# Modelling and planning to increase future habitat of the Red-tailed Black-Cockatoo.

**Report prepared for:**

**Wimmera Catchment Management Authority**

December 2008

# Modelling and planning to increase future habitat of the Red-tailed Black-Cockatoo.


**Authors:**

**Martine Maron**
Australian Centre for Sustainable Catchments
University of Southern Queensland
Toowoomba QLD 4350

**Paul Koch**
Greening Australia (Vic.) Limited
147 Bahgallah Rd. Casterton VIC 3311

**Jeremy Freeman**
Greening Australia (SA) Limited

**Steffen Schultz**
Greening Australia (Vic.) Limited

**Peter Dunn**
**Armando Apan**
Australian Centre for Sustainable Catchments

# CONTENTS

# ACKNOWLEDGEMENTS

Thank you to the Australian Government for providing the funding for this work through the Natural Heritage Trust. The authors would like to thank landholders of the Wimmera, Glenelg-Hopkins and South East NRM regions for providing access to private property. Advice and comments from Luke Geelen, Richard Hill, Peter Menkhorst, Dean Robertson, Craig Whiteford, Peter Wilcock and the south-eastern Red-tailed Black-Cockatoo Recovery Team were particularly helpful in developing scenarios. Richard Hill provided access to unpublished datasets. Alison Howes, Dr. Clive McAlpine and Dr. Carl Smith provided advice on using Netica software. Alison Howes, Jarrod Kath, Wes Leedham, Keegan Roache and Roslyn Schumacher assisted with data processing, analysis and mapping. John Morgan and Andrew Bradey provided useful suggestions about Buloke revegetation. Dale Tonkinson and Nigel Willoughby provided a thoughtful and detailed review of an earlier draft of this report.

# EXECUTIVE SUMMARY


The nationally endangered south-eastern Red-tailed Black-Cockatoo *Calyptorhynchus banksii graptogyne* (RtBC) is an ecological specialist, feeding exclusively on the seeds of Brown Stringybark *Eucalyptus baxteri*, Desert Stringybark *E. arenacea* and Buloke *Allocasuarina luehmannii*. The ongoing decline in quality and quantity of food resources has been identified by the Recovery Team as the major threat to the cockatoo, while the availability of nesting hollows, although not currently limiting, may become important in the future as large hollow-bearing trees are lost.

Given significant ongoing investment in the protection of the RtBC's key resources, it is of critical importance that clear guidelines for the most efficient and effective targeting of investment are available. This study aimed to 1) quantify the extent and condition of habitat currently available to RtBCs; 2) identify how food availability has changed historically and more recently under various land management regimes; and 3) estimate how habitat extent and condition might change in the future under different scenarios of protection, enhancement and restoration.

Recent and historical aerial photography covering the range of the RtBC was used to map Buloke paddock trees, patches and roadside areas in 1963, 1997 and 2004, and field surveys were used to identify the size class distribution of existing Buloke trees in each of these categories. Within the study area, by far the majority of trees are estimated to be smaller than 10 cm DBH. However, due to the extremely high density of these young trees, which occur mainly on roadsides and in ungrazed patches, only a very small proportion will mature to sufficient size to provide a useful resource for the RtBC. The density of trees >30 cm DBH on roadsides is estimated already to be close to the maximum density recorded for mature trees in patches (84/ha). Few areas of vegetation dominated by Buloke were located in public land blocks, and Buloke trees in these areas also tended to be smaller and therefore less suitable for RtBC foraging. As a consequence, trees on private land provide the main Buloke food resource for RtBCs.

During the 41-year period from 1963 to 2004, there was a 48.3% loss of Buloke trees in paddocks, or 1.2% loss per year. The total area of Buloke woodland patches on private land decreased by 37.8%, but the area of roadside Buloke woodland more than doubled. Between 1997 and 2004, the number of paddock trees within the sampled area reduced by 9.8% over the seven years. The pattern of tree loss suggested that deliberate tree removal or altered land management practices contributed substantially to the losses of trees from cropping and grazing landscapes. Although centre pivot irrigation systems occupied only 3% of the sampled area, they accounted for 22% of paddock tree losses.

The availability of stringybark food resources is controlled not only by tree loss and recruitment but also factors affecting the fruit crop such as fire and grazing. The impact of prescribed burns and wildfires on food supply across the RtBC range was calculated. Some 11.1% of stringybark woodlands have been prescribed burnt in the past 10 years, and an additional 15.9% have been burnt by wildfire. The total reduction in food availability in woodlands attributable to prescribed burns is 5.4%, with a 13.2% reduction attributable to wildfire. Therefore, wildfire currently plays a substantially greater role in reducing seed availability than prescribed burns. As the frequency and extent of wildfires are likely to increase with climate change, protecting and increasing the availability of stringybark paddock trees may be an important strategy to reduce the risk of a fire-induced population crash.

In general, grazing of stringybark woodlands had little impact on both capsule density and crop size (as well as canopy health) which are key aspects of food supply, although grazing significantly improved food supply in *E. baxteri* woodlands. However, *E. arenacea* paddock trees were approximately 27 times higher and *E. baxteri* paddock trees approximately 12 times higher in crop size when compared with trees in adjacent ungrazed woodlands. Reduced light competition is likely to be a key factor driving this enhanced productivity, but other factors such as soil nutrient and moisture availability or even superphosphate application may be involved and further research is clearly warranted. The findings suggest that stock exclusion from stringybark remnants will have limited