall deficits indicating a more moisture limited environment (Croton *et al.* 2015). There is a strong relationship between rainfall deficit and leaf area index (LAI) across the jarrah forest (Croton *et al.* 2015). Maximum forest density is ultimately limited by access to water (Figure 2, Figure 10, Figure 11), and moisture stress in trees may occur at least a month earlier in the drier eastern forest than it does in the west (Crombie *et al.* 1988).

Water potential and stomatal conductance increase with the size of the plant from seedling to sapling to tree, reflecting its greater access to moisture. Stomatal conductance of ground coppice is higher than that of saplings and stump coppice is higher than that of trees, reflecting a lower shoot / root ratio and less moisture stress. Stomatal conductance of stump coppice is higher than that of ground coppice in low rainfall areas (Crombie 1997).

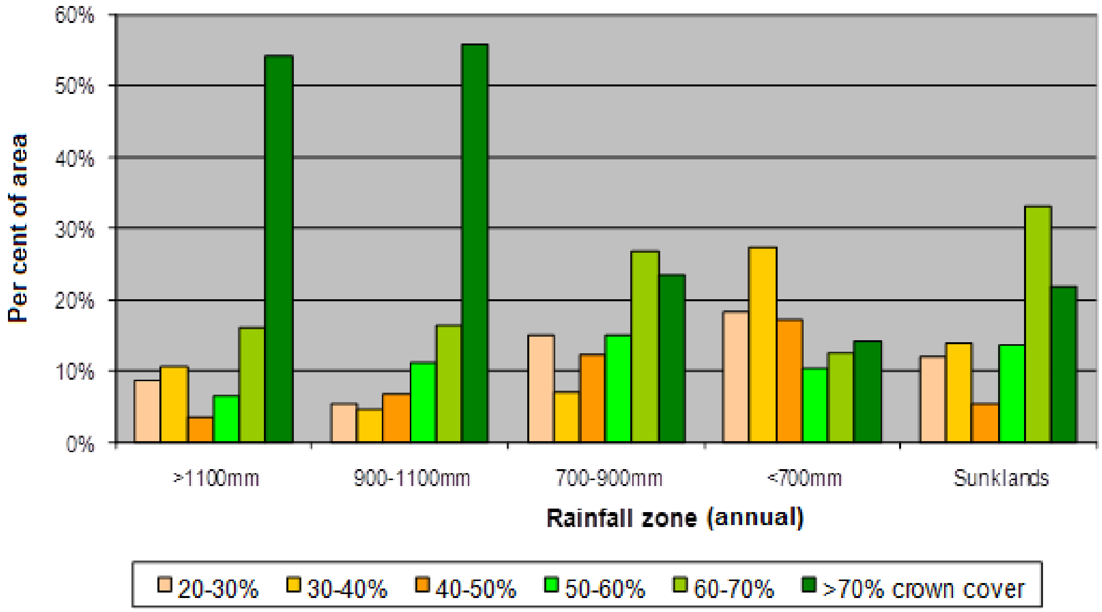


Figure 10: The total crown cover of virgin jarrah forest as it existed in the 1960s (707,000 ha), by annual rainfall. The weighted mean crown density is 62 per cent (>1,100mm), 65 per cent (900-1,100mm), 55 per cent (700-900mm), 46 per cent (<700mm) and 60 per cent for the Sunklands (1,100mm). Sites with a crown cover of less than 20% are generally associated with moist, low fertility sites where difficulties of regeneration may be more limiting than site capacity. These are defined as jarrah woodland and are excluded from the data. Crown cover data is derived from API mapping. Rainfall zones are based on data to 1979 (Hayes et al. 1981).

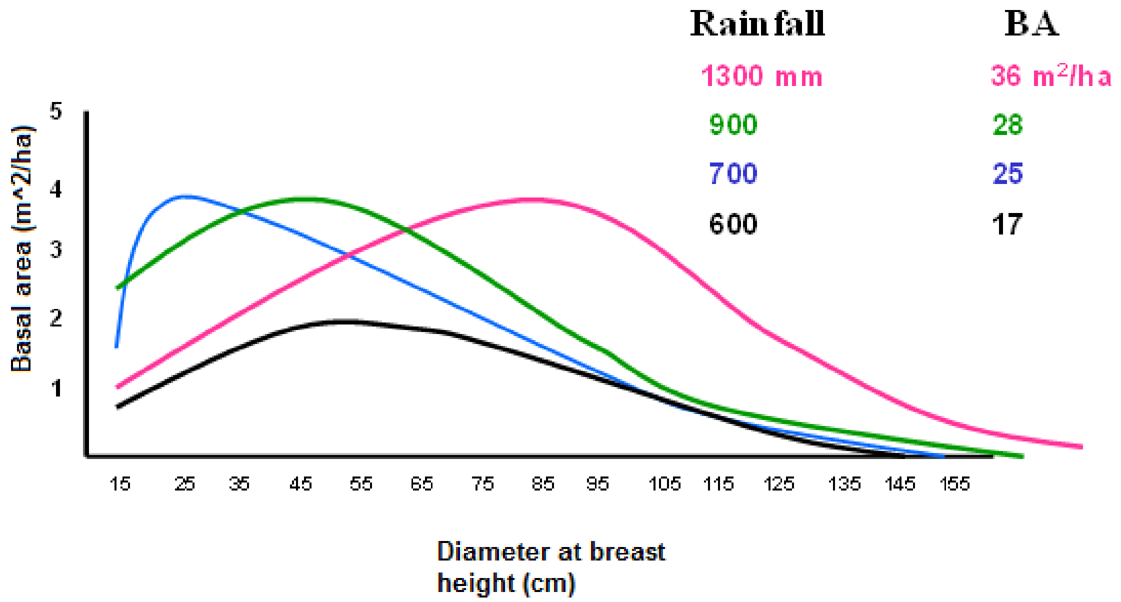


Figure 11: Basal area by size classes for virgin forest in different rainfall zones. Based on data from 1,128 (0.8 ha) inventory plots established in virgin forest that existed in the 1960s. Size class distribution is also influenced by site and disturbance history. While very little virgin jarrah forest remained in the prime northern forest by 1960, limited evidence suggests that it also conforms to this pattern. Rainfall zones are based on data to 1979 (Hayes et al. 1981).

Sapwood area has been shown to be a key determinant of transpiration by trees. A stand of small diameter trees has a larger sapwood area than a stand of larger trees at the same stand basal area. In a study of patches of regrowth and mature jarrah forest, the regrowth patch (with only two-thirds of the basal area of the mature patch) had 30 per cent greater (LAI), nearly twice the sapwood area and double the transpiration of the mature patch (Macfarlane *et al.* 2010).

3 Forest structure

3.1 Range of stand structures

The natural or virgin jarrah forest is a multi-aged forest with cohorts of different regeneration age generally occurring as a fine mosaic. Even-aged patches are usually relatively small. Abbott (1984a) described the structure of the jarrah forest as aggregated at small scale and random at large scale. This structure is a consequence of the lignotuberous method of regeneration, and persistence and tolerance to fire, the details of which have been discussed. The typical range of structure found in jarrah forest includes regrowth, mature, two-tiered and all-aged stands as illustrated in Figure 12.

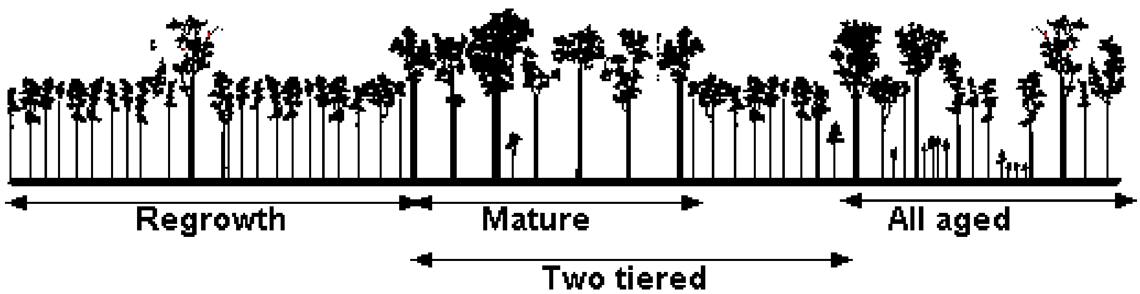


Figure 12: Diagrammatic representation of the structural variation of the jarrah forest with common descriptions of the various elements.

While often portrayed as being dominated by mature trees, the size class distribution of the virgin forest forms a negative exponential curve. The stocking versus diameter class distribution for several virgin forests as it existed in the 1960s is shown in Figure 13 and the basal area versus diameter class distribution is shown in Figure 13. The distributions for cutover forest are shown in Figure 14 (stocking) and Figure 15 (basal area).

The structure of the virgin forest has been altered by disturbance (principally mining, timber harvesting, silviculture and dieback) since the 1870s, resulting in a wider range of structure and a higher proportion of regrowth stands. Approximately five per cent of the jarrah forest is dominated by regrowth younger than 20 years old, 38 per cent by 20-120 year old trees and 57 per cent is dominated by mature trees (Source: FMIS database at 2010). The history of timber harvesting and silviculture is described in Appendix 1.

The last comprehensive mapping of structure was undertaken in the 1950s and 1960s using API. The API mapping classified the forest on the basis of forest type, structure and height to a 2ha resolution. For eucalypt forest, forest type identified the dominant eucalypt species as well as other eucalypt species that contributed more than 10 per cent of tree numbers. Stand structure identified canopy cover of the upper strata (i.e. mature trees) and of the total stand (i.e. the mature trees as well as poles or saplings). Height class reflected the co-dominant height of the original forest (i.e. a measure of site potential). This classification is described in detail in Bradshaw *et al.* (1997a).

While the API information on structure is now dated in areas subject to subsequent disturbance, original tree height and total density are still relevant as indicators of site potential. Species composition is also relevant, although it is recognised that reliability varies with the quality of the aerial photography at the time. Forest structure also provides valuable information when interpreted in the light of subsequent harvest history.

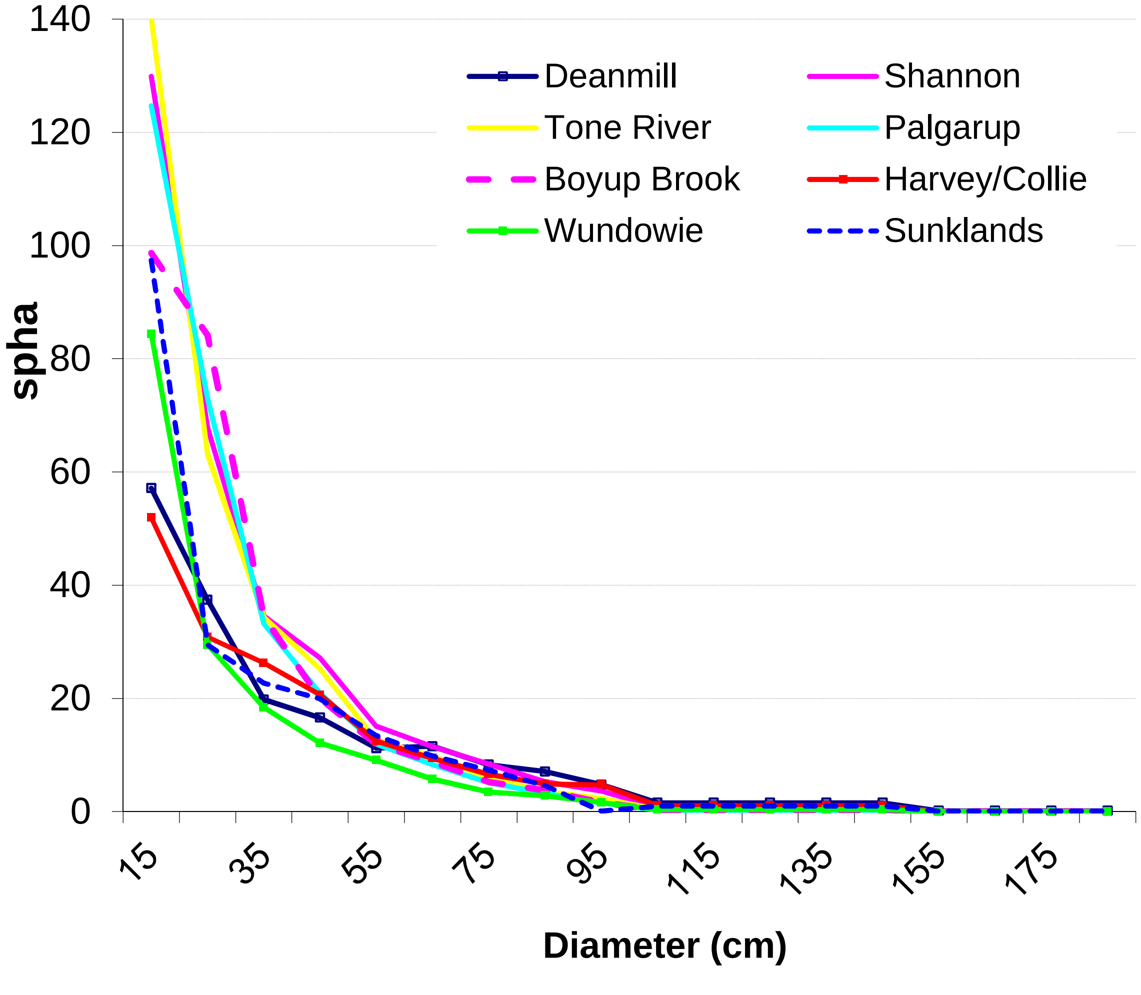


Figure 13: Stems per hectare in virgin forest in a range of sites at the landscape scale. The graph is based on data from 1,128 (0.8ha) inventory plots established in virgin forest that existed in the 1960s. Stem numbers in the dbhob range 100-150cm has been averaged across each dbhob class.

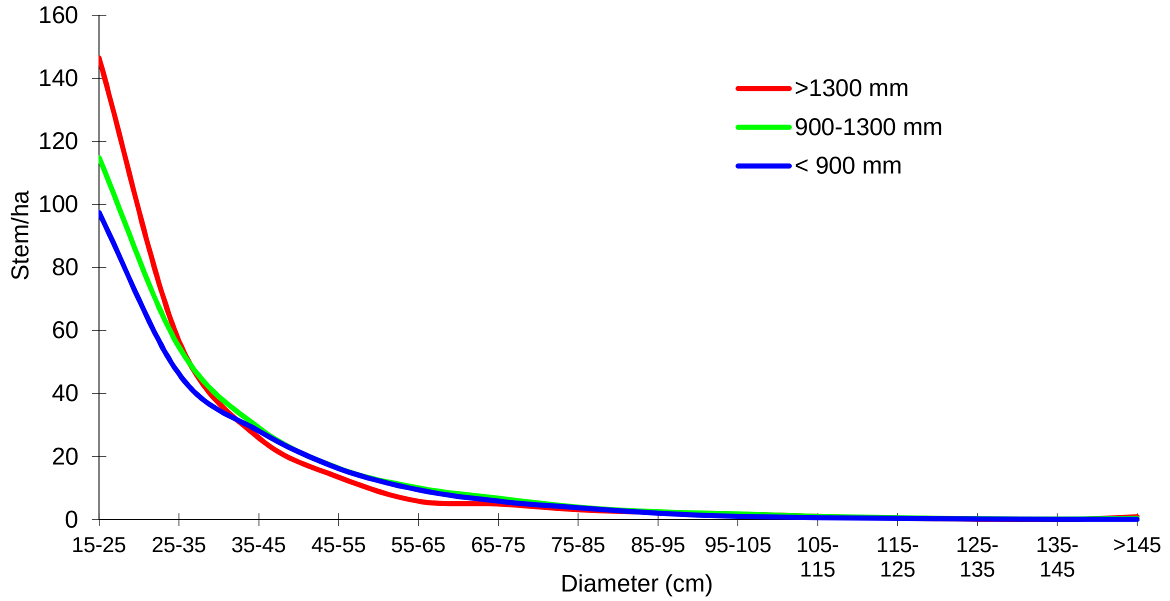


Figure 14: Stems per hectare in previously harvested forest with different annual rainfall based on rainfall isohyets to 1979. Stem distribution data is based on 2,829 sample plots established in the 1990s cf stem distribution for virgin forest in Figure 13.

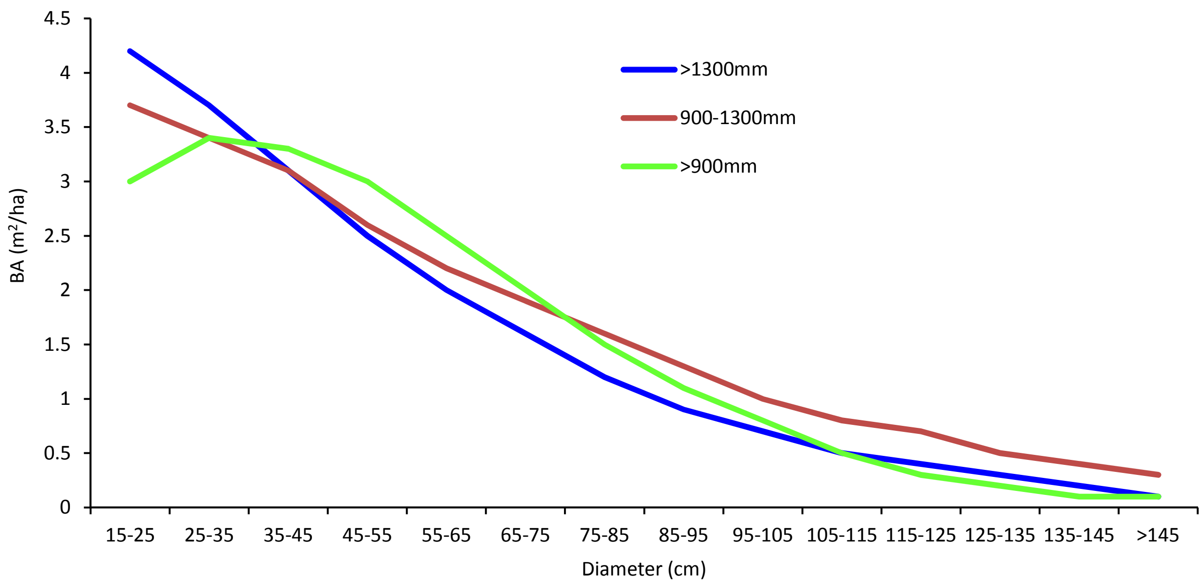


Figure 15: Basal area by size classes for jarrah forest with different annual rainfall fitted to a Weibull distribution. Data is based on 2,829 sample plots established in the 1990s in cutover forest. Rainfall data are based on isohyets to 1979 (Hayes et al. 1981). cf basal area distribution for virgin forest at Figure 11.

3.2 Structural attributes of stand development stages

The following description of the development stages of even-aged jarrah forest is based on that presented in the 1992 draft FMP (CALM 1992).

3.2.1 Establishment stage

This stage of development includes the seedling stage (less than one year, usually with cotyledons present); lignotuberous seedling stage (older than one year, cotyledons absent, one stem with a lignotuberous swelling); seedling coppice stage (obvious lignotuber, multiple shoots); and the ground coppice stage (multiple shoots less than 1.5m in height). The establishment stage finishes when the dynamic coppice shoot exceeds two metres. The initial period of development from seedling to ground coppice may take from five to 20 years or more depending on the competition from the overstorey and fire regimes; the development from ground coppice to sapling does not begin until overstorey competition is reduced sufficiently for their 'release', after which it make take as little as 2-3 years to reach a height of two metres.

3.2.2 Juvenile stage

The juvenile stage is characterised by stands consisting of saplings with a crown of small branches, which will be shed as the tree gains height. Branches are shed from the base of the crown, as the formation of a clear bole commences. This juvenile or primary crown usually contains about four years of growth and vigorous crowns are conical in shape. This stage ends when the diameter over bark (dob) of the co-dominant trees reaches 15cm, at an age of between 15-30 years (Abbott *et al*. 1986). By this time the stand has entered a period of intense inter-tree competition and a sorting into dominance classes that will continue throughout the life of the stand until the stage of senescence.

Crown cover and basal area increase with age. In high and moderate quality stands, 90 per cent of maximum canopy cover is reached 10 years after release, and 90 per cent of maximum basal area by 20 years (Stoneman *et al.* 1989a) (Figure 7).

3.2.3 Immature stage

This stage consists of pole sized trees between 15cm and 45cm in diameter. During this stage apical dominance of the trees reduces, lower branches extend to form semi-permanent and permanent shaping branches and the crowns become more rounded. Secondary crowns of epicormic shoots may develop on larger branches that are damaged by fire or by physical damage. Self-thinning slows. The lateral spread of the crown commences during this stage, which concludes at about 70-120 years.

3.2.4 Mature stage

The mature stage is reached when the diameter reaches 45-60cm depending on site.

During this stage, large persistent branches develop. As the primary crown pushes outwards, it may be weighed downwards, although this is less pronounced in jarrah than other eucalypts. Epicormic shoots and dead branches become more common within the crown. The occurrence of hollows in branch stubs increases from about 130 years of age. This stage of development concludes at about 250 years.

3.2.5 Senescent stage

The age from which senescence occurs may vary considerably depending on the site and the stresses that the stand has been subjected to over its life. At this stage the majority of the crown comprises a secondary crown of epicormic shoots. The epicormic branches are never as efficient as the branches of the primary crown. They may live for a few years or decades, then break and be replaced. This process may be repeated many times before eventual death.

Leaf area decreases and the senescent trees exert less competition and influence within the stand. This provides the opportunity for advance growth that has developed beneath the canopy to be released in gaps created by trees that die and near the extremities of the crowns of standing senescent trees (Figure 16).

Mortality of trees in a stand may occur over an extended period. The largest tree measured by Burrows (1991) in an analysis of the ages of large jarrah and marri trees was 377 years old. It is likely that most trees die by the age of 250 years. The gradual decline and death of trees in the senescent stand provides for an extended period of regeneration and the staggered beginnings of a new generation.

The attributes described above apply to even-aged stands which, as indicated in Section 2.3.7 above, are uncommon or at least not extensive in natural jarrah forest. It is therefore more common that more than one of these development stages will occur within a stand at a variety of spatial scales, both in virgin forest and in forest that has been previously harvested (Figure 12). Nevertheless, it is a useful way of understanding the predictable stages of development that occur in the life of a jarrah stand.

3.2.6 Old-growth forest

Old-growth forest (Figure 16: A stand of old-growth jarrah forest progressing towards a regrowth stand. As the competitive influence of the senescent trees reduces, regrowth becomes established between the old trees, gradually encroaching on them as they senesce and eventually die. (Reserve 4596 near Amphion block). 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)*. 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)*. The focus is on the stand condition rather than individual trees. The interpretation of this broad concept depends on the ecological attributes of each forest type.

In forest that is subject to regular, naturally occurring fire, such as the jarrah forest, ‘ecologically mature’ refers to the overstorey only; the understorey may consist of the full range of seral stages. 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 jarrah 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 (CWD) (Bauhus et al. 2009) are not always a characteristic of old-growth jarrah forest. The understorey, litter depth and CWD attributes are more a function of cutting history and fire frequency than they are of overstorey age. For example, harvested forest may have up to twice the volume of CWD than virgin forest (Figure 20) and harvested forest may be more structurally diverse than virgin forest at some scales (Figure 12).

The values of old-growth forest for flora and fauna have been widely discussed in relation to various forests. However the wide variation of attributes that exist in old-growth jarrah forest suggests that studies of the function of individual attributes such as structural diversity, hollows, and CWD will be more productive than attributing functionality of old-growth forest *per se*.

Figure 16: A stand of old-growth jarrah forest progressing towards a regrowth stand. As the competitive influence of the senescent trees reduces, regrowth becomes established between the old trees, gradually encroaching on them as they senesce and eventually die. (Reserve 4596 near Amphion block).

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*". It further defines ecologically sustainable forest management as "*a management system 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 2013a).

The FMP adopts the slightly modified Montreal Criteria of sustainability as the framework within which to identify management actions consistent with the principles of ecologically sustainable forest management (Conservation Commission 2013a).

The 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 the 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 areas subject to timber harvesting.

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

- 41 per cent of the jarrah forest is within large formal reserves such as national parks, nature reserves and other formal conservation areas
- 11 per cent is in informal reserves such as stream reserves, diverse ecosystem zones and old-growth forest
- 48 per cent is in areas that may be subject to timber harvesting (Conservation Commission 2013a).

In the period 2004 to 2011, approximately 6,600ha of jarrah forest were partially harvested each year, representing 1.0 per cent of the forest available for timber harvesting. Approximately 400ha (0.05 per cent) per year was cut to create gaps, mostly less than 2ha in size. As well as the area subject to timber harvesting, about 800ha (0.1 per cent) per annum was cleared for mining (Figure 17, Figure 34). The average annual area cutover under the FMP is estimated to be approximately 13,500 hectares, inclusive of clearing for mining (1000 ha / yr), thinning of mine-site rehabilitation (not previously undertaken) and thinning of older regrowth jarrah stands (Conservation Commission 2013a). The expected increase in area cutover is also in part due to the low volume per ha available from areas previously cut to shelterwood and now ready for regeneration release (Section 4.2.4).

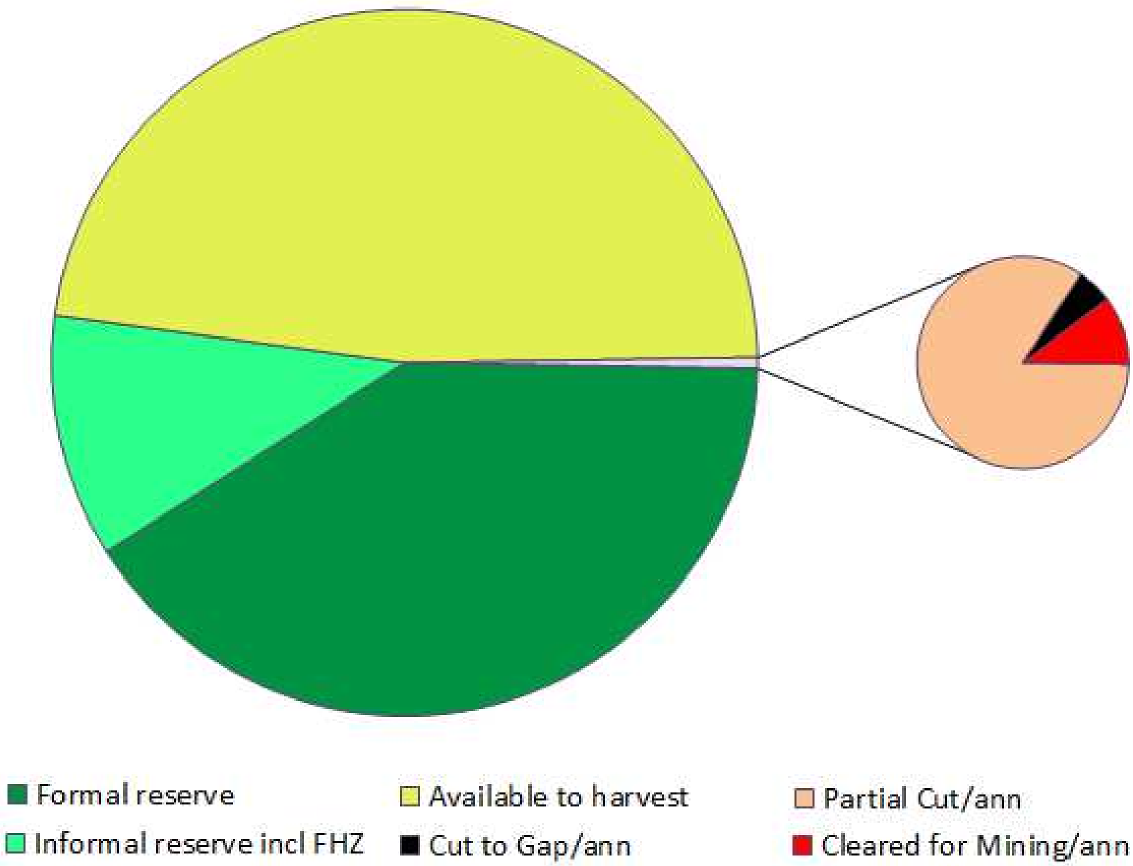


Figure 17: Left - the proportion of the jarrah forest by land use category. Right - the proportion of the annual harvest area in each harvesting category. Source: DEC Annual reports and Conservation Commission (2013b).

4.1 Conservation of biodiversity

Biological diversity (biodiversity) refers to the variability among living biological entities and the ecosystems and ecological complexes of which those entities are a part. It is measured or observed at three different levels: ecosystems, species and genes. Conserving biodiversity ensures that ecosystems remain productive and resilient to disturbance and to changes in the environment in which they exist.

Conservation of biodiversity is considered at several different scales within the whole forest. The goals and strategies adopted at each of these scales are designed to complement each other thereby strengthening conservation values of the whole.

4.1.1 Conservation strategies

Large formal reserves

Large formal reserves (national parks, nature reserves and conservation reserves), which are excluded from timber harvesting disturbance, are an important strategy in the conservation of biodiversity. These reserves have been selected to be large enough to support self-sustaining populations of native species and communities (RFA 1998a; Conservation Commission 2004). The intention is that these ecosystems will continue to function under the influence of natural or low level disturbances. These areas are managed according to formal management plans where available, and the FMP if no specific area management plan exists. These reserves are representative of the twelve forest ecosystems recognised in the jarrah forest and woodlands (Bradshaw *et al.* 1997b; RFA 1999).

Landscape scale management

The biodiversity goal for forest available for timber harvesting at the landscape scale is defined as *seeking to allow for the recovery of biodiversity between one timber rotation and the next (Conservation Commission 2012b).* To assist in achieving that, informal reserves and fauna habitat zones (FHZs) are dispersed throughout State forest (Figure 18). These reserves also contribute to conservation at the broader landscape level. Informal reserves consist of old-growth forest, river and stream zones, travel route zones, diverse ecotype zones (DEZ), less well reserved vegetation complexes, poorly reserved forest ecosystems and 'RFA accredited' linkage zones[4] FHZs are established to help in *maintaining both fauna populations within themselves, and to provide a source for the re-colonisation of nearby areas after timber harvesting*. These may be re-located in the future once disturbed areas are able to provide suitable habitat.

These areas are identified during the harvest planning process and separate guidelines exist for the management of these informal reserves and FHZs (Department of Environment and Conservation 2009).

[4] The areas identified in the Department's corporate database that provide a link between the proposed Milyeanup National Park and an adjacent stream zone, and a corridor between the Helena and Flynn parts of the proposed Helena Valley National Park.

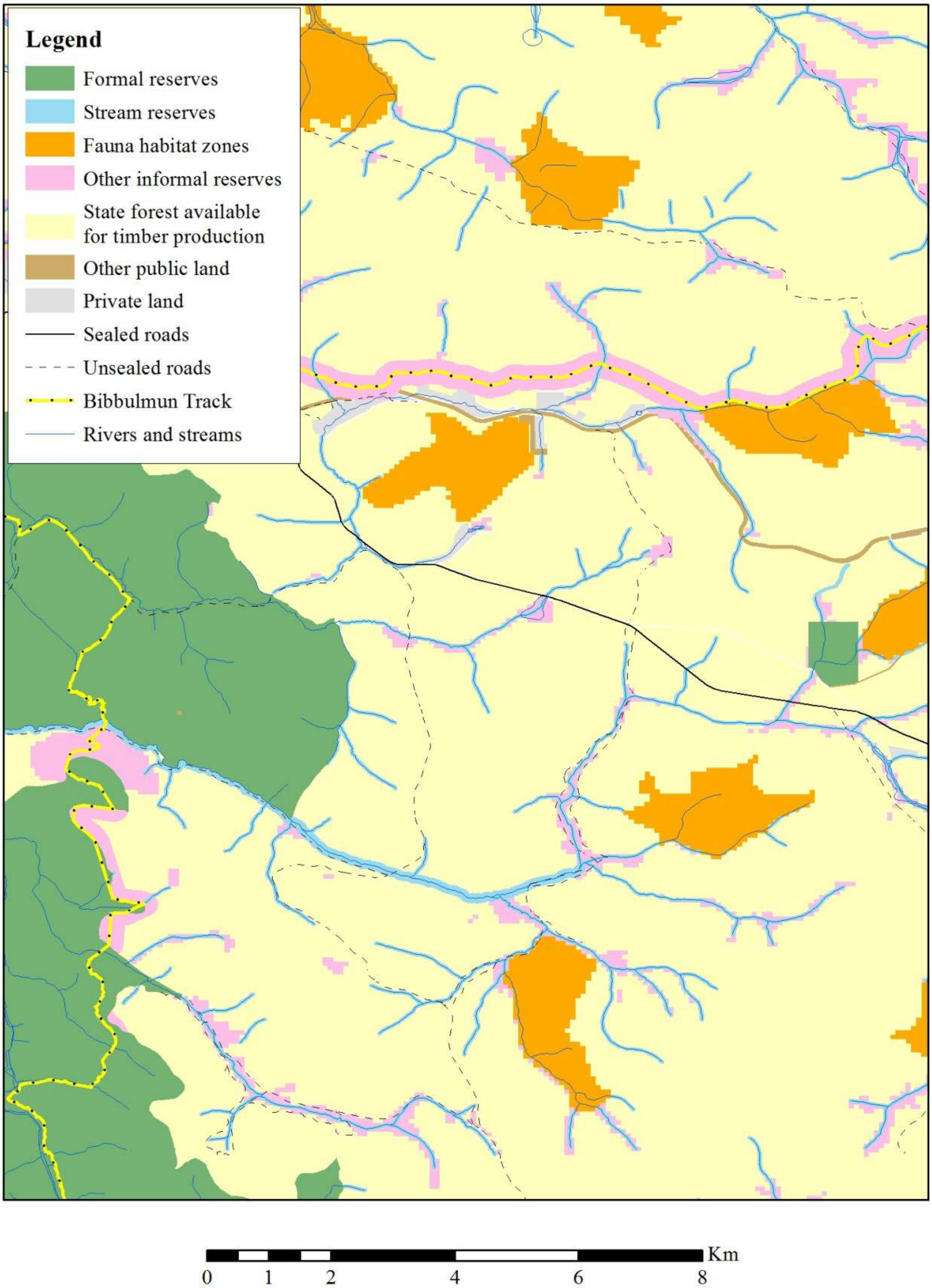


Figure 18: A section of forest showing the distribution of formal reserves, informal reserves (stream reserves and other types) and fauna habitat zones.

Stand level management

The application of silvicultural treatment at a patch scale (including the retention of habitat trees, CWD, understorey and second storey elements) is the principal focus of this material and the associated silvicultural guidelines. These silvicultural 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 the formal and informal reserves and are not intended as stand-alone strategies for biodiversity management.

4.1.2 Biodiversity of the jarrah forest

The jarrah forest supports a rich diversity of plants and animals. There are about 245 vertebrate species within the forest area comprising 29 mammals, 150 birds, 44 reptiles, 11 amphibians and 11 fish. The number of species in other groups may change where new species are described. The broad scale sampling of a range of jarrah forest ecosystems during the first five years of FORESTCHECK has identified 452 species of vascular flora, 1497 species of invertebrates, 450 species of macro-fungi and 315 species of cryptograms (Robinson *et al.* 2010).

The forest Fauna Distribution Information System (FDIS) identified 33 different habitat types for vertebrate fauna across the jarrah 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 per cent of species are in locations or vegetation types that are not affected by harvesting, instead occurring within the extensive formal and informal reserve system. FDIS contains a comprehensive list for vertebrate species in the south west forests, along with their probability of occurrence and their vulnerability to the effects of burning and harvesting. FDIS is used to analyse and predict the likelihood of fauna occurring in a particular habitat type, and to provide guidance on the measures required to reduce potential adverse effects of timber harvesting and burning on these species.

In WA, taxa are assigned a category to reflect their conservation status ((DiMatteo 2010; Department of Environment and Conservation 2012). Species considered threatened or as declared rare fauna are assigned the categories (from most to least serious) Critically Endangered (CR), Endangered (EN) or Vulnerable (VU). A priority species is a species that does not meet the criteria for listing as Threatened Fauna or Declared Rare Flora (e.g. due to lack of information, or not currently threatened), but which either may be suspected to be threatened; or is not threatened, but is rare and in need of ongoing monitoring; or is dependent on ongoing management intervention to prevent it from becoming threatened.

- Priority One (P1): Taxa with few, poorly known populations on threatened lands.
- Priority Two (P2): Taxa with few, poorly known populations, some of which occur on conservation lands.
- Priority Three (P3): Taxa with several, poorly known populations, some on conservation lands.

- Priority Four (P4): Taxa in need of monitoring.
- Priority Five (P5): Taxa that are conservation dependent (i.e. their conservation status is dependent on ongoing active management).

In the text below, the conservation status of taxa occurring in the jarrah forest is indicated after the species name. Where no category is listed, there is currently no concern around the conservations status of the species.

Vascular plants

Within the general forest area[5], 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 the rare and 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). Centres of endemism, relictual and disjunct flora are similarly located.

Where permanent disturbances such as roads, mines, or basic raw material pits are proposed, then the operation is preceded by a flora survey of the site to identify declared rare and priority flora and Threatened Ecological Communities (TEC) and Priority Ecological Communities (PEC). In the case of timber harvesting, the operation 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 (TEC) and (PEC) are required to be excluded from 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 Department's *Planning checklist for disturbance activities*.

Fungi

The jarrah forest has a rich fungal flora despite an apparently hostile environment. A high proportion of these species are mycorrhizal, the diversity of which increases with associated tree age. While fire and weather conditions play an important role in species diversity and abundance, the relationships and significance for nutrient cycling and nutrient uptake are complex and not well understood. Information on the fungi of the jarrah forest have been summarised by Hilton *et al* (1989).

Cryptograms

Cryptograms comprise a number of different life forms including lichens, mosses, and liverworts. Recent studies during the first five years of FORESTCHECK (Robinson *et al.* 2010), have found 280 species of lichens, 27 species of moss, and eight species of liverwort in the plots studied. It is possible that this list may increase as further studies of other forest ecosystems are conducted, and as work progresses to identify and categorise all species collected from the existing studies.

[5] This includes jarrah and karri forests and the associated vegetation within the forested landscape.

Mammals

The jarrah forest supports approximately 29 native mammals including arboreal marsupials, terrestrial mammals and bats which occur across a range of habitats.

The six species of arboreal marsupial species are: the brush-tailed phascogale (*Phascogale tapoatafa*) VU, red-tailed phascogale (*Phascogale calura*) EN, western ringtail possum (*Pseudocheirus occidentalis*) VU, common brushtail possum (*Trichosurus vulpecula*), western pygmy possum (*Cercartetus concinnus*) and honey possum (*Tarsipes rostratus*). All rely on hollows in mature trees for breeding.

The jarrah forest supports approximately 14 terrestrial native mammals and most of these species are widespread across the jarrah forest. Five species require hollow logs for refuges or breeding, these are: the chuditch (*Dasyurus geoffroii*) VU, Gilbert's dunnart (*Sminthopsis gilbert*), grey-bellied dunnart (*Sminthopsis griseoventer*), numbat (*Myrmecobius fasciatus*) VU and yellow-footed antechinus (*Antechinus flavipes*). The quokka (*Setonix brachyurus*) VU, and tammar wallaby (*Macropus eugenii*) P5 prefer habitats where the understorey species make a thick wall of vegetation generally along creek lines and swamps. The southern brown bandicoot (*Isoodon obesulus fusciventer*) P5, also prefers dense cover, although this species is not as dependent on riparian vegetation.

There are a number of species which prefer sparse cover, and the species in this group are all relatively widespread and common and generally do not require any special treatment, apart from the woylie (*Bettongia penicillata*) CR which continues to be under population pressure from a number of threats including loss of habitat from clearing and predation (despite ongoing introduced predator control efforts).

Nine species of bat occur in the jarrah forest, including the white-striped freetail bat (*Tadarida australis*), lesser long–eared bat (*Nyctophilus geoffreyi*), south-western freetail bat (*Mormopterus sp.*), chocolate wattled bat (*Chalinolobus morio*), Gould's wattled bat (*Chalinolobus gouldii*), Gould's long-eared bat (*Nyctophilus gouldi*), western long–eared Bat (*Nyctophilus sp.*), southern forest bat (*Vespadelus regulus)* and the western false pipistrelle (*Falsistrellus mackenziei)* P4*.* 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 jarrah forest supports approximately 150 bird species that occupy the entire stratum of the forest (Dell *et al.* 1989a). These bird species 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 150 species, sustainable harvesting and jarrah silviculture may affect 37 species (Christensen *et al.* 2005). These are listed in Appendix 2. Of the 37 species, 26 rely solely on tree hollows for nesting sites. Four species listed as endangered or vulnerable rely on tree hollows for nesting. They are the red-tailed black-cockatoo (*Calyptorhynchus banksii naso*) VU, Carnaby's cockatoo (*Calyptorhynchus* *latirostris*) EN, Baudin's cockatoo (*Calyptorhynchus baudinii*) EN and Muir's corella (*Cacatua pastinator pastinator*) SP. The masked owl (*Tyto novaehollandiae novaehollandiae*) P3, barking owl (*Ninox connivens*) P2 and the specially protected peregrine falcon (*Falco peregrinus*), also rely on tree hollows for nesting.

There are 11 other bird species found in the jarrah forest which may be affected by harvesting but do not rely on hollows. These species are generally birds of the reed beds and swamps including the Australasian bittern (*Botaurus poiciloptilus*) EN, black bittern (*Ixobrychus flavicollis*) P3, and the little bittern (*Ixobrychus minutus*) P4. Habitats for these species are generally located in informal reserves and therefore excluded from timber harvesting.

Reptiles

The jarrah forest supports approximately 54 species of reptiles, comprising four species of dragons and monitors, seven species of gecko, five species of legless lizards, 21 species of skinks and 17 species of snakes. A number of these species are listed, including: the Darling Range ctenotus (Ctenotus delli) P4, jewelled south-west ctenotus (*Ctenotus gemmula*) P3, south-western mulch skink (*Glaphyromorphus australis* [*G. "koontoolassi"*]) P1, short-nosed snake (*Elapognathus minor*) P2, and the southern death adder (*Acanthophis antarcticus*) P3, the woma (*Aspidites ramsayi*) and the carpet python (*Morelia spilota imbricata*) are specially protected species. Five species, the heath monitor (*Varanus rosenbergi*), western bearded dragon (*Pogona minor*), bobtail (*Tiliqua rugosa*), woma (*Aspidites ramsayi*) and the carpet python (*Morelia spilota imbricata*) have been identified in FDIS as species to monitor with respect to fire.

Amphibians

The jarrah forest supports 20 species of frogs. Of these species, there are four species that are listed as endangered or vulnerable: orange-bellied frog (*Geocrinia vitellina*) VU, white-bellied frog (*Geocrinia alba*) CR, sunset frog (*Spicospina flammocaerulea*) VU and the Nornalup frog (*Geocrinia lutea*) P4. Based on FDIS, it is possible that all could be adversely affected by timber harvesting activities or prescribed fire (Christensen *et al.* 2005).

Fish

The jarrah forest supports eight species of fish. Of these species, there are two species that are listed as endangered or vulnerable: Balston's pygmy perch (*Nannatherina balstoni*) VU and mud minnow (*Galaxiella munda*) VU. There are two priority species, the black-striped minnow (*Galaxiella nigrostriata*) P3 and the pouched lamprey (*Geotria australis*) P1. Based on FDIS, it is expected that none of these species will be adversely affected by timber harvesting activities or prescribed burning (Christensen *et al.* 2005).

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 brought about 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 (or non-declining) yield of sawlogs has been seen as a surrogate for the sustainability of other values, since sustaining sawlog values also requires the sustainability of age structure. While timber yield was aimed at producing larger sized logs (coming from old, large trees) and management was less intense, this assumption was valid to a large degree. However, as management becomes more intense, and goals more diverse, the protection of other values that may have been provided for by default, must be considered more explicitly. Structural goals provide a more direct means of addressing the issue of structural diversity and a more direct means of evaluating the sustainability of a range of values. Whole of forest and landscape level structural goals have been reviewed by Bradshaw (2002) and Burrows *et al.* (2002a). 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 as of sawlogs as the primary regulator of sustainability.

Structural diversity 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 effect of different management options not only on structure but also on biodiversity. It is possible to develop a range of metrics appropriate to structural diversity 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 19. For this illustration, the details of the development stages are not included; the emphasis is on the dynamics of change.

Landscape scale

While the principal contributor to landscape heterogeneity in the area available for timber harvesting is the widely dispersed informal reserves and FHZs, these are supplemented by the sequence and intensity of timber harvesting.

Phased timber harvesting and regeneration was originally introduced into silvicultural practice to limit the size of gaps created mainly for aesthetic purposes. It is the practice of temporarily deferring timber harvesting in portions of a coupe, and specifying a return period (less than a rotation) for subsequent disturbance. It was also implemented to limit the degree of canopy reduction at any one time in second-order catchments with high groundwater salinity to avoid the discharge of saline water. This requirement has subsequently been removed except for some catchments in southern forests and other specific salt risk catchments, because of reduced risk of saline groundwater discharge as a result of the more recent extended period of below average rainfall (Conservation Commission 2013a).

Phased regeneration spreads disturbance over time and increases structural diversity at the local management scale and is important for wildlife habitat (Wayne *et al.* 2001; Webela *et al.* 2010; Farr *et al.* 2011; Robinson *et al.* 2011). The areas retained are called temporary exclusion areas (TEAS).

The jarrah forest is essentially an uneven-aged forest with patches of even-aged cohorts of varying size and integrity occurring as a complex mosaic. The continued maintenance of this structure also contributes to structural diversity at the local scale. Silvicultural practices aimed at satisfying a range of goals do not emulate natural disturbance regimes in every respect, but they aim to operate within a range that can occur naturally and which has resulted in satisfactory conservation outcomes in the past. The extent to which current silvicultural practices conform to the principles of what has become known as 'ecological forestry' has been evaluated by Stoneman (2007).

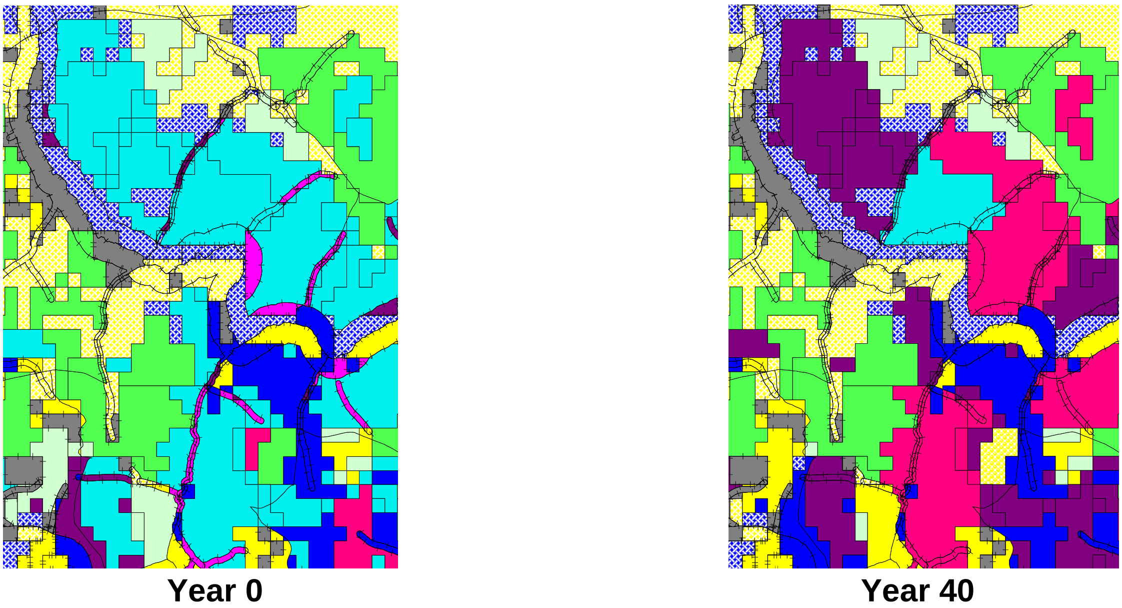


Figure 19: An illustration of the changing spatial relationship between stands at different stages of development from establishment, juvenile, immature, mature and senescent, represented by different colours in a mixed jarrah / karri landscape over a 40-year period. Within-patch structural variation adds to this complexity. Overlain on this changing pattern is the constantly changing age of the understorey associated with time since the last burn.

Local scale

There are a number of structural elements at the local scale that contribute to the maintenance of biodiversity.

Overstorey trees with hollows

Large trees, particularly those with hollows, are a key structural element of jarrah forest. Six mammal species and 34 species of birds and reptiles use hollows in standing trees (Abbott *et al.* 2002; Christensen *et al.* 2005).

Hollow occurrence is most likely in trees with moderately senescent crowns with damage to intermediate sized branches, with the largest hollows occurring in highly senescent crowns. Although jarrah trees bear more hollows than marri trees, the hollows in jarrah are typically, significantly smaller than those in marri. The typical minimum age for the formation of useable hollows in jarrah and marri is 130 years, which equates to trees approximately 50cm dbhob. Large hollows, used by species such as the forest red-tailed black cockatoo are more likely to be found in senescent trees above 70cm dbhob (Whitford *et al.* 2001a). The number of hollows in individual trees increases with tree diameter, however due to the tree size class distribution in the jarrah forest having a negative exponential shape (Figure 13, Figure 14), trees with diameters between 40cm and 80cm contribute approximately 50 per cent of all hollows in the jarrah forest.

Retention of trees likely to contain hollows in areas subject to silvicultural treatment ensures that this important structural element persists to provide refuge and nest sites for a variety of hollow using animals.

There are limited studies on the number of hollows required by individual species. In a study of western ringtail possum (*Pseudocheirus occidentalis*), Wayne *et al.* (2000) recommended the retention of five habitat trees per hectare in areas where they were known to occur, together with the retention of other habitat elements such as balga (*Xanthorrhoea preissii*), riparian vegetation and patches of other undisturbed vegetation. Western ringtail possum is one of the species thought most vulnerable to anthropogenic disturbance (Wayne *et al.* 2006).

There are advantages and disadvantages for fauna in the grouping or dispersal of habitat trees, with grouped trees offering more secure refuge for individuals but there being greater potential for territorial conflict.

Although there is a slow rate of natural fall of habitat trees, a proportion of existing habitat trees will die of old age over the next 50-100 years and few are likely to live beyond 300 years. Young regeneration will be unable to fulfil the role of habitat trees for as much as 130 years while suitable hollows develop. Replacement habitat trees, called secondary habitat trees, should be of intermediate age (about 30-60 years old), and be relatively healthy individuals from the dominant or co-dominant layer with a long life expectancy.

Abbott and Whitford (2002) analysed the risk to species of a loss of tree hollows based on their reliance on hollows, their home range, the occurrence of hollows of suitable size, and their dependency on public-owned forest. They concluded that the species most at risk were:

- rufous treecreeper *(Climacteris rufa)*
- common brushtail possum*(Trichosurus vulpecula)*
- red-tailed phascogale *(Phascogale tapoatafa)*

- sacred kingfisher, *(Todiramphus sanctus)*
- red-capped parrot *(Platycercus spurious)*
- western rosella *(Platycercus icterotis)*
- Baudin's cockatoo *(Calyptorhynchus baudinii)*
- forest red-tailed black cockatoo *(Calyptorhynchus banksii naso).*

However, when predation and territorial issues are included, the western ringtail possum can be added to the list (Wayne *et al.* 2006).

Hollow logs

Hollow logs are used as den and refuge sites for fauna such as chuditch (*Dasyurus geoffroii)*, common brushtail possum (*Trichosurus vulpecula),* quenda (*Isoodon obesulus*) and numbat (*Myrmecobius fasciatus)* (Christensen *et al.* 1984; Serena *et al.* 1989; Wayne *et al.* 2001).

In the jarrah forest there are a significant number of logs on the forest floor in most situations. The proportion of those logs with hollows is often relatively small, so retaining hollow logs and the retention of logs of suitable dimension is an important aspect of maintaining biodiversity. Williams and Faunt (1997) found that about 25 per cent of logs larger than 70cm contained hollows, but only one per cent of all logs encountered contained hollows large enough for chuditch, the largest hollow nesting mammal. Fire is an important element in hollow formation in standing trees (Inions *et al.* 1989; Williams *et al.* 1997). Logs are more likely to be burnt away in autumn than is spring fires, since in spring, log moisture content is higher.

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; Wardlaw *et al.* 2009; Gates *et al.* 2011). Recent FORESTCHECK surveys have quantified the nature of CWD in various silvicultural treatments and provided information on its importance for the diversity of fungi and cryptogams (Hollis *et al.* 2009; Robinson *et al.* 2010; Whitford *et al.* 2010; Cranfield *et al.* 2011).

Timber harvesting has the capacity to add CWD to the site, in a range of sizes (Figure 20). However, harvesting for non-sawlog material has the potential to reduce the volume of CWD and may need to be constrained to prevent undesirable loss. The abundance of logs (McCaw 2011) can also be adversely affected by heaping and burning. Current silvicultural guidelines require the exclusion of some logs from heaping and burning operations, the retention of some large logs and all logs with suitable hollows logs (pipe 6-15cm and at least 1.5m long), in jarrah and wandoo forest harvesting operations.

Diameter class midpoint (cm)

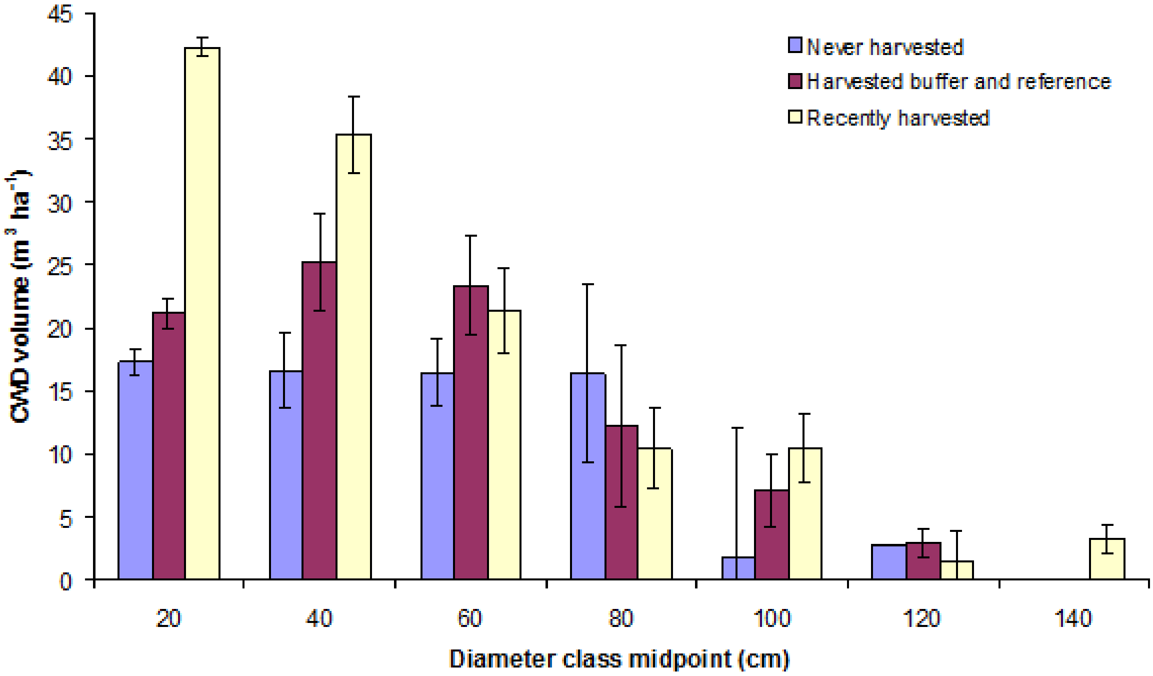


Figure 20: The volume of CWD in virgin forest and cutover forest from 48 FORESTCHECK sites spread across the jarrah forest. Most increases due to timber harvesting are in the smaller size classes (Whitford et al. 2010).

Mid-storey trees

The jarrah forest vegetation complexes include a mid-storey of smaller trees, though the stocking of these is variable throughout. It is desirable to ensure that where a mid-storey exists, that a representation of these elements is retained to maintain the structural and compositional diversity of the forest at the local scale. Some of the longer-lived mid-storey species of the jarrah forest include:

- balga (*Xanthorrhoea preissii*)
- kingia (*Kingia australis*)
- bull banksia (*Banksia grandis*)
- sheoak (*Allocasuarina fraseriana*)
- woody pear (*Xylomelum occidentale*)
- river banksia (*Banksia seminuda*)
- snottygobble (*Persoonia longifolia* or *Persoonia elliptica)*
- WA peppermint (*Agonis flexuosa*) and
- WA Christmas tree (*Nuytsia floribunda*).

Mechanical disturbance of the vegetation and soil occurs incidentally during harvest and other disturbance operations, but may also be undertaken after timber harvesting to reduce competition to jarrah advance growth or to enable seedlings to establish to create a seedbed, and reduce competition for new seedlings. A balance between disturbance and protection is required (Burrows *et al.* 2002a; Burrows *et al.* 2002b). Scattered individuals of these second-storey species should be retained. However, thickets of sheoak and banksias can prevent the release of advance growth or the establishment of new regeneration and their density needs to be reduced to allow the opportunity for new tree regeneration to establish. Biodiversity can be managed by removing thickets only where jarrah or marri exists or previously existed, by retaining scattered individuals of mid-storey species, and retaining a proportion of the thickets as small clumps.

The dry ‘skirt’ of balga is used by the western ringtail possum, but not the common brushtail possum, and is particularly valuable where the more aggressive common brushtail possum competes with western ringtail possum for other refuges. The retention of groups of balga in association with habitat trees is preferred to the retention of scattered individuals.

Understorey and soil stored seed is protected by avoiding unnecessary soil disturbance. Where disturbance is required to establish seedlings, (e.g. *Taxandria* spp. thickets in southern forest) it should be limited to that which is necessary to achieve the regeneration standards.

Overstorey species composition

Where timber harvesting takes place with the intent of releasing established lignotubers, little can be done to influence the species composition of the next generation since this is already established in the composition of the lignotuber pool (Section 2.4.). However, where there is an under-representation of jarrah in the lignotuber pool, it is possible to supplement the new regeneration by encouraging the coppice development from small jarrah stumps.

Where regeneration is being established from seed, the composition of the seed source can be used to influence the species composition of the future lignotuber pool. While it is unrealistic to expect to be able to recreate the original mature overstorey composition (even if this were a desirable aim), it is important to ensure that the original species are adequately represented in the new regeneration.

Where thinning is involved, species representation can be assured by the retention of an appropriate representation of species in the retained trees.

A situation that occurs in significant parts of the jarrah forest is the predominance of *Allocasuarina* in stands that were originally dominated by jarrah. This is sometimes the result of past timber harvesting that removed jarrah when insufficient ground coppice was available to regenerate the stand. *Allocasuarina* became established instead, and dominated the site to the exclusion of any further opportunity for jarrah regeneration. Many fire events have occurred in these sites since that time without resulting in the establishment of new regeneration, indicating that these stands may not be restored to their original composition without further intervention. This could require the physical removal of some of the *Allocasuarina* followed by deliberate actions to regenerate jarrah. A similar situation where *Banksia grandis* has supplanted jarrah is found less frequently.

4.1.4 Genetic conservation

Jarrah and understorey species

While most jarrah and understorey species regeneration is derived from natural means with the seed source available *in situ, a* small proportion of regeneration is carried out by artificial seeding or planted nursery stock. While genetic differentiation throughout the jarrah forest is low, three sub-species have been recognised (as explained in 2.1 Taxonomy), although this separation is not supported by genetic analysis (Wheeler *et al.* 2003). Provenance trials conducted for mine-site rehabilitation by Alcoa showed little difference in performance by families collected from the main range of jarrah, with the exception of poorer performance from coastal plain families (Wheeler *et al.* 2006). However, there is a trend of reduced survival of seedlings from southern forest when planted in the north, suggesting that there may be some adaptation to different climatic conditions (Alcoa unpublished, cited in Wheeler *et al.* 2003).

Traditionally, regeneration operations requiring the use of supplementary seed or seedlings have strived to use 'local' seed. Where knowledge of the population genetic structure of a species exists or can be reasonably inferred, this should guide seed collection areas. Recently, guidelines for seed collection for regeneration (and rehabilitation) have moved away from the requirement for only using 'local' material, as the scientific basis for this has been increasingly questioned, and additional considerations for optimal regeneration outcomes are now recognised. 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. Resistance to *Phytophthora cinnamomi* should be considered in the choice of seed origin for specific areas and 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 Fire and biodiversity

A goal of the FMP is to seek *to use and respond to fire in a manner that promotes the maintenance of ecosystem health and vitality, the conservation of biodiversity, and mitigates the risk of adverse impacts of bushfire.* 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 fire frequency and season, and at the local level by ensuring patchiness and spatial distribution of prescribed burns (CALM 2000; Adams *et al*. 2003).

The negative exponential relationship between the proportion of the landscape affected has been shown to be the most stable model of age distribution in naturally occurring fire-prone environments (Burrows 2008). This distribution also provides conditions for maximum species richness while maintaining the opportunity to manage species that require long periods between fires. The most common 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 area in relation to time-since-fire for the whole jarrah forest is shown in Figure 29.

4.1.6 Mining and biodiversity

Mining for bauxite has a significant effect on biodiversity through disturbance created by removing all vegetation from the site and a substantial part of the soil profile. Following mining, the sites are now rehabilitated with native species. While considerable success has been achieved in the restoration of understorey species (Grant *et al.* 2007; Koch 2007), structural complexity will not be restored for many decades or longer. Current rehabilitation practice is predicted to have a negative impact on a number of species in riparian zones because of reduced streamflow resulting from the high LAI of rehabilitation (Dundas *et al.* 2012). More than 15,000ha of forest has been cleared for bauxite mining to 2013 and 45 per cent of State forest and timber reserves are currently under bauxite mining leases, with another 37 per cent under pending mining leases. It is estimated that 83,000ha may be cleared in the long term, fragmenting about 337,000ha of jarrah forest (Conservation Commission 2012a). Most of this will be in the northern jarrah forest.

4.1.7 Climate Change

Changes to the amount and seasonality of rainfall will increase moisture stress and may affect health and species composition in some environments, with the most pronounced effects expected in areas of low rainfall and shallow soils.

An analysis of potential impact under the CSIRO high-severity climate scenario (CSIRO 2007) identified the nine LMUs at most serious risk (Maher *et al.* 2010). These are LMUs that would be outside the rainfall and evaporation threshold for jarrah forest (Gentilli 1989; CSIRO 2007; Maher *et al.* 2010). (Croton *et al.* 2015) predict reductions in LAI across the jarrah forest, particularly in areas with reducing rainfall and/or increasing evapotranspiration. 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 as well as biota on sites with shallower soils. There is already evidence that some perennial streams have become seasonal in recent years and the period of flow has decreased in many seasonal streams. Reduced abundance of some species associated with these changes has been observed (Storey *et al.* 2010; Pennifold *et al.* 2011).

Declines in streamflow have been proportionally greater than declines in rainfall in south-western WA. This may be due to groundwater becoming disconnected from surface water (Hughes *et al.* 2012; Kinal *et al.* 2012). The implication is that where disconnection has occurred, the capacity to generate streamflow is greatly reduced. Where disconnection has occurred, groundwater would need to be recharged to the point of connection again before significant improvements in streamflow would be realised. The greater the delay in responding to declining groundwater levels, the less likely it will be that streamflow can be improved in the short term. Hydrologic process modelling (Croton *et al.* 2001a; Croton *et al.* 2001b) calibrated with the measured flow and rainfall data obtained from the gauging network, provides an understanding of the changing water balance in the northern jarrah forest.

The concept of using silviculture to manage vegetation water use and thus water yield in forested catchments ('silviculture for water production') has been developed from research undertaken locally (Ritson *et al.* 1981; Stoneman 1986; Marshall *et al.* 1992; Stoneman *et al.* 1996; Bari *et al.* 2003; CSIRO 2009), nationally (Vertessy 1998; Erskine 2004; Feikema *et al.* 2006) and internationally (Douglas, 1983; Stednick 1996; Calder 1998; Kaye *et al.* 1999; Brown *et al.* 2004). There is potential to apply 'silviculture for water production' to increase water for consumption or irrigation purposes as well as the maintenance of riparian biodiversity.

Partially (thinning) or completely removing (gap creation) trees from an area will reduce interception of rainfall allowing additional infiltration and reducing transpiration, resulting in increased soil moisture and in some cases, a rise in groundwater and potentially, streamflow. However, the increases from a single treatment are transient and water yields can quickly decline to pre-treatment levels as the remaining trees and / or regeneration from stump and ground coppice grow. Control of regeneration and stump coppice development has been shown to prolong the increase in water yield for up to 25 years (Stoneman 1993). Unthinned even-aged stands of juvenile and immature regrowth have been demonstrated to use more water than more mature forest development stages. Within a catchment, controlling the amount of younger growth stages may also be required to manage water availability.

A series of 27 small catchment studies conducted in south-west WA during the 1980s and 1990s examined the effects of a range of vegetation removal activities on water yield (Bari *et al.* 2003). Permanently clearing vegetation (for example, for agriculture) resulted in sustained water yield increases of 20 to 30 per cent, depending on average rainfall, with a greater increase in higher rainfall areas. Forest thinning of high rainfall catchments increased water yield by a maximum of 8 to 18 per cent. The increase was dependent upon the characteristics of the catchment and the amount of vegetation removed. However, the increases from thinning were not permanent and water yields returned to pre-thinning levels after 12-15 years. Control of regeneration and stump coppice development has been shown to prolong the increase in water yield for up to 25 years (Stoneman 1993; Kinal pers. comm.).The effect of thinning to alleviate this problem on biodiversity has been investigated as part of the Wungong project (Water Corporation 2010). A vegetation survey of plots maintained at a wide range of overstorey stand density for 45 years (Inglehope thinning plots) showed no difference in understorey species richness, and while some species appeared to be favoured at low density and some at high density, most species were unaffected (Mattiske Consulting Pty Ltd 2004).

4.2 Maintenance of productive capacity

Under the FMP, the principal timber production goal for the jarrah forest is the sustained yield of sawlogs. It is defined as the yield of first and second grade sawlogs that the forest can produce for an extended period (to at least the year 2070) at a given intensity of management (Conservation Commission 2013a; Ferguson *et al.* 2013), given a certain area of forest available for harvesting. Sustaining the yield in an absolute sense has always been seen as secondary to the achievement of other conservation goals. The yield of other log types is that produced in the course of producing sawlogs.

The principal source of sawlog yield for the next several decades will be from the mature trees in stands that were selectively cut in the past. Regrowth originating from the 1920s to the 1940s becomes a major source of sawlogs for a further 40 years from the start of the FMP. Stands regenerated today will not contribute significant volumes of sawlogs until about 2130.

4.2.1 Factors influencing silvicultural practice

A number of factors influence the silvicultural practice that may be applied to different stands or patches within a stand subject to timber harvesting.

The influence of stand density on growth

Stand density is a critical factor influencing both stand growth and the growth of individual trees. The growth response for a first thinning of jarrah at different ages and densities is illustrated in Figure 21. The data show that:

- at low density, stands are effectively ‘free-growing’ and stand growth (expressed here in terms of basal area) increases with density until the trees begin to compete intensively with each other at ‘critical density’
- the point of ‘critical density’ increases with stand age
- in young stands, total stand growth continues to increase with density beyond the ‘critical density’ until it reaches the maximum for the site at that age
- older stands maintain the same growth over a wide range of stand densities, but growth reduces above ‘suppression density’
- maximum density increases with stand age while maximum stand growth reduces with stand age.

These follow the general relationships common to all forest species (Assmann 1970; Smith 1986). ‘Critical density’ approximates the density of the dominant and co-dominant trees in the stand.

While leaf area (expressed as LAI) increases with stand density (Figure 22), the increase beyond ‘critical density’ is relatively small, and leaves become less efficient at higher density. The increased growth efficiency (growth per unit of leaf area) of the leaves at lower density explains the maintenance of stand growth at lower density, at least until the stand density is reduced below ‘critical density’; from below this point the leaf area is too low to maintain stand growth despite continuing increases in efficiency.

Thinning 40-year-old stands from 1,200 spha down to 200 spha (‘critical density’) results in a 100 per cent increase in the diameter growth of the 200 fastest stems and no loss of total stand growth. The response in younger stands is much less dramatic; to achieve a similar growth response would require thinning to much less than ‘critical density’, with a consequent loss in stand growth, though the loss of stand growth at this stage may be of no consequence to future sawlog production (Stoneman *et al.* 1989b).

The second thinning of a jarrah pole stand, at age 62 (22 years after the previous thinning) produced a similar growth response to the first thinning, with optimum stand density at about 50 per cent of maximum density (Stoneman *et al.* 1996). This is contrary to the response reported for other species by Assmann (1970), where the increase in stand growth was principally a response to the first thinning where stands were released from suppression density. For those species, the ‘critical’ and optimum density increased slightly with the second thinning, while there was a reduction in the range of optimum density (i.e. in the range of density with >90 per cent of maximum growth).

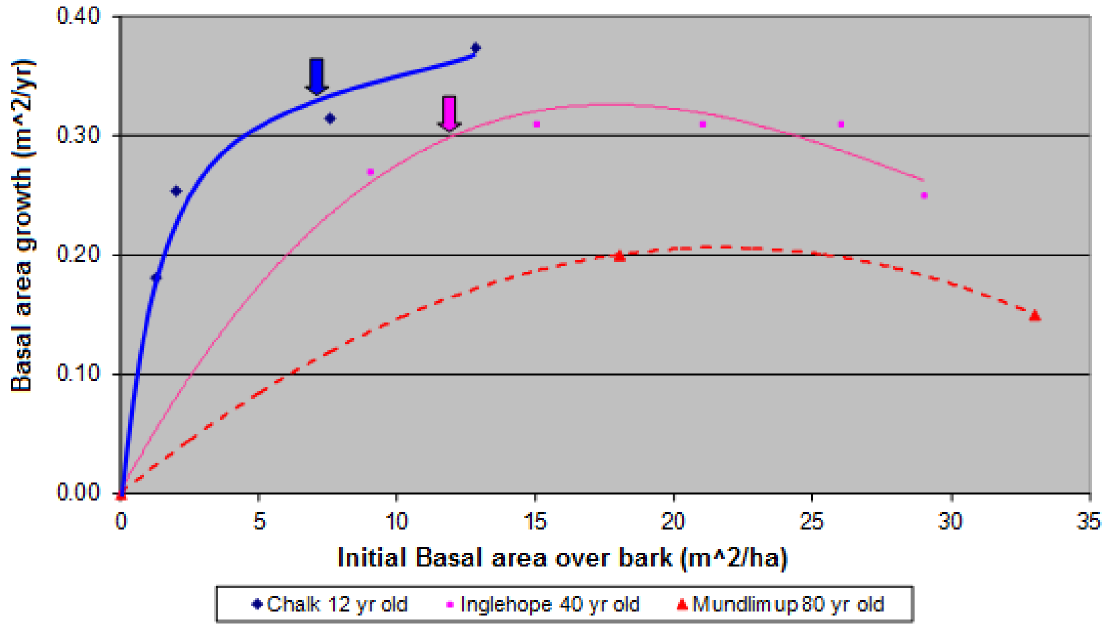


Figure 21: Basal area growth at different age and density in high to moderate rainfall jarrah stands. Based on data derived from the first thinning of even-aged stands at 12 year old (Chalk plots), 40 year old (Inglehope plots) and 80 year old (Mundlimup plots) stands. The notional form of the Mundlimup response is based the generalised relationship (after (Langsaeter 1941)) fitted to the limited data points of a thinned and an unthinned plot using the same form of equation as derived for the Inglehope plots. The critical density at each age is highlighted with arrows. Data points for the Chalk and Inglehope plots are treatment means(Abbott et al. 1986; Stoneman et al. 1989b)

Nevertheless, it is expected that thinning may need to be more conservative as the stand ages, if maximum growth is to be maintained. As the trees in a stand become larger, the removal of a single tree leaves a relatively large space. More time is required for surrounding trees to occupy the site vacated by the removed trees. The immediate reduction in stand growth that results from the removal of some of the trees will take longer to compensate as the remaining trees respond, and may result in a net reduction in growth if the thinning is too severe. Eventually as the trees reach maturity, they will reach the stage when they can no longer continue extending to occupy the site and they have effectively become free-growing. At this point, thinning is no longer appropriate and the development of regeneration is more appropriate.

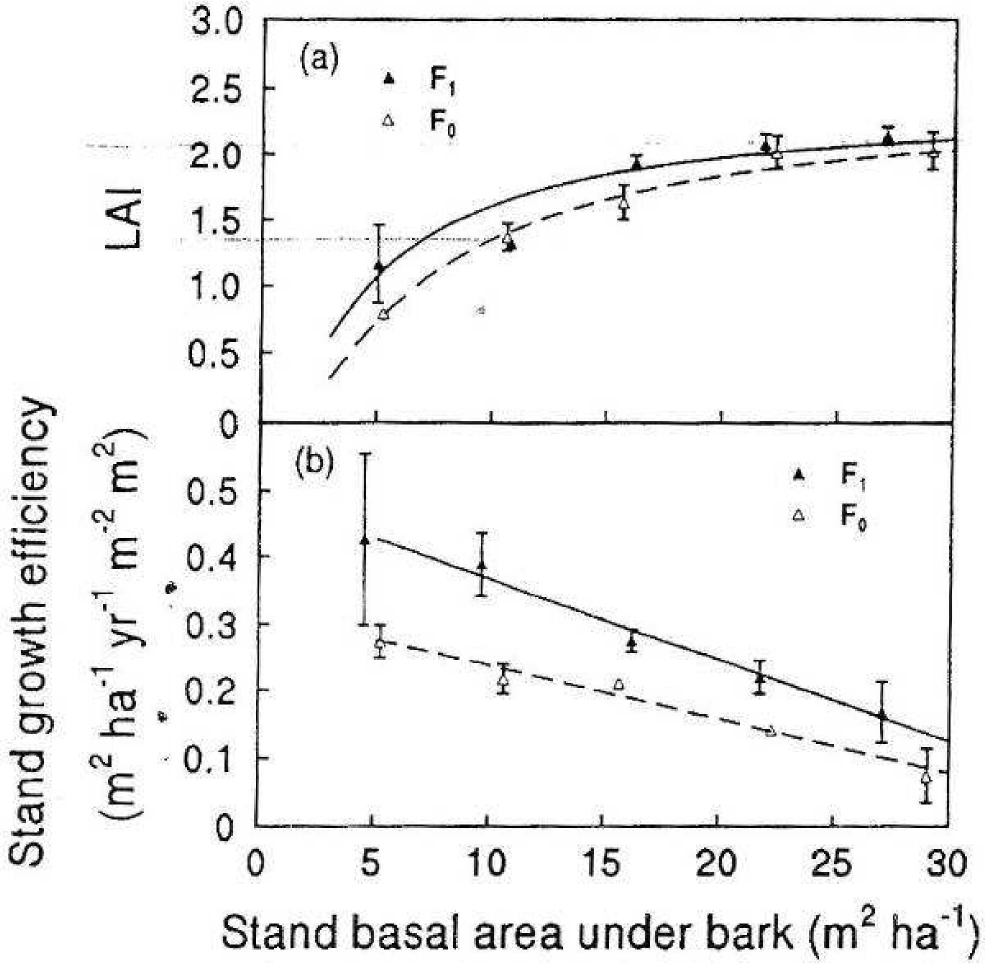


Figure 22: (a) Stand (LAI) for fertilised (F_1) and unfertilised (F_0) treatments in relation to stand basal area in a 60 year old jarrah regrowth stand. (b) Growth efficiency (basal area growth per unit of leaf area) in relation to stand basal area. (after (Stoneman et al. 1996).

Tree attributes for predicting growth

The strongest predictor of future growth of an individual tree is crown depth, crown width and diameter (Figure 23). Diameter relative to other members of the same cohort is a reflection of past growth rate. Crown density is a poor predictor of future growth, because of the variation in leaf density during the flowering cycle (Stoneman *et al.* 1989b).

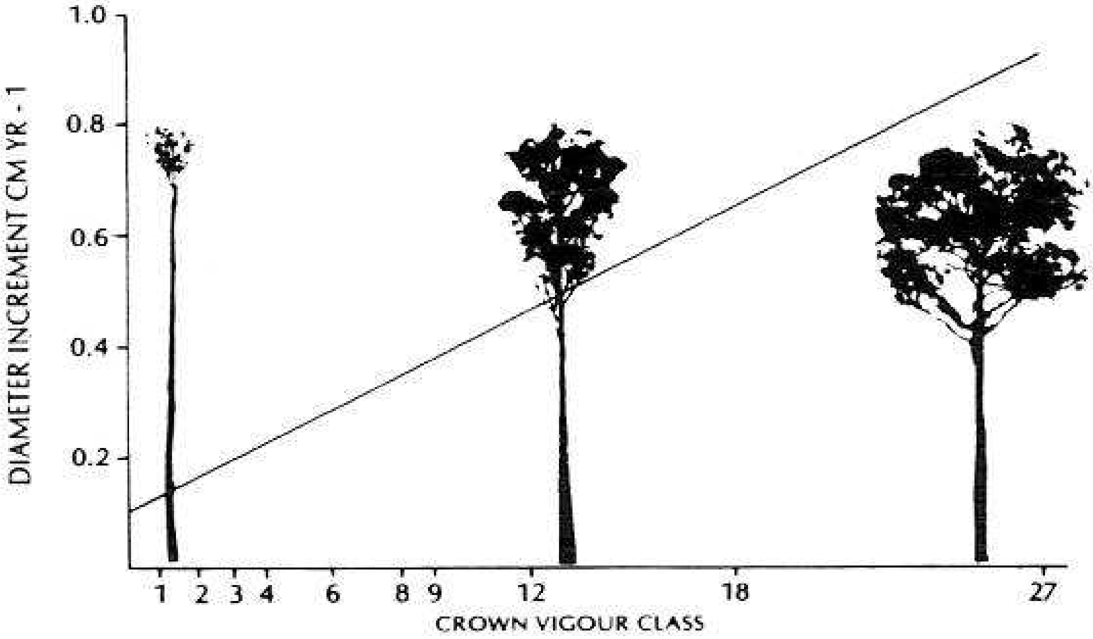


Figure 23: Diameter increment is closely correlated with crown characteristics – expressed as the product of three classes of crown width, crown depth and crown density (after (Loneragan 1956)).

Influence of rainfall on site potential

It is important to recognise that different parts of the forest have different capacities and to reflect this in silvicultural practice.

For silvicultural purposes, the jarrah forest is divided into west and east on the basis of rainfall and site potential. Eastern jarrah is characterised by lower dominant height, lower density of trees and more open understorey. The growth rate of eastern jarrah forest is less than that of the higher rainfall western forest and it follows that 'critical' and optimum density occurs at a lower basal area in the eastern jarrah forest. At maximum density, the basal area growth of stands receiving less than 1,000mm / annum of (historical) rainfall is about one quarter of that with a rainfall greater than 1,000mm / annum. At optimum density, the low rainfall growth rate is about two-thirds that of the high rainfall zone (Hong Yan, pers. comm.[6]). This suggests that growth is more responsive to reduced stand density in the low rainfall areas than in high rainfall areas, presumably due to increased moisture stress in low rainfall sites (Crombie 1997).

The lignotuber pool is generally less abundant and less uniform in the eastern forest. The establishment of regeneration is sometimes more difficult and may take longer because of the harsher and more variable climatic conditions. Despite the lower density of overstorey and understorey, competition for moisture is more intense than in the western forest.

[6] Forest Management Branch, Department of Environment and Conservation.

The boundary between the two zones is not precise but generally follows the historical 900mm rainfall isohyet. It is further defined by vegetation systems or complexes, but these too are soft boundaries and the application of different silvicultural practices should be informed by local field observation.

Vegetation types applying to the eastern forest are:

- Ecological Vegetation System Vp1, Vp2, Ip3, Jp2, Jp3 and associated mixed jarrah / wandoo forest W11, W12
- *Vegetation Complexes* Y5, Y6, D3, D4, DK1, FA1, and associated mixed forest DM2, MH, Pn, Ck.

Ecological vegetation system Ip3 and vegetation complex D4 represent a transition type between east and west.

Influence of tree size on product yield

Tree diameter is a critical determinant of sawlog production. Not only must the tree reach a critical minimum size before it yields any sawlog volume, but tolerance to defect and end-product recovery increases significantly with diameter. Figure 24 illustrates the volume of sawlog relative to the gross bole volume of a tree in relation to diameter at breast height and small end diameter (sed) of the log.

Where there is no significant market for logs other than sawlog, maximising the growth of potential sawlog is more important than maximising total stand growth.

Crop trees are those selected to be grown on to produce sawlogs. Choosing co-dominant trees from within their cohort with a well-developed crown will ensure that diameter increment of crop trees is maximised (illustrated in Figure 23). Thinning for wood production in the jarrah forest is typically focused on promoting the diameter growth of potential sawlog ‘crop’ trees until they reach sawlog size, then maximising the stand volume growth by retaining a slightly higher density in subsequent thinnings. Thinning reduces the time for the selected crop trees to reach sawlog size compared to the time that it would take if they were not thinned.

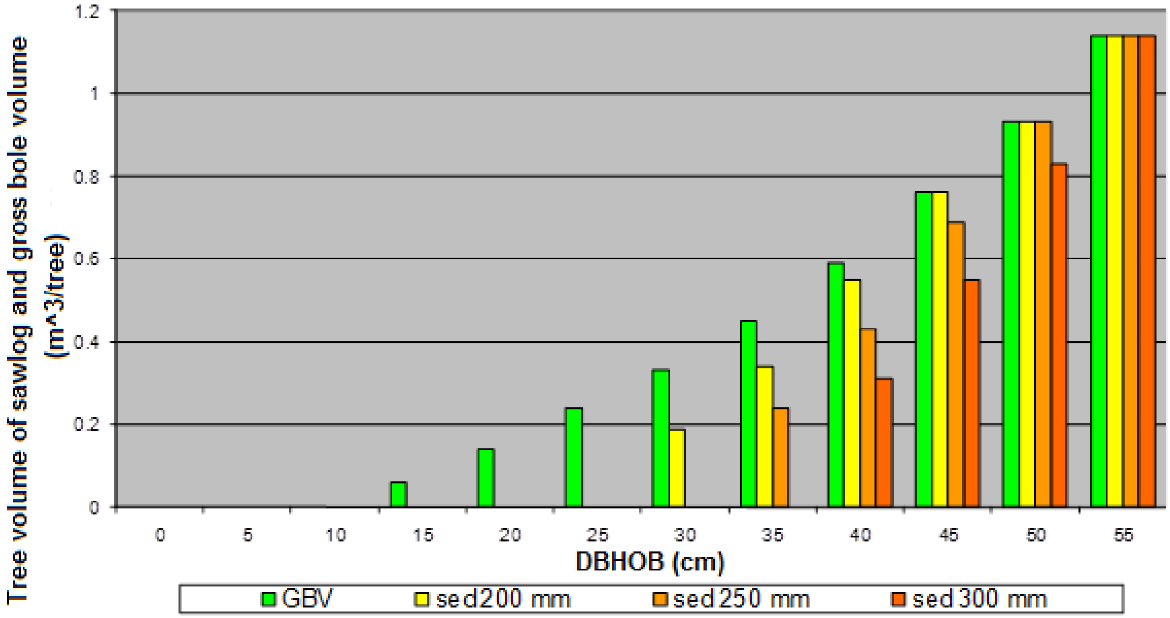


Figure 24: The volume of sawlog of various small end (sed) diameters relative to gross bole volume (GBV) based on a tree with an 8m bole. The minimum dimensions for a grade one sawlog are 2.1m long and 200mm sed. A final crop tree is expected to be grown to a minimum of 50cm diameter at breast over bark (dbhob) i.e. one where all of the bole volume is a sawlog size with a small end diameter of 250mm at 8m).

The maximum number of final crop trees required is based on the number that the stand can support before suppression occurs (at approximately 25m^2 / ha for pole stands). This equates to 125 spha at 50cm dbhob. Maximising the diameter and volume growth on the best 125 spha is therefore often the primary aim. Based on similar logic, 250 spha at 35cm dbhob could be supported. Trees of 35 cm dbhob will produce sawlogs from more than 50 per cent of the bole and are therefore an important source of sawlogs from intermediate thinning and in this sense can be considered as secondary crop trees. In terms of optimal stand density, stems in excess of 250 spha will not produce significant volumes of sawlogs and are surplus to the requirements of sawlog production.

The influence of coppice development

The strong coppicing capacity of jarrah and marri has relevance to thinning practice. Unless it is prevented, the coppice from thinned stumps will quickly negate the effect of thinning by rapidly restoring stand leaf area and therefore competition for moisture. The growth of coppice and regrowth following routine thinning may vary from 0.25-2m^2 / ha / annum over 20 years, which equals or exceeds the growth that can be expected on the retained trees. This quickly restores stand leaf area and reduces crop tree growth.

Stump coppice development may be prevented by the application of approved herbicide(s). Herbicide may be applied by stem injection to the standing trees, applied to the stump of the felled tree, sprayed on the foliage of young coppice (Whitford *et al.* 1995) or by stem injection to saplings off stumps created by the thinning. The latter two are is the least preferred options. Stump application of glyphosate generally has close to 100 per cent success rate, while injection of marri and jarrah is usually 80 per cent or 90 per cent effective, respectively. Stem injection is the method used in non-commercial thinning, but coppice from commercial operations and regrowth that develops following thinning is generally not controlled in routine operations. Control of this coppice and regrowth is essential to maintain water yield (Section 4.4.2) and is similarly beneficial for wood production.

The lignotuberous regeneration process can also result in a rapid restoration of leaf area following thinning, and thinning to less than 'critical density' can be expected to result in considerable regeneration release and competition with crop trees for water and nutrients (Figure 25).

Figure 25: The smaller trees in this stand are coppice and regrowth that have developed in the fifteen years since commercial and non-commercial thinning. The rate of coppice and regrowth development is highly variable across the forest.

Coppice, especially from small stumps, can be an important source of regeneration to supplement ground coppice where regeneration is required. The established root system of stump coppice ensures rapid growth that will exceed that of released ground coppice. Stump coppice is a particularly useful source of regeneration in the drier eastern forest where ground coppice may be insufficient and where seedling establishment may be less reliable.

The influence of competition on the establishment of regeneration

As indicated in Section 2.4, there are some stands that have inadequate numbers of advance growth to respond to the removal of the overstorey and adequately regenerate the site. Competition for moisture affects seedling survival and is considered to be the principal reason for a lack of advance growth (Stoneman 1992).

While the emergence of jarrah seedlings may be enhanced under canopy, their persistence is adversely impacted by competition from the overstorey (Stoneman *et al.* 1994a; Stoneman *et al.* 1994c). To establish regeneration from seed requires a reduction in competition from the overstorey, and sometimes the understorey as well, while retaining sufficient overstorey to provide an ongoing seed source for multiple regeneration events. This process is described a 'shelterwood'. Reduction of overstorey competition in 'shelterwood' is aimed at reducing density to less than the 'critical density', i.e. to about 8-10m^2 / ha in western forest and 6m^2 / ha in eastern forest.

Due to severe competition on sites with established understorey and second storey, jarrah seedlings struggle to establish. Understorey and second storey develop on all sites until the resources available to it are fully utilised and equilibrium is maintained by mortality. The relatively open understorey and second storey observed in the drier forest areas does not imply that there will be less competition for seedlings, but rather moisture has become limiting at a lower density. Competition will be at least as severe as that in moister forest with denser vegetation. Establishing jarrah seedlings where there is inadequate lignotuberous advance growth for regeneration requires a temporary reduction in understorey and second storey competition. Removing the competition will allow jarrah seedlings to establish alongside new understorey and second storey seedlings rather than competing with established understorey and second storey. Sometimes fire alone is sufficient to remove competition. On other sites dominated by rootstock species, moderately intense fire will not kill the competing understorey and second storey (e.g. established banksia, or sheoak), or it will only kill the above ground portion (e.g. *Taxandria* spp.) and the vegetation will re-shoot. In these cases the competition needs to be mechanically removed. The soil disturbance associated with understorey removal also improves the conditions for germination. Some compromise between the requirements for effective regeneration and the maintenance of understorey diversity is required (Burrows *et al.* 2002a).

Influence of overstorey competition on regeneration development

For ground coppice to develop into saplings, the removal of the overstorey is required. While the removal of a single tree may be sufficient to allow some ground coppice to develop into saplings, most saplings will be subject to competition from surrounding trees. Competition will increase as saplings develop. There have been a number of studies that have examined the effect of retaining mature trees from the previous rotation (legacy trees, sometimes referred to as overwood) on regeneration development which have been summarised by Bassett and White (2001). While no studies of legacy tree competition on the growth of regeneration (Stoneman *et al.* 1994c) of have been conducted in jarrah forest, it can be deduced that the effective distance for overstorey suppression will be greater in eastern forest than in the wetter western forest.

At an overstorey density greater than ‘critical density’, all of the regeneration will be under some competition from the overstorey, and that competition will increase with time. Optimum conditions for regeneration release are therefore at the lowest possible legacy tree retention and well below ‘critical density’. Observation of stands treated in the 1930s indicates that the retention of up to 10 per cent crown cover (about $5m^2$ / ha) results in the development of a satisfactory pole stand and may improve the form of the regeneration, compared to those grown without an overstorey (Kessell *et al.* 1937).

Observations suggest that a minimum gap size (in which all overstorey is removed) of two tree heights is the desirable minimum for regeneration release. There is no maximum from a regeneration release perspective and maximum gap size would be limited by other considerations, such as aesthetics or habitat retention. The term ‘gap creation’ is used to avoid the limitations that might be implied by, and possible misinterpretations of, terms such as clearfelling or group selection (Bradshaw 1992).

Influence of stand structure

Stand structure, in terms of the size and spatial arrangement of the different cohorts which constitute the stand, has a major influence on the capacity to manage the forest over multiple harvest cycles.

Where trees are to be harvested in successive cycles, it is important that the felling of large trees in a mixed-aged stand does not destroy the regeneration that resulted from previous regeneration events. Experience in the management of stands created by various methods of timber harvesting over previous decades has shown that small patches of regeneration cannot be effectively protected during the subsequent felling of large mature trees (>30m height). Larger patches of regrowth of the kind commonly created during the period 1920 to 1940 can be protected. However, regrowth in stands that were previously harvested under a single tree or group selection system with very small gaps are invariably smashed during subsequent felling operations. Observation and computer simulation modelling has shown that the regrowth in patch sizes of at least four times mature tree height suffer minimal damage from the felling of large surrounding trees. Smaller patches of regrowth can be protected if they are surrounded by immature trees with smaller crowns.

A system that involves single tree removal of large trees should be avoided. Creating patches of regeneration and retaining patches of older trees, in patches with a minimum size of two to four tree heights, will create a forest structure where regrowth can develop without undue competition and which can be protected during subsequent felling cycles. The integrity of these patches can be maintained by applying a single silvicultural method to each patch when it is being treated. When the commercial timber harvesting and the non-commercial silvicultural operations have been completed, the result should be a forest consisting of discrete patches, where each particular stage of growth in each part of the stand is in a condition most suited to its development and future management.

The long-term retention of legacy trees in the form of habitat trees is somewhat independent of this process. Habitat trees exist where they exist and thus there is often limited choice in where they are to be retained. However, the retention of a habitat tree within a small patch of regeneration will effectively suppress the development of all of that regrowth and should be avoided. Retention of habitat trees in small groups, where this is possible, will reduce their adverse impact on the development of regrowth.

The opportunity may arise in the future to re-consider single tree selection. In stands where there are no longer large veteran trees to be felled and which have been thinned from below at least once to remove the bulk of the poor quality trees, it may be more appropriate to thin from above to provide an earlier sawlog yield without excessive falling damage. As these stands age and thinning is no longer appropriate, regrowth could be released in the larger spaces between the trees, thereby converting to a single tree selection method or a small group selection method. The removal of unwanted small and poorer quality stems would still need to accompany each operation, to ensure that stand quality did not degrade due to repeated removal of only the better trees.

Factors affecting tree form

Bole length is reduced if a fork develops below the level of normal crown break. This may be caused by open growing conditions or the damage to the leading shoot by fire, insect attack, frost or ‘carrot-topping’, following which several shoots develop and there is a failure of one shoot to exert dominance. The relative significance of these factors is difficult to determine, since the increased incidence and severity of frost in low-lying areas also corresponds with the greater likelihood of ‘carrot topping’ on less favourable sites. The premature development of large branches also reduces the potential bole length.

Kessell and Stoate (1937) observed that regrowth in large openings had a greater tendency to develop forks than regrowth where a relatively small percentage of overhead canopy cover was retained. While bole length is greater and branch size smaller in regrowth in small gaps, a survey of pole-sized trees in various gap sizes found no consistent trend in bole height once the gap exceeded two tree heights (Deas *et al.* 1985). The height at which forks occurred was unrelated to gap size. However, a higher proportion of forked trees occurred close to the northern (shaded) edge of the gap. Jacobs (1955) has suggested that insect attack may be lessened by the slower drying of the leaves on the northern edge of the gaps.

The proportion of regrowth jarrah stems forking below 2m in bauxite rehabilitation remained relatively constant at about 60 per cent, at densities that ranged from 600 to 10,000 spha. Regeneration from seed had better form than planted stock (Koch *et al.* 2005). Recent data from bauxite rehabilitation established from seed indicates that at densities up to 2,500 spha, approximately 60 per cent of trees fork below 2m and 75-80 per cent of trees below 4m. Planting or seeding at higher tree densities provided a greater absolute number of better stems but has little effect on the proportion of better stems. However, the proportion of forked jarrah stems in bauxite rehabilitation was considerably less when seedlings were planted in mixture with a faster growing nurse crop such as *E. globulus* or *E. diversicolor* that provided overhead shade and shelter (Danielle Wiseman, pers. comm.)

In a twelve-year-old stand that was thinned before it had developed its ultimate bole height, bole height was unaffected by density, at least at 250 spha or more (unpublished Chalk plot data).

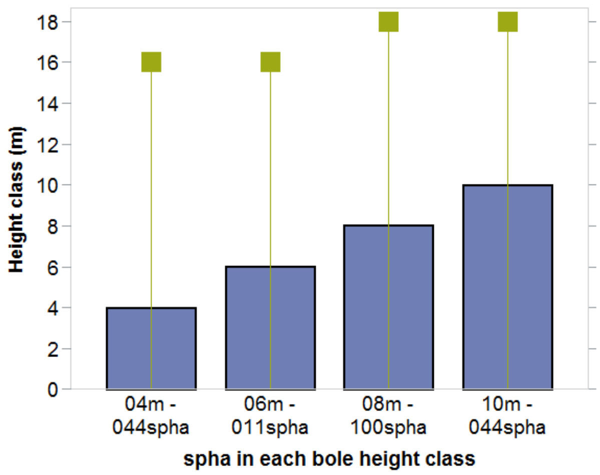
The proportion of forked stems in jarrah regrowth in canopy gaps that are naturally regenerated appears to be relatively high, but sampling[7] of well stocked 20-40 year old regrowth stands indicates there are generally adequate numbers of stems with bole of 8m or more from which to select crop trees (Figure 26, Figure 27).

Kinks in the bole have similar origins as forks except that in this situation one shoot dominates. The size of the kink depends on the length of the growing tip that has been damaged. While kinks will eventually overgrow, they will have a lasting impact in the form of 'wandering heart' (brittle or decayed heartwood) within the tree, a significant cause of the low recovery from smaller diameter logs.

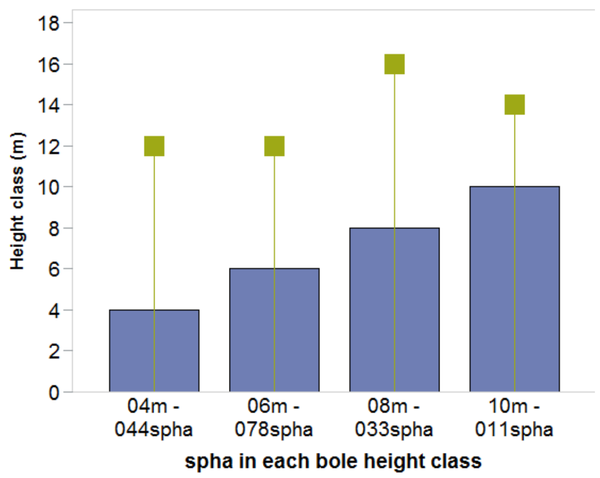
[7] Department of Environment and Conservation, Forest Management Branch permanent sample plots

Dwalgan block

Jarrah | **Marri**



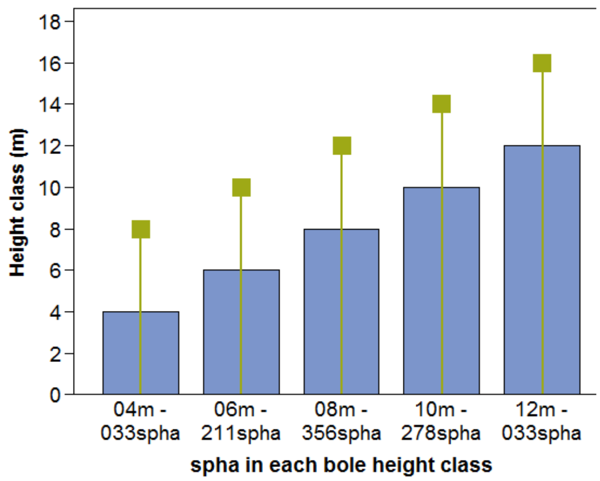
a: Bole height (blue column), average total height (green square) and stocking (spha) of jarrah in each bole height class.



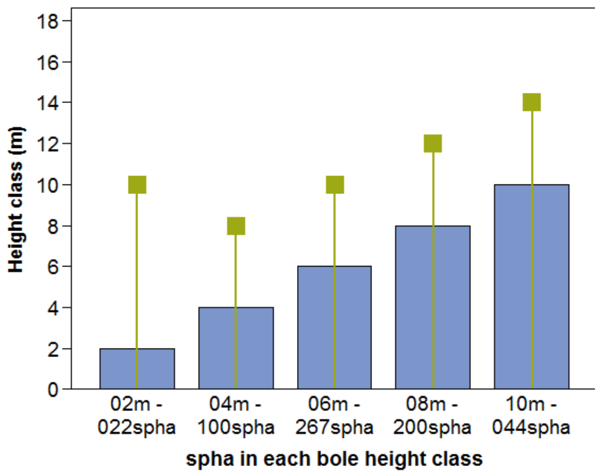
b: Bole height (blue column), average total height (green square) and stocking (spha) of marri in each bole height class.

Gordon block

Jarrah | **Marri**



c: Bole height (blue column), average total height (green square) and stocking (spha) of jarrah in each bole height class.



d: Bole height (blue column), average total height (green square) and stocking (spha) of marri in each bole height class.

Figure 26: The relationship between total height (green) and bole height (blue) from plots in two contrasting regrowth stands. The higher bole / total height ratio in Gordon block (c&d) is a reflection of both higher stocking and higher quality forest. Dwalgan block (a&b) is 36 years old eastern jarrah forest. This plot has a density of 365 spha and consisted of 144 spha jarrah and 44 spha marri with a bole length of 8m or more. Gordon block is 25 years old western forest. This plot has a density of 1,544 spha, 311 spha of jarrah and 244 spha of marri with a bole length of 8m or more.

Nursery raised seedlings that are planted and fertilised have a high survival rate. Except where they are planted in the wetter southern forest, often with a taller understorey, a high proportion develop forked stems. Marri behaves in a similar fashion. Research is required to determine the extent to which this might be rectified by later coppicing.

4.2.2 Monitoring of regeneration

Determining the density of regeneration required at different stages of development to ensure that the stand develops appropriately, depends on the management goals and the growth habits of the species. The number of stems required at different stages generally follows a negative exponential pattern to ensure that there are enough stems at the youngest stages to allow for mortality over time, to maintain appropriate stem form, and provide sufficient stems to provide a choice of crop trees for the future. In the absence of long-term studies, it is necessary to base the density requirement decision on observation of existing stands at different stages and sites, taking into account the reasonable expectations of stands that are naturally rather than artificially regenerated. Several reviews have been undertaken in the past and standards have been based on observations of what can be normally expected in natural stands at different stages in terms of both density and variability. The later the stage of development, the greater the confidence that can be placed in its future survival, especially in harsher environments (Abbott *et al.* 1986; Department of Environment and Conservation 2004). Different regeneration standards are also required to reflect the natural differences that are expected between western and eastern jarrah forest. The silvicultural guidelines give details of the current standards.

There are a variety of methods that can be used to monitor density of regeneration (Lutze 2001). The method providing the most useful information is the stocking rate, a measure that determines the percentage of the area (percentage of plots) where the density of stems exceeds a defined standard (e.g. 60 per cent stocked at >1,000 stems per hectare). This method, while more time consuming to measure, has the advantage over the 'average density' method by providing information on how much of the regeneration reaches a minimum standard; and it provides the flexibility to re-analyse the stocking rate for any nominated density. The required standard can be defined by both density and per cent stocking.

It is desirable that regeneration occurs as closely as possible after timber harvesting (where it is required) and this is usually not difficult to achieve where regeneration is to be released in gaps, provided that any necessary follow-up culling work is done promptly.

However, it is unrealistic to expect that regeneration of seedlings in shelterwood operations can be achieved so promptly. Co-ordination of timber harvesting, culling, site preparation, seed crop and burning opportunities is complex and delays are difficult to avoid. It is also unrealistic to expect that adequate regeneration will always be established in one regeneration event, especially in the eastern forest. Where regeneration is being established under shelterwood, the earliest that it could be expected before that regeneration is sufficiently developed to release is about 20 years. It must be expected then that sawlog trees retained as shelterwood will be unavailable for timber harvesting until that time and this has an important influence on yield scheduling.

4.2.3 Fire management

Fire has an important role in various aspects of silviculture. Different silvicultural methods (e.g. thinning or regeneration release) have different fire requirements, each with different technical aspects and often with multiple goals. Where the burning requirements of different stands conflict, compromise is required.

Pre-harvest burning

Advance burning may have several goals including: improved faller safety, improved access, hazard reduction, and evaluation of advance growth. The timing of the advance burn varies with the purpose. Conditions for evaluation of the lignotuber pool are ideal eighteen months after a burn, but dieback mapping cannot be done within three years of a burn. Burn intensity, within reasonable limits, is not usually a critical silvicultural factor except where it may cause the abortion of buds or premature seed fall from a shelterwood stand.

Post-harvest burning

Tops disposal burning is required as a fire hazard reduction measure, aiming to remove flash fuels and woody debris up to 2.5cm in diameter that results from timber harvesting, including thinning. This will help protect retained trees from heat damage in subsequent unplanned fires.

Regeneration release burning is carried out in gaps where advance growth is already established and has been released by timber harvesting. The purpose of this type of burn is to enhance the development of the advance growth by the stimulation of the growth of ground coppice and saplings, the temporary reduction of some understorey competition, the burning back of poorly-formed or damaged saplings, and hazard reduction. The goal is to remove woody debris to 7.5cm diameter. Regeneration release burns should be conducted at mild to moderate intensity depending on the type of regeneration present. A mild intensity burn is sufficient to stimulate dynamic ground coppice.

Regeneration establishment burning is carried out in the shelterwood cut areas with the object of creating a seedbed, stimulating seedfall, and temporarily reducing understorey competition. Regeneration establishment burning needs to be of moderate intensity in order to create effective seedbeds. Crown scorch is acceptable. The goal is to remove woody debris up to 7.5cm in diameter. Regular burning thereafter will promote the growth of developing lignotubers and provide further opportunities for regeneration to establish.

Regeneration establishment burning should be conducted at moderate intensity within 18 months of the completion of harvesting to maximise the combined effect of site disturbance and ashbed, subject to there being an adequate seed crop. Burn intensity should not compromise the survival of crop trees or habitat trees.

Post-regeneration fire exclusion

Developing saplings are vulnerable to damage from low intensity fire until they reach a height of 5-6m and have a dbhob of 10cm, by which time the bark is thick enough to resist cambial damage and the tip is high enough to escape damage from a low intensity fire (Peet *et al.* 1971). Fire should therefore be excluded from areas cut to gaps until the saplings have reached this stage. This generally takes 10-20 years, depending on site quality.

Post-regeneration burning

Following the period of fire exclusion, burning for hazard reduction or other purposes may be resumed. The specifications and management of the first burn following fire exclusion, when fuel loads are high and the saplings are sensitive to damage, are critical if damage is to be avoided. Fire intensity of no more than 220kW / m in spring or 120kW / m in autumn is required to keep scorch to about 5m (Burrows 1997).

Following this, regular prescribed burning may resume.

4.2.4 Summary of silvicultural methods in the jarrah forest

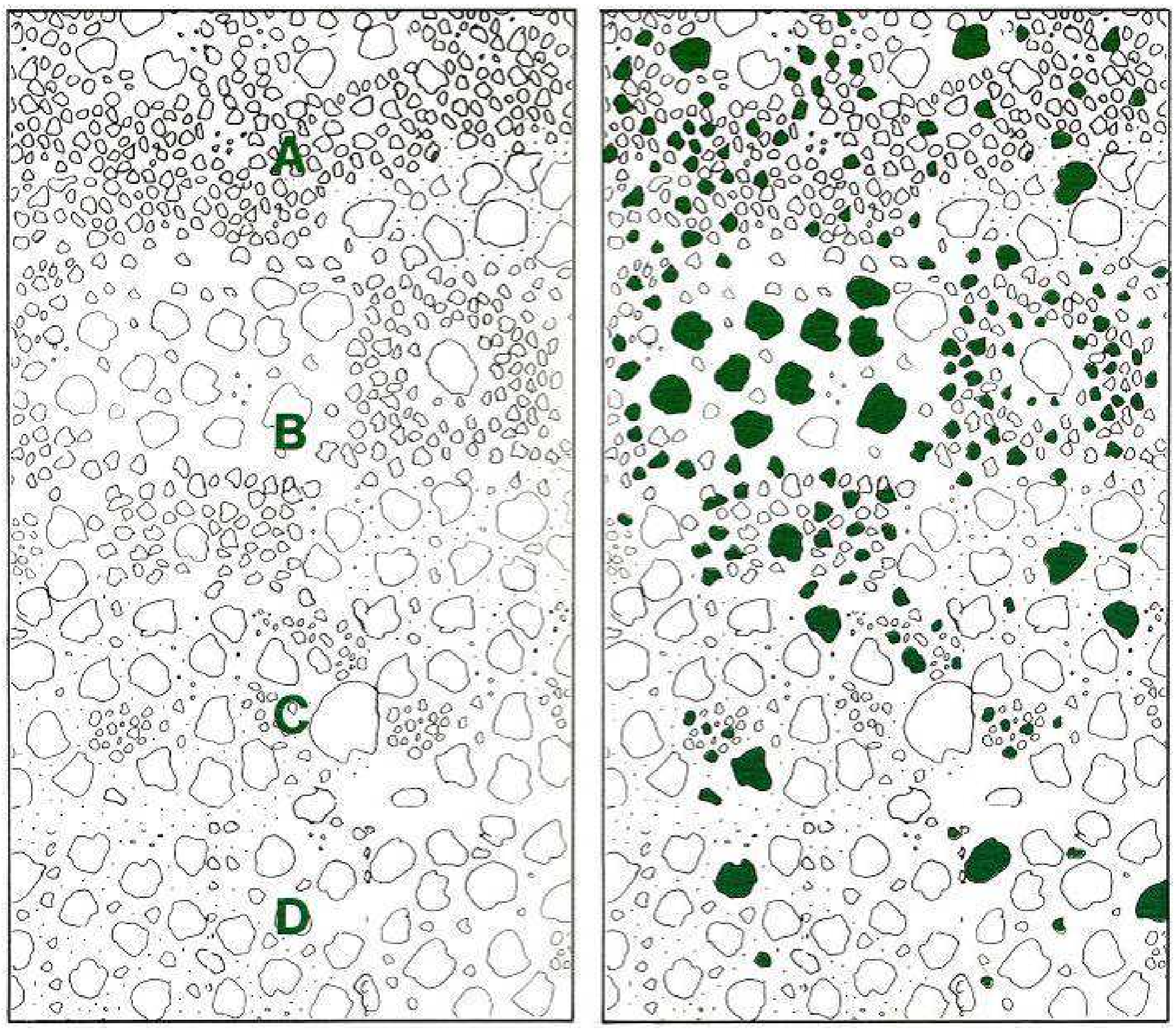
The jarrah forest is a complex mosaic of stand structure (Figure 12, Figure 27) and although discrete structures can be recognised, structural variation occurs continuously throughout the forest, often blending from one to another without distinct boundaries.

The aim of silviculture is to apply the treatment most appropriate to each patch within that mosaic (Bradshaw 1987).

Silvicultural practice is based on the principle that if a tree is to be removed from a stand it should be done to achieve one of three principal objectives:

- to promote the growth on the remaining trees. Regeneration is not required at this stage. The silviculture treatment to achieve this is called thinning;
- to reduce competition through removal of the overstorey to allow ground coppice to develop unimpeded into saplings. The silviculture treatment to achieve this is called gap creation;
- to reduce competition through partial removal of the overstorey to allow seedlings to establish and develop into ground coppice. The silviculture treatment to achieve this is called shelterwood (Bradshaw 1987).

These objectives do not apply to high impact dieback sites, the treatment of which is described below.



(i) (ii)

Figure 27: (i) An illustration of the structural variation in the jarrah forest.

A - Even-aged regrowth from gap creation.
B - Distinct groups of retained mature trees left after earlier group selection cutting together with patches of saplings and pole created by group selection cutting with follow-up silvicultural treatment.
C - All aged stands from single tree selection.
D - Mature forest.

(ii) An illustration of the variation in distribution of crop trees (coloured green) in different parts of the forest which will influence the choice of silvicultural method.

Thinning (to promote growth on selected trees)

Where there are sufficient crop trees in immature regrowth stands, the principle objective is to promote growth on retained trees by thinning. Thinning aims to reduce stand density to 'critical density' and then thin again before the stand reaches suppression density. Thinning is more conservative as the stand ages if sawlog growth is to be maximised. Thinning to less than 'critical density' will result in some reduction in stand growth. However, this will increase the diameter growth of retained trees, reducing the time to reach the desirable size for a sawlog.

In the absence of markets for non-sawlog material, thinning of immature stands is generally carried out with a combination of commercial (sawlog) thinning followed by the culling of non-commercial trees. For wood production purposes only, those culls directly competing with the crop trees need be removed. However, where water production is part of the management objective, further culling to the nominated density to be retained is required to achieve the expected outcome.

The emphasis of thinning is thinning from below, to promote the growth of the larger crop trees. Where some trees have already reached sawlog size, a combination of thinning from above and below may be appropriate provided that a full stocking of crop tree quality trees is retained. However, the commercial removal of trees where only a portion of their bole is of sawlog size, or which have just reached minimum sawlog size, should be delayed where possible as these trees are just starting to produce high value material.

Although there has been no operational thinning of juvenile stands, due to lack of suitable markets, this method is suitable to apply to these stands.

Gap creation (to release regeneration)

Where a patch within the stand has an inadequate stocking of crop trees suitable for thinning and adequate advance growth is present, regeneration release is an appropriate objective.

The object of gap creation is to release ground coppice to grow into saplings and beyond with minimal competition from existing overstorey, and in sufficiently large patches so that the felling of overstorey trees in the future will not damage the developing regrowth.

Effective gap creation nearly always requires the removal of some cull trees.

Shelterwood (to establish regeneration)

Where there is an inadequate stocking of crop trees for thinning and an inadequate stocking of ground coppice, saplings and potential coppice suitable for immediate release, then regeneration establishment is appropriate.

The object of shelterwood harvesting is to reduce the competition from the overstorey (and the understorey where necessary), establish regeneration and maintain a forest cover while the regeneration goes through the prolonged period of development to the ground coppice stage. While the process is the same as classical shelterwood systems, the term is used loosely here since the regeneration does not require the shelter of the overstorey to survive; its primary purpose is to provide a seed source for multiple regeneration opportunities and to maintain a forest environment for aesthetic and biodiversity purposes.

Following timber harvesting, culling is generally necessary to reduce the density to that required for shelterwood.

When the shelterwood cut area has been successfully regenerated and adequate ground coppice has become established (which may take 20 years or more), the retained trees may be removed to create a gap and release the ground coppice to develop into saplings. Figure 28 illustrates how these practices are sequenced over time.

Selective cut in dieback infested stands

Regardless of the existing structure, stands which are infested with *Phytophthora cinnamomi* or are determined not to be protectable from autonomous spread in the near future, and where the impact of the disease is expected to be high, are selectively cut. The aim is to minimise the potential impact of the disease, to promote an ecosystem of resistant species and retain the productive potential of the forest should the disease impact remain low.

These stands are thinned uniformly, retaining 15m^2 / ha, with a preference for the more *Phytophthora* dieback resistant blackbutt or marri. Healthy jarrah that has survived for a long period on infested sites and which exhibit apparent field resistance may also be retained. These trees may form an important genetic resource and potential source of seed. The objective of retaining a relatively high stand density is aimed at maintaining lower soil temperature and soil moisture to minimize the potential impact of the disease. The limit of 15 m^2 / ha is based on a limited study of the effect of different thinning intensities on *P. cinnamomi* lesion growth at a time when soil moisture was at a generally higher level than it is today (Bunny *et al.* 1995). This requires further research to confirm its suitability for ongoing application.

Completing the silvicultural method

As mentioned above, the application of the silvicultural method (thinning, gap creation or shelterwood) nearly always requires the culling of surplus trees. Where excessive culling would be required to achieve the objective, or where budget constraints limit the amount of work that can be done, it may not be possible to achieve the intended objective. Achievement of the silvicultural method is deferred indefinitely, or until markets become available for at least some of this cull material. These stands are described as selectively cut. This has implications for future availability of wood resources.

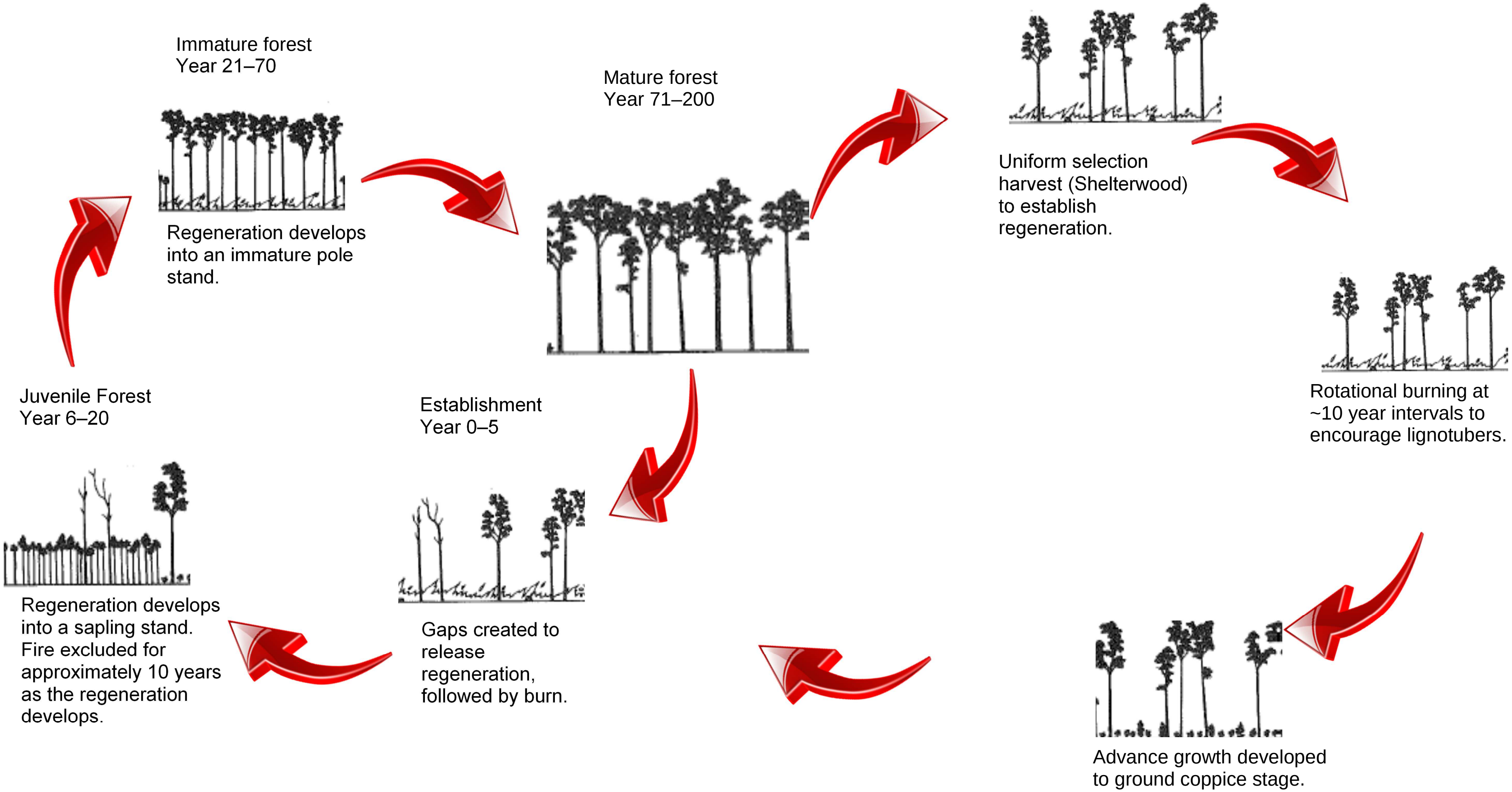


Figure 28: A diagrammatic illustration of the sequencing of silvicultural treatments over time.

Mining rehabilitation

Stands subject to bauxite mining are completely cleared of vegetation prior to mining and rehabilitated with native species after mining. Current establishment criteria for mine rehabilitation aims to have most areas between 800 and 1200 spha with a maximum density of 2,200 spha and a minimum of 650 spha. This is aimed at producing 300 spha with a bole of 3m or more. This standard is well below the potential for jarrah (Figure 26). To produce 300 spha with a more realistic 4m minimum bole for example, requires a minimum (not average) density of approximately 1,300 established spha. This would be achieved with an average density of 2,200 spha (A. Grigg, Alcoa pers. comm.). Alternatively the use of a fast growing nurse crop (e.g. karri) to provide overhead shade would achieve a similar result at lower density, though further research is required to quantify this.

Current rehabilitation standards aim to achieve a balance between sawlog production and stand resilience in a drying climate. An alternative approach would be to establish rehabilitation at 2,200 spha followed by thinning to 300 spha at about age 10 and a further thinning a decade later. This would promote both sawlog and water production values by maintaining an average LAI in the order of 1.15 over the first 30 years (see section 4.4.2.).

4.2.5 Climate change

When considering climate change it is expected that the long-term productive capacity of south-west forest ecosystems (and plantations) will be affected by the predicted higher atmospheric carbon dioxide concentrations and associated drier and warmer conditions. Accordingly, the ‘high severity’ climate change scenario (see (Maher *et al.* 2010) based on CSIRO (2007)), has been applied when modelling sustained yields for the FMP. Climate modellers project reductions in rainfall as well as changes to seasonality and reliability of rainfall. As a consequence, growth rates could slow over the long term due to declining moisture availability. 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 CO_2 and temperature is uncertain. While it is possible that increasing CO_2 concentration will increase water use efficiency and growth rate, it is uncertain which factors will ultimately become limiting since the interactions between water use, CO_2 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 CO_2 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 sustained yield calculations. The physiological growth model 3-PG[8] (an acronym for Physiological Processes Predicting Growth) has been widely used around the world to simulate the growth of even-aged forests, and DPW has calibrated the model for use in both jarrah and karri forests. Predictions of the growth in standing volume over time were made using the calibrated 3-PG model (using the historical rainfall and temperature recorded at nearby meteorological stations).

The performance of the calibrated 3-PG model was tested using simulated data against observed data collected from selected permanent sample plots (PSPs). The model was initialised with the 1964 measurement of the Inglehope thinning trials and tree growth was simulated up to the 2010 measurement using monthly climate data observed during the growth period. The simulation results were then compared with the actual data of the last measurement for model validation.

Under a medium severity climate change scenario (CSIRO 2007), the periodic annual increments (PAIs) of jarrah did not reduce significantly from 2010 to 2030, but reduced by 7 per cent and 14 per cent in 2030-50 and 2050-70, respectively. In contrast, a high severity climate change scenarios (CSIRO 2007) reduced PAIs significantly by 10 per cent, 30 per cent and 45 per cent in 2010-30, 2030-50 and 2050-70 respectively, in relation to projected growth under 1990-2010 conditions. According to the simulation results, most reduction in stand volume increment in the jarrah forest occurred in dry summers as a result of the interaction between rainfall decline and temperature increase, with most jarrah growth occurring in winter and spring.

This modelling did not take into account the potential effect of increased CO_2 concentration leading to increased growth, or explore how varying the timing and intensity of future 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 broadscale 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 CO_2.

Climate change has implications for site potential. Reducing rainfall will cause increased competition for moisture and increased moisture stress. A new equilibrium stand density will eventually be reached through mortality. The future relationship between rainfall and stand density is difficult to predict but is likely to be different to the present relationship (Figure 10) because of the additional influence of soil type on site potential. The pattern of this adjustment and the time it will take to adjust to these new conditions is presently unknown.

Thinning reduces stress and improves the health of the remaining trees and also 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.

[8] 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 can achieve sustained yield of sawlogs and to 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

A number of organisms can adversely affect the health of the jarrah forest. These include jarrah leaf-miner (*Perthida glyphopa)* (Abbott *et al.* 1993), *Armillaria luteobubalina* (Shearer *et al.* 1988), gum-leaf skeletoniser (*Uraba lugens*) (Farr *et al.* 2004), and *Phytophthora cinnamomi* (Dell *et al.* 1989b)*. Armillaria* and canker (*Quambalaria coryrecup*) affect marri (Paap 2006). A variety of health surveillance activities are undertaken and have been summarised by (Robinson 2008).

Phytophthora cinnamomi, the cause of jarrah dieback*,* is an introduced plant pathogen and remains the most significant jarrah forest health issue. First observed in the jarrah forest in the 1920s, the causal agent was not isolated until 1964 (Podger *et al.* 1965). The disease affects more than 20 per cent of native plant species in the south-west, the most susceptible species belonging to the Proteaceae, Leguminosae, Myrtaceae, Epacridaceae and Xanthorrhoeaceae families, all important components of the jarrah forest flora. Marri and blackbutt are resistant to the disease. Disease spread was most rapid from the 1940s to the 1960s when the disease was unknowingly spread in infected soil and gravel and many of the most susceptible sites (water-gaining and lateritic cap rock sites) were severely impacted. Up to 30 per cent of the most severely affected western scarp in the northern jarrah forest was estimated to be infected by the 1970s (Batini *et al.* 1972). With a high proportion of the sites in low landscape positions already infected, autonomous spread of the disease has slowed. An estimated 15 per cent of the jarrah forest available for timber harvesting is currently infested and this was projected to increase to 34 per cent by 2060 (Ferguson *et al.* 2003; Strelein *et al.* 2004). More recent work by the Department shows that the autonomous rate of spread has declined in the past decade in the northern and eastern jarrah forest, presumably as a result of reduced rainfall. This work suggests autonomous rate of spread has remained much the same in southern forests.

Disease impact from *P. cinnamomi* ranges from minor, where only a small proportion of the understorey is susceptible and the jarrah remains unaffected (low impact), to what is described as ‘graveyard’ (very high impact), where all of the jarrah and most of the understorey is highly susceptible to the pathogen. The end expression of the disease (impact) is influenced by soil type and landform, position in the landscape, moisture status, rainfall and species composition. Although disease impact cannot be precisely determined, areas of forest are separated for management purpose into low-moderate, high and very high impact sites on the basis of vegetation complex.

There is some evidence that *Phytophthora* lesion growth in jarrah may be increased by reducing stand density in areas already infested with the disease (Bunny *et al.* 1995). This is likely to be due to increased dry season moisture availability and increased phloem moisture content of the retained trees (Tippett *et al.* 1987). This study indicated that this density limit may have been about 15m² / ha. However, this has not been confirmed by field monitoring and it is possible that this threshold may have changed (lowered) under current climatic conditions. Further research is required.

There is wide variation in genetically controlled resistance of jarrah to *Phytophthora cinnamomi*. However, survival within a diseased area is an unreliable indicator of resistance (Stukely *et al.* 1994).

4.3.2 Fire management

Prevention of serious damage by fire is a major consideration in the maintenance of forest health and vitality. While the jarrah plant is both resistant and resilient to fire at all but the most extreme fire intensities, it can be significantly damaged by severe fire. This may vary from damage to the bole and crown of the tree, to the death of the above ground parts of the tree or the death of the whole plant. Moderate to high intensity fire will kill the above ground components of most understorey and second storey plants and nearly all of the 'non-sprouting' plants.

In the climate of the jarrah forest, severe fire weather conditions occur every summer and into autumn as do ignitions from both human and natural sources. Fires that become established in these conditions where heavy litter fuel loads exist cannot be effectively suppressed by direct attack.

The principal means by which the success of fire suppression may be improved and fire damage minimised is by strategic fuel reduction burning (McCaw 2013) (i.e. prescribed burning).

Prescribed burning is carried out in the jarrah forest not only to reduce fuel loads, and improve the ability to control bushfires and reduce their damage, but also to regenerate the forest after timber harvesting, to promote nutrient cycling, and to maintain diversity in understorey age structure for biodiversity reasons.

Bushfires can generally be controlled by direct attack when fire intensity is below 2000kW/m of fire line. In jarrah forest, on flat ground and under moderate summer weather conditions, this generally occurs when the fuel load is less than 8 tonnes / ha. The difficulty of controlling bushfire and the number of days per year when 'uncontrollable' conditions occur increases with forest fuel load (Gill *et al.* 1987; Boer *et al.* 2009). Fuel loads in the jarrah forest reach eight tonnes / ha at about five to ten years after a previous fire, depending on forest density and site (Sneeuwjagt *et al.* 1985). Using prescribed burning for hazard reduction should recognise this. The work of (Boer *et al.* 2009) which compared the extent of historical prescribed burning and the incidence and extent of bushfires suggests that approximately 45 per cent of the landscape having a fuel age of less than six years provides effective bushfire risk reduction.

The frequency of burning and the spatial diversity of fuel age required for biodiversity is more complex since no regime is suitable for all of the wide range of organisms involved (Burrows *et al.* 2003; Friend *et al.* 2003). Floristic composition and structure change with time since fire, creating different functional habitats at a localised scale.

Diversity in post fire ages, and spatial diversity of seral stages and functional habitat are fundamentally important for ecosystem health and biodiversity (Burrows 2008).

The extent and the frequency at which prescribed burning should be done has been a controversial issue. Proponents have argued that insufficient burning is being done to give effective protection, while opponents argue that frequent burning adversely affects biodiversity. In arriving at an appropriate balance the following issues need to be considered.

While a long period of very frequent burning at a point in the landscape will change species composition of the understorey, the frequencies used for fuel reduction burning have not been shown to result in significant changes, though variation in frequency (and seasonality) is beneficial for diversity (Burrows *et al.* 2003). The jarrah forest has high resilience to a wide range of fire regimes (Wittkuhn *et al.* 2010).

Areas of high fuel load will inevitably be burnt at some stage and when they do, the fire will be intense, with few unburnt patches or refuges left within the burnt areas (Cheney 2010). Low intensity prescribed burning on the other hand aims to maintain a fire induced fine grained mosaic of fuel age. An individual prescribed fire operation with an objective of bushfire risk management aims to retain about 30 per cent of the area unburnt on any one occasion. Therefore a proportion of the fuel load mosaic with a higher fuel load (and understorey age) will always be present and the low fuel mosaic in which it is embedded decreases the probability of its ignition during bushfire events. Fuel age frequency distributions that optimise both bushfire risk and biodiversity management outcomes generally conform to a negative exponential distribution that reflect a fine grained, fire induced mosaic of fuel age.

The issues involved and the resolution of these sometimes competing objectives are discussed in the Special Fire Edition of Landscope (CALM 2000). The proportion of the jarrah forest with different times since the last fire (classified as fuel age) (at 2015) is shown in Figure 29.

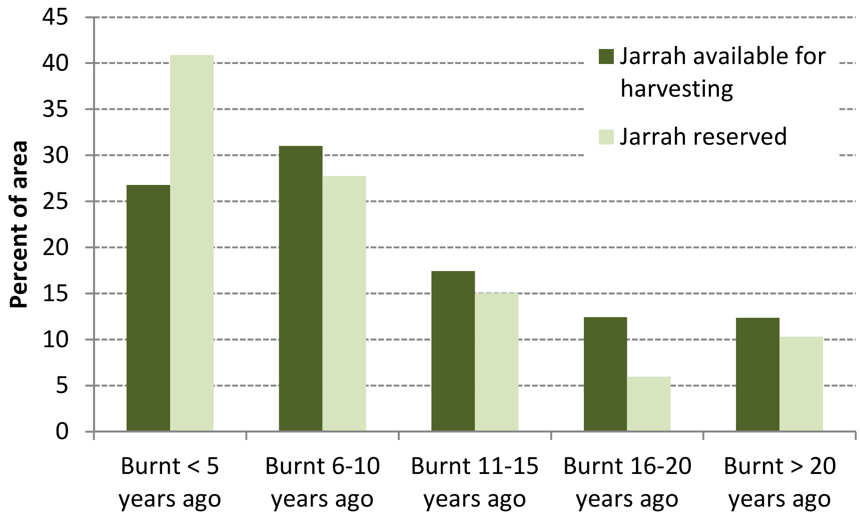


Figure 29: The percentage of the area of reserved jarrah forest and jarrah forest available for harvesting by fuel age class (as at April 2015). Source: Fire Management Services.

Based on the fuel age distribution (Figure 29), fuel load is such that a direct attack on a headfire under common summer fire weather conditions could be expected to succeed on about 40 per cent of the forest available for harvesting and about 60 per cent of the jarrah forest in reserve (where fuel age is less than 8 years). Under those weather conditions crown fires and tree mortality could be expected on those areas where fuel age exceeds 20 years (about 10 per cent of the jarrah forest available for harvesting and about 12 per cent of the jarrah forest in reserves) (Sneeuwjagt *et al.* 1985; Bell *et al.* 1989; Burrows 2000).

The relationship between fire and other aspects of the ecosystem in the south-west of WA have been reviewed in Abbott and Burrows (2003).

4.3.3 Climate change

The reduction in rainfall predicted under climate change scenarios (CSIRO and Bureau of Meteorology 2007) will, if realised have widespread impact in the long term, increasing drought stress and reducing the carrying capacity of stands in lower rainfall areas (Figure 11 11). A combination of reducing rainfall and increasing evapotranspiration is predicted to result in significant reduction in the maximum LAI that can be supported throughout the forest (Croton *et al.* 2015). Reductions in rainfall since the 1970s have already seen significant reductions in stream flow and a lowering of the water table. Stream biota will be affected first, followed by the understorey which is subject to moisture stress for 1-2 months longer than the deeper rooted trees (Crombie *et al.* 1988). The deep rooted tree strata are likely to be the last to be affected, though drought deaths are already evident on some areas of shallow soil (Davison 2011). Jarrah is likely to be affected before marri (Crombie *et al.* 1988).

Under a medium severity scenario for climate change (CSIRO and Bureau of Meteorology 2007), an estimated additional 63 000ha of department managed jarrah forest will be in areas receiving less than 600mm / ann rainfall by 2030, less than needed to support a jarrah forest (Figure 2). More than 80 per cent of this area is within the conservation estate. However, this would be partly offset in areas where karri forest retracts and the area becomes more suited to jarrah and marri, reducing the net effect to 56 000ha (Maher *et al.* 2010). Under this scenario the percentage of jarrah forest where there is negligible streamflow (<700 mm / ann) will increase from nine per cent to 18 per cent and there will be an increase the number of perennial streams becoming seasonal, with a consequent impact on stream biota.

Moisture stress can be reduced and forest health improved by thinning to maintain stand density closer to the critical density (**Error! Reference source not found.**) (Stoneman 1986).

The effects of climate change on pests and pathogens in the jarrah forest are uncertain. While a drying climate may inhibit dieback activity in northern forests, higher temperatures may improve conditions for dieback in southern forests (Christensen 1975). Monitoring by the Department over the last decade has indicated that the autonomous rate of spread has decreased in the northern and eastern jarrah forest and remained much the same in southern forests. The effect on insect activity is less predictable at this stage and may be adverse or beneficial.

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 the potential impacts have been carried out for the jarrah forest, issues of particular relevance would be any increase in the incidence of lightning ignitions and any further incursion of tropical cyclones down the west coast south of latitude 30^{o}S (McCaw *et al.* 2003; Maher *et al.* 2010). Other factors that may be of even greater significance include changes to weather patterns that impact on 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 and fertility is perhaps the most fundamental element of sustainability. Timber harvesting has some potential to affect soil nutrient levels, however, these effects are small relative to the existing stores of readily extractable nutrients in the soil under current management practices (Hingston *et al.* 1989). While jarrah responds to addition of both nitrogen (N) and phosphorus (P), no fertiliser application is undertaken operationally except for the addition of fertiliser at the time of planting in rehabilitation sites (Stoneman *et al.* 1996).

Fire has an important role in nutrient cycling in the generally infertile soils of the jarrah forest (Wittkuhn 2002; O'Connell *et al.* 2004). Jarrah has a slower rate of litter decomposition than many other eucalypt forests and a significant store of nutrient is held in the litter. While some nutrient (mainly N) is lost to the atmosphere by the burning of litter and understorey, fire has a positive role in releasing organically-bound nutrients in the litter into available inorganic forms; and by promoting the regeneration of N-fixing understorey which not only fix N from the atmosphere, but also increase the rate of decomposition and mineralisation of the litter. Most N fixation from *Macrozamia*, *Acacia* and *Bossiaea* occurs in the first three years after a fire and reduces to very low levels at six years (Hingston *et al.* 1989). Fire increases the spatial variability of nutrients in the soil (the ash-bed effect is an extreme example) and this heterogeneity is important for biological diversity (Adams *et al.* 2003).

A study of different fire regimes (from no fire to four fires in 20 years) in a low rainfall jarrah forest indicated that unburnt sites had generally higher surface soil nutrients, lower tree growth rate and no apparent difference in tree health (Burrows *et al.* 2010).

Regular burning is important for nutrient cycling (Wittkuhn 2002; O'Connell *et al.* 2004; Department of Environment and Conservation 2007).

Soil structure can be adversely impacted by soil compaction and the occasional mixing of soil profiles during timber harvesting operations (Rab *et al.* 2005). The impact is minimised mainly by restricting machine activity during moist soil conditions and concentrating, rather than dispersing traffic over the site (Department of Environment and Conservation 2010; Whitford *et al.* 2011; Whitford *et al.* 2012).

4.4.2 Water

There are several aspects to the relationship between silviculture and water. These relate to quality and availability for environmental and consumption purposes.

Sedimentation

The principle 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 and maintenance 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, timber harvesting operations can result in increased sedimentation for a period of two to three years if harvesting occurs through the stream zones, particularly in winter. Steam sedimentation is readily prevented effectively by the retention of undisturbed, vegetated stream buffers (Borg *et al.* 1988; Croke *et al.* 1999; Water and Rivers Commission 2001).

Water quality

In areas with high groundwater or soil salinity, reduction in vegetation cover sufficient to cause the water table to rise and intercept the surface will cause saline discharge into streams. While permanent clearing in these areas has been shown to result in a major increase in stream salinity, this has not been the case for forest operations such as thinning or regeneration. Groundwater discharge from thinning in the high rainfall area results in an insignificant increase in stream salinity because the groundwater in these areas has low salinity levels. However, thinning or other timber harvesting operations that cause groundwater to intercept the surface in the intermediate and low rainfall zones could cause saline discharge into the streams (Bari *et al.* 2003). 'Phased regeneration' in which a proportion of each 2nd order catchment is retained at higher density (or uncut) during the period of regeneration (and lowered water use) was introduced to reduce this risk. The zone of greatest risk is the intermediate zone (900-1,100mm rainfall / annum), where the moderately saline groundwater is closer the surface. The lowering of the water table in recent years (by as much as 14m in the Swan Region from 1975 to 2010) as a consequence of reduced rainfall, has significantly reduced the risk of groundwater discharge, enabling phased harvesting to be dispensed with in most of the eastern forest (Conservation Commission 2013a).

Water availability

The maintenance of 'normal' streamflow is important for sustaining stream biota, streamside and swamp vegetation and maintaining water supply for human consumption or irrigation. Reduction in rainfall over the last thirty years has seen a reduction in streamflow from the northern jarrah forest of about 50 per cent, and a reduction in the period during which streams flow. Some fourth order streams have ceased to flow in summer for the first time on record. Reductions in groundwater levels by as much as 14m from 1975 to 2010 has resulted in a disconnection between the groundwater and surface system from about 2000 in many areas. The rainfall required to produce a particular level of streamflow is now considerably greater than before disconnection occurred (Kinal *et al.* 2011; Kinal *et al.* 2012; Reed *et al.* 2012).

Reducing stand density reduces leaf area, reduces interception, reduces evapotranspiration and increases streamflow where thinning is sufficient (Stoneman *et al.* 1989c).

A series of 27 small catchment studies conducted in south-west WA during the 1980's and 1990's examined the effects of a range of vegetation removal activities on water yield (Bari *et al.* 2003). Permanently clearing vegetation (for example, for agriculture) resulted in sustained water yield increases of 20 to 30 per cent, depending on average rainfall. A greater increase in water yield was associated with higher rainfall areas. Forest thinning of high rainfall catchments increased water yield by a maximum of eight to 18 per cent. The increase was dependent upon the characteristics of the catchment and the amount of vegetation removed. However, the increases from thinning were not permanent and water yield retuned to pre-thinning levels after 12-15 years when the stands reached a density of about 20-25m^2 / ha (Bari *et al.* 2003).

In southern catchments where the groundwater response to timber harvesting was measured, additional groundwater recharge resulted from the reduction in vegetation cover. Groundwater levels and groundwater discharge area initially increased. After about 10 years as vegetation cover increased, groundwater levels and groundwater discharge decreased. Base flow (subsurface and deep groundwater flow) was responsible for two thirds of the streamflow increase (Bari *et al.* 1994).

While stream buffers reduce sedimentation, they also reduce groundwater recharge and are expected to delay or reduce the streamflow response to canopy reduction (Bari *et al.* 1993).

Although the responses described above occurred in the period since reduced rainfall, there has been a further general decline in the groundwater level since then as a consequence of the last 30 years of reduced rainfall. While similar treatments applied in the future can be expected to cause an increase in the recharge of the groundwater, the increase in streamflow is expected to be much less and it may take longer to occur, depending if the groundwater rises sufficiently to intercept the surface and how long this takes (Kinal *et al.* 2011). Heavier thinning is required now to achieve the same response (Reed *et al.* 2012).

Following the initial increase in streamflow following thinning, most of the subsequent decline in streamflow is due to the rapid increase in density from stump coppice and other regrowth. Successful regrowth and coppice control could be expected to increase streamflow response for several decades. Post thinning growth of coppice and regrowth may be two to five times that of the trees retained after thinning, with water use expected to be even higher. Preventing the development of coppice and regrowth into saplings (until they are needed for regeneration) is more critical to increasing streamflow from thinning than is the intensity of the initial thinning (Reed *et al.* 2012). Observation indicates that understorey and second-storey species respond positively to the lower stand density, but the extent to which this adversely affects streamflow is unknown at present.

Hydrological modelling undertaken as part of the Wungong Project (Croton *et al.* 2001b; Water Corporation 2005; Reed *et al.* 2012) suggests that thinning to a LAI of 0.6 (basal area over bark ~ 10m^2 / ha) and thereafter maintaining an average LAI of 1.0 is required to restore the streamflow duration and groundwater conditions of the 1990s. This is substantially less than the current LAI (1.6 to 2) for a fully stocked jarrah forest. This reduction in LAI could be achieved by thinning to $10m^2$ / ha about every 10 years with regular control of coppice and regrowth. Modelling indicates that this would restore streamflow to about 70 per cent of 1990-2000 levels after 20 years.

Trials to reduce forest density to increase water yield for potable domestic consumption were initiated by the Water Corporation in the Wungong catchment in the northern jarrah forest. The strategy involved thinning to be followed by coppice and regrowth control and included the monitoring of a variety of environmental effects (Water Corporation 2005). Early trials showed limited but positive results in terms of increased streamflow and a rise in the groundwater levels (Reed *et al.* 2012). Since 1987, thinning (to date without coppice and regrowth control) to increase water yield has been undertaken in two small jarrah catchments in southern forests near Manjimup. Both catchments demonstrated an initial positive response to thinning (Bradshaw 2010). The extent to which a subsequent reduction in streamflow is due to regrowth and coppice development or to a disconnection of the groundwater is unknown.

The increasing area of bauxite mine rehabilitation has the potential to significantly reduce water yield (Croton *et al.* 2005). Attention is now being directed toward the management of vegetation density in mine rehabilitation to reduce water use while maintaining the potential for a range of other values.

Under a medium severity scenario for climate change (CSIRO 2007), the percentage of jarrah forest where there is negligible streamflow (<700mm / annum) could increase from nine per cent to 18 per cent, consequently increasing the number of perennial streams becoming seasonal (Maher *et al.* 2010).

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 jarrah 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 therefore 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 jarrah forest in the FMP area is projected to increase over the period of the FMP (Department of Environment and Conservation 2013).

5 Integrated management

A number of factors need to be evaluated and weighed in the development of management approaches 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 effect of silvicultural practice on each forest value is important for making objective decisions about competing goals and values. 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 considered in the development of any silvicultural practice.

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

Reduced stand density (to a point) 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 stand growth.

Water production is enhanced at lower stand density. However, low stand density may reduce water quality if it increases the risk of stream salinity in some parts of the catchment, though this issue has become 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 at some stage provision needs to be made for recruitment and stand replacement as mature trees senesce and die.

While wildlife may benefit from the retention of more habitat trees, timber production is disadvantaged where retained habitat trees make up a significant proportion of the retained density. Retained habitat trees either displace crop trees or inhibit regeneration.

Biodiversity in the understorey and the streams may be advantaged by reduced stand density, especially in the advent of reduced rainfall, though it would be disadvantaged if it increased the impact of *Phytophthora cinnamomi.*

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

Thinning to produce bio-energy has the potential to reduce greenhouse gas emissions and is compatible with improving forest health, streamflow and groundwater and growth rate. Thinning could assist in maintaining biodiversity under conditions of a drying climate, as long as adequate provision is made for maintenance of habitat and protection of soils.

Forest management often requires managing competing forest values. Managing competing values requires a clear enunciation of the objectives for each of the values in question and an understanding of the effect of silvicultural practices on each of them.

Appendices

Appendix 1 Changes in silvicultural practice

Silvicultural systems and methods

Classical silvicultural systems are generally divided into those that are designed to create and maintain even-aged stands (clearfelling systems) or uneven-aged stands (selection systems) (Smith 1986; Florence 2004). The word ‘system’ is used when there is a planned series of treatments over time for tending, harvesting and re-establishing a stand. The word ‘method’ is used when referring to a single treatment, for example shelterwood, at a specific point in time. The essential difference between even-aged and uneven-aged systems is the size of the gap created in the canopy by timber harvesting 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 degrees and extent of disturbance created in the regeneration process and a varying extent and intensity of the associated edge effects (Bradshaw 1992). Within any of these various sources of regeneration may be employed (seeding, planting, lignotuberous advance growth or coppice in the case of the jarrah forest). 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 may also be supplemented with various forms of ‘legacy tree retention’ i.e. the retention of individuals or groups of trees or patches of older forest designed to provide older elements of the forest during the extended period before the regenerating forest again provides these values (Florence 2004; Forestry Tasmania 2009).

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

Wide varieties of silvicultural practices have been employed in the jarrah forest since the 1870s and are summarised below. Since a true silvicultural system has not been applied through time, the use of the term method is more appropriate when discussing jarrah forest silviculture. In the past elements of all of the above silvicultural methods have been used in combination. The end results of most applications still exist and provide an invaluable insight into the expected outcome of current practice. The structure that has resulted from those practices in the past has a major influence on the choice of current practice.

While silvicultural practices involve the manipulation of natural processes it cannot emulate nor does it attempt to emulate those processes in all respects. Nevertheless systems that work with, rather than against, natural processes, and are within the range of what can occur naturally, are likely to be the most successful for a range of objectives. The extent to which current silvicultural practices conform to the principles of what has become known as 'ecological forestry' has been evaluated by Stoneman (2007).

Past silvicultural practice

Silvicultural practices employed in the jarrah forest fall into several distinct periods and are described in detail by Bradshaw (1999) and (Stoneman *et al.* 2005). The location of earlier timber harvesting has been summarised by Heberle (1997). Past practices are summarised below.

1870-1920

This was a period of largely uncontrolled timber harvesting for sawlogs, mostly in the western side of the northern jarrah forest. Better quality forest was effectively clearfelled while poorer forest was selectively cut for the better quality trees. Regeneration from the existing lignotuber pool was generally successful but subsequent bushfires damaged much of it.

1920 –1940

Following the passing of the Forests Act in 1918, treemarking and 'tops disposal' were gradually introduced and plans setting out the objectives and practices were progressively developed for each Working Circle. Group selection was most commonly employed. This involved the retention of groups of predominantly immature trees; the thinning of groups of poles and saplings; and the release of regeneration in the gaps or 'blanks' created by the removal of mature sawlog trees. Cull trees were ringbarked (until 1935), banksia that might inhibit regeneration was felled, damaged saplings were 'coppiced' and where ground coppice were insufficient cull trees were partially ringbarked to promote seed production. In the absence of knowledge about seedling and lignotuber development there was an emphasis on using stump coppice as the most reliable source of regeneration (Kessell 1931). The area was burnt and protected from fire until the regrowth was old enough to withstand prescribed burning. Narrow strips about 100m wide that separated the compartments of regrowth were prescribe burnt to protect the regrowth.

Gap sizes were highly variable, from two tree heights to contiguous areas of more than one thousand hectares (Stoate 1923; Forests Department 1927a, 1927b). In addition to treating areas that were currently being harvested, areas cutover prior to 1920 were also treated and by the end of the period 169,000ha had been treated for regeneration or thinned, a program made possible by the large workforce made available during the Great Depression (Bradshaw 2005). A range of treatments undertaken in this period is illustrated in Figure 31.

The stand structure typically resulting from group selection is predominantly a mixture of structure type a), b) and c) illustrated in Figure 30. Structure type d) is more common in poorer quality forest and in areas not afforded the intensive regeneration treatment described above.

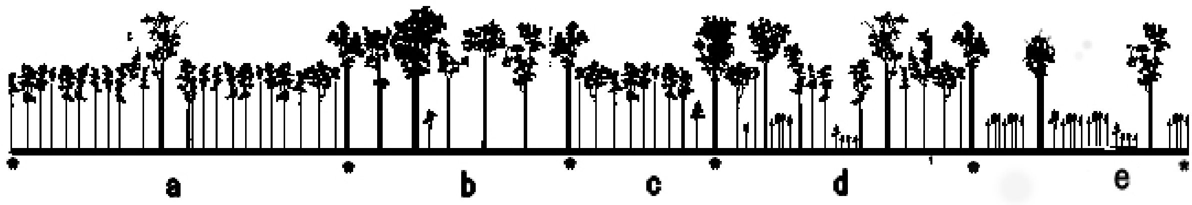


Figure 30: The range of stand structures found in the jarrah forest. a) large groups of pole-sized trees resulting from the removal of groups of mature trees, with occasional mature trees remaining. b) groups of mature trees retained during the previous harvest, some of which were immature at the time. 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 habitat trees retained.

Figure 31: The range of silvicultural treatments undertaken between 1920 and 1940.
Top: Culling and regeneration of large clearfelled areas created before 1920.
Centre: Regeneration 'cleaning' in gaps created under a group selection system.
Bottom: Thinned regrowth.

1940 to mid-1960s

Following WW II, demand for timber increased for the post war reconstruction and several new sawmills were established in the virgin southern jarrah forests. Sleeper hewing, the main means of production of railway sleepers until that time, ceased in 1945.

The intensive regeneration and culling work had ceased with the outbreak of war due to lack of funds and labour. After the war, the method began to change and by 1950 it had changed to what was effectively a single tree selection method aimed at removing the least thrifty trees. The area cutover each year increased in consequence of the less intensive timber harvesting. Culling was not resumed. The main reasons for the change were a desire to open up the forest and provide access for fire control (to be assisted by the construction of the mill railway network); a conviction that cull trees would eventually be saleable; and a general lack of funds and labour. The increased area being cutover meant that the intensive fire protection strategy of previous decades was no longer achievable.

The stand structures (Figure 30) that typically resulted up until the 1950's were:

- groups of mature trees retained during the previous harvest, some of which were immature at the time
- smaller groups of pole-sized trees resulting from the removal of several mature trees
- multi-aged stands resulting from the removal of single trees.

After the 1950's, multi-aged stands resulting from the removal of single trees predominated.

Mid-1960s to mid-1980s

While the intention of the fire protection strategy for regrowth produced up to the 1940s was to burn the regrowth when it could withstand fire, achieving that aim was more difficult. A lack of resources and the heavy fuel that had accumulated under the regrowth over the previous 40 years provided few opportunities for safe prescribed burning of the fire breaks. The heavy fuels also made it difficult to burn under the regrowth and little fuel reduction burning had been achieved by 1960 (Harris 1975). Much of the protected regrowth was burnt in the 1961 Dwellingup bushfire. Following the Royal Commission into that fire, prescribed burning was increased throughout the forest and regeneration resulting from timber harvesting was no longer afforded a fire-free protection period.

The identification of *Phytophthora cinnamomi* as the cause of jarrah dieback in 1965 led to a change in policy aimed at reducing the area cutover each year and hence the area put at risk of infection (Batini *et al.* 1972). To this end all sawlogs were removed from harvested areas from 1970. Culls were not removed and there was no attempt to create distinct gaps for regeneration.

During the period some 15,000ha of the pole stands that had originated in the 1920s were thinned, this time using herbicides that prevented the development of coppice from the thinned trees (Kimber 1967).

Towards the end of this period marri woodchip was removed from some areas in southern forests resulting in more intensive timber harvesting of those areas.

The stand structures that typically resulted from this period were multi-aged stands resulting from the removal of single trees and some large groups of saplings with habitat trees retained (Figure 30).

Bauxite mining in the northern jarrah forest began in 1962 and mined areas were rehabilitated with dieback resistant exotic eucalypts until 1988.

Towards the end of this period multiple use management was being more explicitly defined and objectives for wildlife, water and timber production, catchment protection and recreation were being developed (Forests Department 1977).

Mid-1980s to 2003

A review of silvicultural practice in the mid-1980s led to significant changes in practice. While most of the many multiple use values had been catered for in the past, largely by default, the potential for more intensive management made it imperative that they be more clearly enunciated and specifically catered for. An examination of stands resulting from the variety of past practices showed that those that could be most successfully managed into the second timber harvesting cycle were those where distinct groups of even-aged patches had been created by timber harvesting and subsequent culling. These were stands that were harvested or treated in the period 1920-1940. The gaps created were large enough to allow regeneration to develop without excessive suppression from the larger trees and most importantly they were large enough that the regrowth did not suffer excessive damage from the felling of surrounding trees in subsequent felling cycles. The opposite was true for the areas harvested under subsequent methods.

The revised methods involved the thinning of patches of regrowth that had resulted from the regeneration work of the 1930s; the creation of gaps of sufficient size to allow ground coppice to be released and protected from future harvest; and the partial harvesting of patches with insufficient advance growth in order to establish seedling regeneration. The emphasis was on creating manageable patches of forest, each with a single management objective (Bradshaw 1985)(Figure 32). Culling was re-introduced; this time using herbicides rather than ringbarking and released regeneration was protected from fire for 10-15 years. In effect this was a re-introduction of the 1930s practices with some modifications.

Areas that were infected with *Phytophthora cinnamomi* were selection cut with an emphasis on retaining resistant trees species such as marri and blackbutt.

Amendments to practice were progressively made to give specific attention to other values in both the planning stage and in silvicultural practice. During this period the management purpose of significant areas was changed from multiple use to conservation. Changes within the multiple use forest included the retention of undisturbed buffers around all streams and some roads; the retention of habitat trees, logs and understorey in harvested areas; varied thinning intensity and gap sizes in different visual amenity zones; and varied basal area retention in zones of different salinity risk. The timing and prescription references for these changes can be found in (Stoneman *et al.* 2005).

From 1988 bauxite mine rehabilitation was undertaken with planted or seeded native species, principally jarrah, marri and blackbutt.

2003 to 2013

Further changes occurred as a part of the development of the FMP 2004-2013. The most significant of these were the large increase in formal conservation reserves, the creation of Fauna Habitat Zones and the reservation of all old-growth forest. These measures increased the area of formal conservation reserves and forest conservation areas to 1,264,100ha and reduced the area of State forest and timber reserves to 1,209,600, an increase of 58 per cent and decrease of 37 per cent respectively, relative to 1982 levels (Conservation Commission 2004).

Other changes to practice included an increase in the number of habitat trees retained per hectare, a limitation on culling in gaps and a definition of dieback risk in terms of vegetation complex for application of the dieback prescription. Changes to the eastern jarrah forest included a reduction in the retained basal area of shelterwood, a decrease in the regeneration stocking requirements and increase in the use of coppice for regeneration (Conservation Commission 2004; Department of Environment and Conservation 2004; Stoneman *et al.* 2005).

Changes from 2014

The principle changes to silvicultural practice in the FMP (Conservation Commission 2012a) include the following:

- Greater attention to the retention of marri for habitat, particularly in areas where marri occurs in low numbers.
- Removal of phased harvesting from areas no longer regarded as a salinity risk i.e. most of the eastern jarrah forest except for a part of the Warren Region.
- Provision for thinning to reduce moisture stress under a drying climate aimed at protecting ecosystem health.
- Provision for the use of thinning to increase water yield from harnessed catchments.
- Refinement and clarification of existing silvicultural practices.

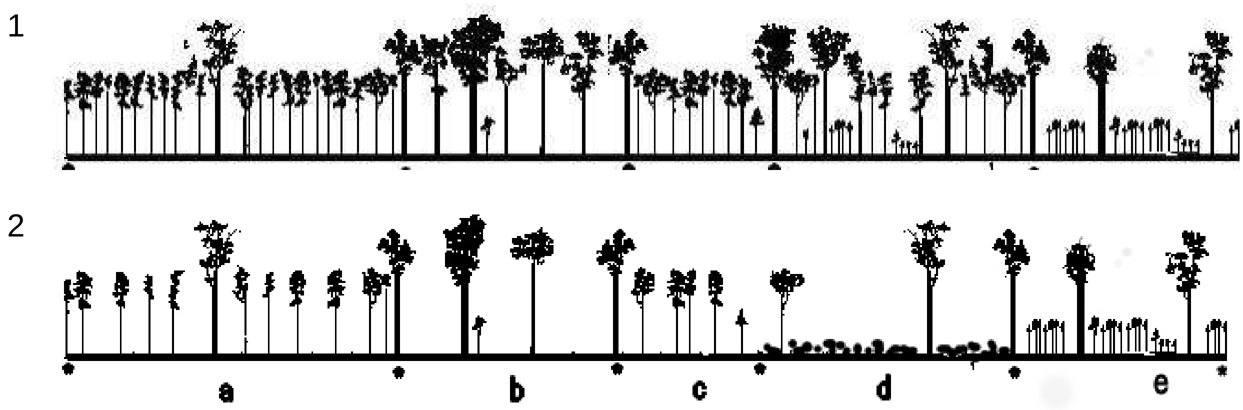


Figure 32: Diagrammatic illustration of the typical silvicultural methods (1. Before treatment, 2. After treatment) applied to stands of different structure under the methods applying since the mid-1980s. a) and c) Groups of poles are thinned to promote the growth of retained trees. b) Stands of trees not suited to thinning but where the advance growth is inadequate are partially harvested to retain trees as a seed source and as a 'shelterwood' until the seedlings develop into ground coppice. d) Stands of trees not suited to thinning but with adequate advance growth are 'cut to a gap'. Saplings and poles that have developed in small canopy gaps cannot be protected from the harvesting of the large trees. e) Recently harvested sapling stands are not treated but will be thinned at a later date. Habitat trees are retained throughout. To complete the operation to this stage some non-commercial culling is usually required as a follow-up to commercial operations.

Trends in harvest intensity

Harvest intensity can be expressed in terms of the gap that is made in the canopy with each timber harvest or silvicultural activity. Thinning is intended to retain a complete canopy at a reduced density. Other timber harvesting activities are designed to create gaps in the canopy for regeneration and vary in size from two tree heights (0.3 ha) to many hectares. Figure 33 shows the trend in canopy disturbance in the key periods described above. This indicated that the most intensive disturbance took place in the period before 1940 when large areas of virgin forest were harvested and regenerated. There has been a trend to more conservative treatment since that time (Figure 34). Much of the activity in the most recent period has been in stands that were previously harvested. Figure 34 shows the changes in the silvicultural method of timber harvesting since 1987.

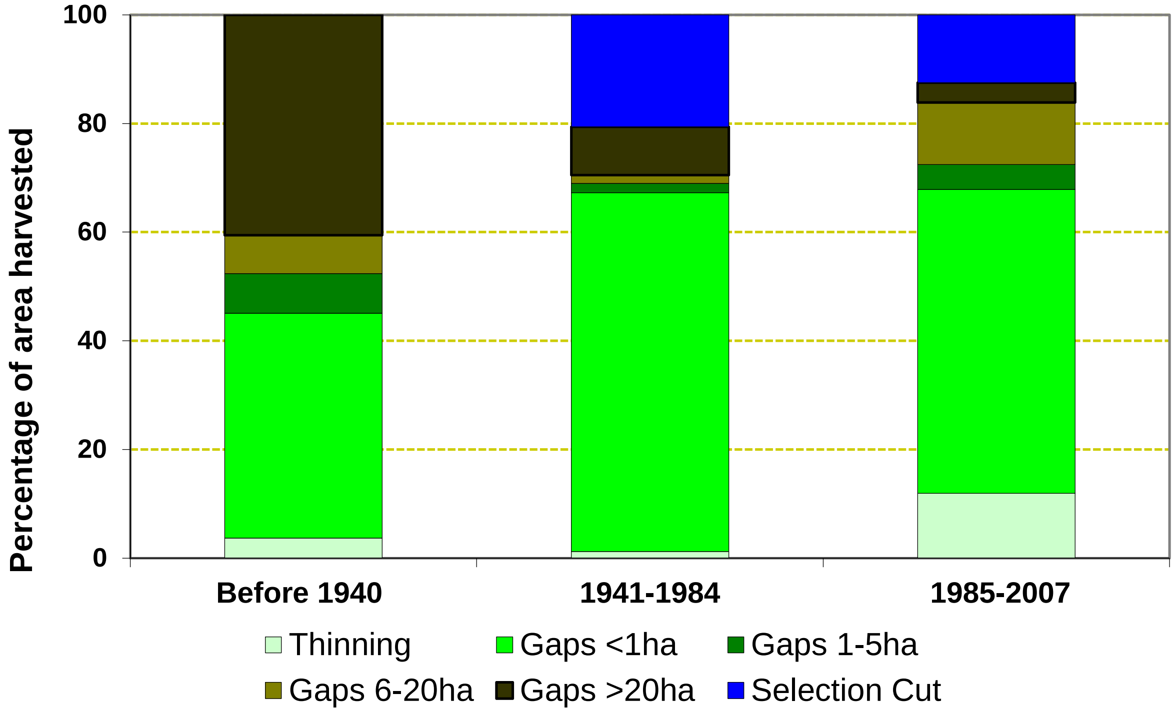


Figure 33: Intensity of timber harvesting that has been applied in the jarrah forest. Information for the period to 1940 has been largely interpreted from API data. Later information mainly relies on records of known activity (Source: (Bradshaw 1999).

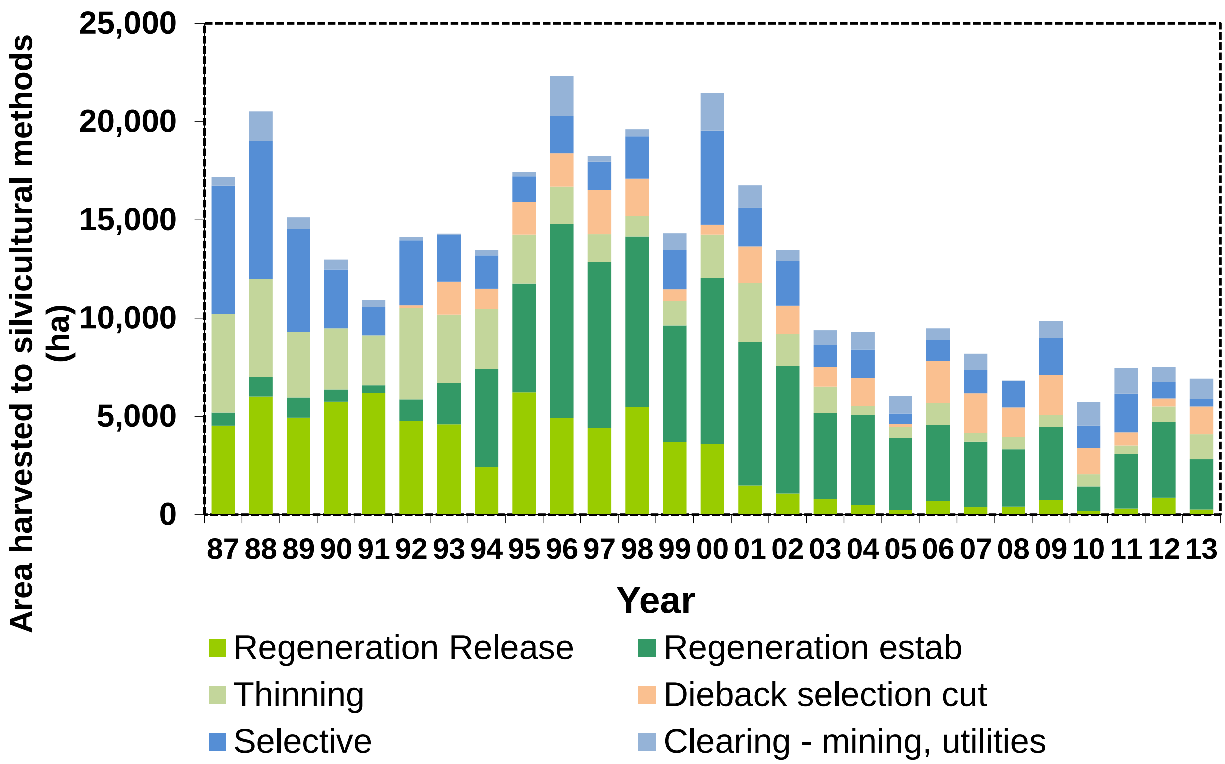


Figure 34: Trends in the area and method of jarrah harvesting for 1987 to 2011. Selective harvest refers to areas that have been harvested, but where too many culls remain to achieve the stated silvicultural method.

Appendix 2 Vertebrate fauna that may be affected by timber harvesting

The species listed in the following tables occur in jarrah 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 WA 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 a special habitat requirement
- indirect predicted impact of timber harvesting e.g. increased in fox predation.

After consideration of these factors, the FDIS report (Christensen *et al.* 2005) highlighted only 13 species where species-specific measures were recommended in addition to those required by the FMP or the jarrah silvicultural guidelines. These 13 species are indicated with an asterisk in the following tables. This information was current as of December 2014.

Arboreal marsupials and terrestrial mammals

Fauna group	Common Name	Scientific Name	Mean Body Mass (g)	Diet	Breeding	Population Status
Arboreal mammals	*brush-tailed phascogale	*Phascogale tapoatafa*	230 males 160 females	Invertebrate's small birds and small mammals.	Tree hollows.	Vulnerable
	red-tailed phascogale	*Phascogale calura*	39-68 males 38-48 females	Invertebrate's small birds and small mammals.	Tree hollows and limbs.	Endangered
	*****western ringtail possum	*Pseudocheirus occidentalis*	800 -1 130	*Agonis flexuosa* leaves and, leaves of myrtaceous species.	Tree hollows, ground hollow or balga.	Vulnerable
	common brushtail possum	*Trichosurus vulpecula*	1 300-4 500 male 1 200-3 500 females	Herbivore and tolerant to poisonous native plants.	Tree hollows.	Not listed
	western pygmy possum	*Cercartetus concinnus*	8-21	Flowers and pollen.	Tree hollows.	Not listed
Terrestrial mammals	*chuditch	*Dasyurus geoffroii*	700-2000 males 600-1120 females	Opportunistic feeders, invertebrates to small mammals and birds.	Hollow logs.	Vulnerable
	*numbat	*Myrmecobius fasciatus*	405-752 males	Termites.	Hollow logs.	Vulnerable
	quenda or southern brown bandicoot	*Isoodon obesulus*	550-1 850 males 400-1 200 females	Omnivore.		Priority 5
	*****quokka	*Setonix brachyurus*	2 700-4 200 males 1 600-3 500 grams	Herbivore.	Dense understorey or swamp thickets.	Vulnerable
	*****tammar wallaby	*Bettongia penicillata*	6 000-10 000 males 4 000-6 000 females	Herbivore.	Fire regenerated thickets.	Priority 5

Fauna group	Common Name	Scientific Name	Mean Body Mass (g)	Diet	Breeding	Population Status
	water-rat	*Hydromys chrysogaster*	400-1 275 males 340-992 females	Omnivore.		Priority 4
	western brush wallaby	*Macropus irma*	7 000-9 000	Herbivore.		Priority 4
	*woylie	*Bettongia penicillata ogilbyi*	1 100-1 600	Herbivore.		Endangered

Bats

Fauna group	Common Name	Scientific Name	Mean Weight (g)	Foraging	Roosting	Breeding	Population Status
Bats	chocolate wattled bat	*Chalinolobus morio*	8-11	Between understory and canopy.	Hollows, caves, logs.	They give birth in October.	Not listed
	Gould's long-eared bat	*Nyctophilus gouldi*	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	*Chalinolobus gouldii*	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
	lesser long–eared bat	*Nyctophilus geoffreyi*	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

Fauna group	Common Name	Scientific Name	Mean Weight (g)	Foraging	Roosting	Breeding	Population Status
	south western freetail-bat	*Mormopterus spp.*	8-10.5	Above the canopy uncluttered habitat.	Tree hollows.	Young born early summer.	Not listed
	southern forest bat	*Vespadelus regulus*	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
	western false pipistrelle	*Falsistrellus mckenziei*	17-26	Between crown break and understory canopy.	Tree hollows, or ground logs on ground where tree hollows not available.	Reproductive biology has not been studied.	Priority 4
	western long–eared bat	*Nyctophilus spp.*	11-20	Amongst shrub stratum.	Tree crevices, foliage and under bark.	Reproductive biology has not been studied.	Not listed
	white-striped freetail-bat	*Tadarida australis*	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
Mid-storey birds	Australian owlet-nightjar	*Aegotheles cristatus*	Predominantly feed on insects either on the wing or by pouncing upon they prey from their current position.	Medium	Not listed
	barking owl	*Ninox connivens*	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	*Tyto alba*	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
	southern boobook	*Ninox novaeseelandiae*	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
	masked owl	*Tyto novaehollandiae*	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
	elegant parrot	*Neophema elegans*	Seeds from grasses and herbaceous plants as well as fruit from fruit trees.	Small	Not listed
	rufous treecreeper	*Climacteris rufa*	Insects, ants, snails, small reptiles and seeds.	Small	Not listed
	sacred kingfisher	*Halcyon sancta*	Mainly feed on land occasionally capturing prey on the water feeding on reptiles, insects and larva.	Small	Not listed
	western rosella	*Platycercus icterotis*	Seeds, grass, fruit and flowers they feed on the ground in open areas or in trees.	Small	Not listed
Canopy birds	Baudin's cockatoo	*Calyptorhynchus baudinii*	Eucalyptus fruit and flowers while occasionally feeding on insect larva.	Large	Endangered
	red-tailed black-cockatoo	*Calyptorhynchus banksii naso*	Feeds on fruit, seeds and flowers e.g. Eucalyptus and Banksia, occasionally feeds on insect larva in trees.	Large	Vulnerable
	Australian ringneck	*Barnardius zonarius*	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
	striated pardalote	*Pardalotus striatus*	Feed in the foliage in the canopy on insects and the insect larva.	Small	Not listed

Aerial hawking birds	dusky woodswallow	*Artamus cyanopterus*	Insects and nectar from flowers.	Small	Not listed
	nankeen kestrel	*Falco cenchroides*	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
	peregrine falcon	*Falco peregrinus*	Birds, rabbits, marsupials as well as other day active mammals.	Medium	Specially protected
Non-forest birds	Carnaby's black-cockatoo	*Calyptorhynchus latirostris*	Nuts and fruit from native and introduced plants, along with insect larva.	Large	Endangered
	Muir's corella	*Cacatua pastinator*	Mainly on Eucalyptus seeds and flowers but will eat larvae, bulbs and stock feed.	Large	Specially protected
	tree martin	*Cecropis nigricans*	Insects from open areas, generally captures the insects on the wing.	Small	Not listed
Seasonal birds	purple-crowned lorikeet	*Glossopsitta porphyrocephala*	Lives solely in trees in which it feeds on the fruit.	Small	Not listed
	red-capped parrot	*Purpureicephalus spurius*	Seeds from Eucalyptus trees as well as grasses and other understory species	Small	Not listed
Water birds	Australian shelduck	*Tadorna tadornoides*	Open paddocks, grasslands and waterways concentrating on insects and molluscs.	Large	Not listed
	Pacific black duck	*Anas superciliosa*	Vegetation consisting of seeds and aquatic plants and insects sought on the water occasionally.	Medium	Not listed
	chestnut teal	*Anas castanea*	Omnivorous birds mainly feeding around the edge of water on a rising tide searching for exposed food items	Medium	Not listed
	grey teal	*Anas gibberifrons*	Around open water food consists of plants, insects small vertebrates. They also filter water or mud with their bill.	Medium	Not listed
	Australian wood duck	*Chenonetta jubata*	Feeds mainly on grassed areas and occasionally in shallow water, eating grasses and herbs.	Large	Not listed

Birds (not hollow using)

Fauna group	Common name	Scientific name	Foraging	Nesting	Population status
Non-hollow nesting birds	Australasian bittern	*Botaurus poiciloptilus*	Forage mainly at night on a range of animals, including birds, mammals, fish, frogs, reptiles, crustaceans and insects.	Nests built on platform of reeds in secluded places.	Endangered
	Australian bustard	*Ardeotis australis*	Omnivorous picking at food as they slowly wander across the landscape.	On bare ground on sandy ridges amongst protective vegetation.	Priority 4
	black bittern (SW population)	*Ixobrychus flavicollis*	Feeds on fish and amphibians.	Nest built on a branch overhanging water.	Priority 3
	bush stonecurlew	*Burhinus grallarius*	Largely feeds on insects, molluscs, reptiles and seeds, it will occasionally take small mammals and marsupials.	Built beside logs in sparse grass or herb understorey.	Priority 4
	crested shrike-tit (SW Subsp)	*Falcunculus frontatus leucogaster*	Has a preferred diet of insects but will feed on fruit and seeds.	High in a tree generally in a fork.	Priority 4
	eastern curlew	*Numenius madagascariensis*	They eat crabs and molluscs foraging at night, sometimes during the day with slow deliberate movements.	Shallow depression lined with grass.	Priority 4
	hooded plover (W subsp)	*Charadrius rubricollis rubricollis*	Insects and some small molluscs	Sandy scrape in the beach just above the high water mark generally before the dunes.	Priority 4
	little bittern	*Ixobrychus minutus*	Small aquatic invertebrates, tadpoles and insects.	Nests on vegetation sometimes above the water.	Priority 4
	*****malleefowl	*Leipoa ocellata*	Opportunistic feeders on what food source is abundant at the time.	Builds a mound nest and they bury the eggs.	Vulnerable
	*****noisy scrub-bird	*Atrichornis clamosus*	Insectivorous	Made from pliable long leaved sedges in clumps of shrubs or piles of debris.	Endangered

Fauna group	Common name	Scientific name	Foraging	Nesting	Population status
	white-browed babbler (W Wheatbelt)	*Pomatostomus superciliosus ashbyi*	Diet consists of insects, small reptiles, invertebrates, crustaceans, seeds and nuts.	In forks of trees.	Priority 4
	western bristlebird	*Dasyornis longirostris*	Insectivorous.	Made from pliable long leaved sedges in clumps of shrubs.	Vulnerable
	Australian painted snipe	*Rostratula benghalensis australis*	Wetland invertebrates.	Ground scrapes or mounds in water.	Endangered
	rufous-fieldwren	*Calamanthus campestris montanellus*	Insects , spiders and other small creatures.	Grass nest at or near ground.	Priority 4

Reptiles and amphibians

Fauna group	Common name	Scientific name	Activity pattern	Foraging	Common shelter site	Breeding	Population status
Reptiles	Bunbury skink	*Glaphyromorphus “koontoolasi”*	Unknown	Unknown.	Unknown.	Unknown.	Priority 1
	carpet python	*Morelia spilota imbricata*	Mainly nocturnal	Strangle and swallow their food whole, mammals, birds and reptiles.	Tree hollow, caves.	14-35 eggs laid in a sheltered site late spring, eggs take 63-71 days to hatch.	Priority 4 and Specially Protected
	dell's skink	*Ctenotus delli*	Unknown	Insects.	Leaf litter.	Unknown.	Priority 4
	jewelled south-west ctenotus	*Ctenotus gemmula*	Unknown	Insects.	Leaf litter.	Unknown.	Priority 3

Fauna group	Common name	Scientific name	Activity pattern	Foraging	Common shelter site	Breeding	Population status
	short-nosed snake	*Elapognathus minor*	Unknown	Invertebrates and small reptiles.	Leaf litter.	Gives birth to live young (approx. 10 per litter).	Priority 2
	southern death adder (SW Population)	*Acanthophis antarcticus*	Nocturnal.	Insects, small mammals and marsupials.	Leaf litter shrubs.	Gives birth to 10-30 live young in late summer.	Priority 3
	woma or Ramsay's python	*Aspidites ramsayi*	Nocturnal.	Vertebrates (mammals birds, reptiles).	Hollow logs, caves, animal burrows, thick vegetation.	Eggs laid September-October and young born January.	Priority 1 and Specially protected
Amphibians	***Nornalup Frog	*Geocrinia lutea*	Calls only at night although active both day and night.	Invertebrates.	Dense vegetation adjacent to streams and tunnels of matted vegetation and clay.	The females lay their eggs in a burrow or shallow chamber.	Priority 4
	***orange-bellied frog	*Geocrinia vitellina*	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
	***sunset frog	*Spicospina flammocaerulea*	Calls only at night although active both day and night	Invertebrates.	Swamp habitat.	Eggs laid in shallow still slowly flowing water late spring early summer.	Vulnerable
	***white-bellied frog	*Geocrinia alba*	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	pouched lamprey	*Geotria australis*	Occur in mud burrows upper reaches of coastal streams for their first 4 years before migration to the ocean.	Generally 400-550 long although can grow longer.	Stop feeding while migrating up stream.	Spawn in fresh water.	Priority 1
	mud minnow	*Galaxiella munda*	Survives for 1 year and breeds in winter spring period.	Maximum of 58.	In areas where water is fast flowing and there is submerged vegetation feeding on insects and larvae.	Eggs laid in flooded vegetation.	Vulnerable
	black-stripe minnow	*Galaxiella nigrostriata*	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
	Balston's pygmy perch	*Nannatherina balstoni*	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

Glossary

Advance growth	A term used to describe the established regeneration stages, including established seedlings, coppice, saplings and poles (Jacobs 1955). In jarrah forest the term is used to refer to the ground coppice stage of regeneration (Abbott *et al.* 1982).
Basal area	The sum of the cross-sectional areas of trees in a given stand measured at 1.3 metres above the ground. It is usually expressed as square metres per hectare.
Biological diversity (Biodiversity) (described in CLM Act)	The variability among living biological entities and the ecosystems and ecological complexes of which those entities are a part and includes: (a) diversity within native species and between native species; (b) diversity of ecosystems; and (c) diversity of other biodiversity components.
Bole	The tree trunk from the ground to the crown break. The bole does not include the major branches supporting the crown.
Breast height	The standard point of tree diameter 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.
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 of ground 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 with no saleable value due to faults or degrade.
Culling	The reduction in the density of unwanted vegetation, usually to reduce competition to retained crop trees or for establishing or releasing regeneration.
Cutting cycle	The planned interval between partial harvests in an uneven-aged stand.
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 95 per cent of optimum density.
Density, maximum	The maximum density that can be achieved by a stand of a particular age and site before competition-induced mortality occurs.
Density, optimum	The stand density at which maximum stand volume or basal area growth occurs.
Density, suppression	The density at which growth in stand volume or basal area growth begins to slow as it approaches maximum density.

Dieback (Phytophthora dieback)	In the south-west of Western Australia a disease of plants caused by infection by the soil-borne organisms of the genus Phytophthora, of which P. cinnamomi is the most widespread.
Diverse ecotype zone (DEZ)	Open jarrah forest (<30 per cent canopy cover), flats, sedgelands, rock outcrops and swamps excluded from timber harvesting.
Epicormic	Vegetative shoots arising on the tree 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.
First order stream	A stream which does not have any other streams feeding into it.
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 (regeneration establishment)	A discrete opening in the overstorey canopy that reduces competition and allows seedlings to become established and / or develop.
Gross bole volume	The volume of a tree from ground level to the break of the crown or to 100mm diameter where the tree has not yet established a clear crown break.
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 other animals.
Harnessed catchment	A catchment that is used for water supply purposes.
Hybrid	The progeny produced from a cross between two genetically different plants, usually different species.
Impact - dieback	The effect on vegetation from the presence of Phytophthora species, referred to as either predicted or current impact.
Influence (forest influence, edge influence)	The biophysical effects of the residual trees on the surrounding environment, including effects on microclimate, light availability, seed and litter-fall and evapotranspiration.
Landscape management units (previously referred to as Landscape conservation units)	An agglomeration of vegetation complexes and ecological vegetation systems, as defined and mapped by Mattiske and Havel (2002), to form more compact management units that recognise the underlying ecological characteristics. See Appendix 3.
Landscape heterogeneity	The diversity, size and spatial arrangement of habitat patches at the landscape scale. A landscape that is heterogeneous is considered to provide the diversity of resources required by some fauna species, such as hollows in large trees, food sources, and structural components of the vegetation, are all provided within an area of a size that is readily and efficiently accessed by colonies of fauna.
Leaf area Index	Index of the one-sided green leaf area per unit ground surface area in broadleaf canopies.
Legacy tree (also referred to as Overwood)	Trees (usually mature to senescent, including habitat trees) retained from the previous stand within a regrowth 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.

Lignotuber pool (Advance growth)	The full range of lignotuber stages present at any one time. Includes lignotuberous seedlings, seedling coppice, incipient ground coppice and dynamic ground coppice.
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.
Overstorey	Species comprising the upper canopy layer of the forest. Common overstorey species include *Eucalyptus marginata*, *Corymbia calophylla, E. wandoo, E. patens* and *E. diversicolor*.
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).
Prescribed burning	The controlled application of fire under specified environmental conditions to a predetermined area and at the time, intensity and rate of spread required to attain planned resource management objectives.
Recruitment	The process of plants growing and moving from one growth stage or a subsequent stage.
Regeneration (noun)	Naturally or artificially established young trees.
	The renewal of a tree crop by natural means or through human assistance.
Regrowth	A term covering a range of forest growth stages before maturity.
Rehabilitation	The process necessary to return disturbed land to a predetermined surface, vegetation cover, land-use or productivity.
Reserve – formal	One of the land category categories of national park, nature reserve, conservation park, or CALM Act sections 5(1)(g) or 5(1)(h) reserves for the purpose of conservation.

Resilience	The capacity of an ecosystem to withstand external pressures and, over time, return to its prior condition, including its ability to maintain its essential characteristics such as taxonomic composition, structural forms, ecosystem functions and processes (adapted from (Thompson *et al.* 2009), who cite (Holling 1973).
Rotation - timber	The period between regeneration establishment and the final harvest.
Rotation - physiological	The number of years between regeneration and death due to old age.
Salinity	The mobilisation of stored salt to the surface soil groundwater leading to increased salt concentrations in water courses.
Sawlog	A log suitable for processing into sawn timber.
Scorch	Injury to foliage as a result to exposure to heat from fire.
Second order stream	In the classification of streams when two first-order streams come together, they form a second-order stream. If a first-order stream joins a second-order stream, it remains a second-order stream. It is not until a second-order stream combines with another second-order stream that it becomes a third-order stream.
Second order catchment	The catchment associated with a second order stream.
Second-storey	The structural layer between the shrub and herb storey and the overstorey (canopy). In the jarrah forest, this layer often includes species such as *Banksia litoralis*, *B. grandis, Allocasuarina* spp., *Xylomelum occidentale, Persoonia longifolia, Melaleuca* spp. and *Acacia acuminata.*
Selection cutting	The removal of selected trees from a stand on a tree-by-tree basis rather than a group basis. This silvicultural method is used in the jarrah forest to retain and promote resistant species and individuals in *Phytophthora* dieback infested forest.

Shelterwood (regeneration establishment)	A jarrah silvicultural treatment that involves a partial reduction in the density of overstorey trees and action to establish regeneration under the remaining mature trees.
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.
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 undergoes 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.
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.
Stand age	The age of the oldest dominant cohort in a stand.
Stand density	A quantitative measure of the amount of trees on a given area, expressed either absolutely in terms of number of trees, basal area, or volume.
Structure	When applied to a forest, is the horizontal and vertical distribution of the alive and dead vegetation.
Stocking	A measure of stand density in terms of the number of stems per hectare.
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 >1,000 stems / ha.

Stool coppice	A growth stage where shoots have developed from a stump cut off at ground level.
Stream reserve	Areas of forest including watercourses and riparian vegetation that has been set aside to provide forest undisturbed by timber harvesting; and to protect water quality, riparian vegetation and aesthetic values.
Structural complexity	The diversity in stand structure.
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.
Temporary exclusion area (TEAS)	An area that is excluded from timber harvesting for a particular period of time.
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.
Tops disposal	The removal of slash away from the base of retained trees to avoid damage in any subsequent fire.
Understorey	Herb and shrub layer. This vegetation layers occurs beneath both the overstorey and second-storey.
Virgin forest	Forest that has never been harvested or cleared. It may be of any age.
Wandering heart	The irregular plane of development of the pith and associated brittle or decayed wood within a tree caused by bends or kinks in the tree during its early development.
Working circle	A term used in early forest administration to describe an area of forest managed under a separate management plan.

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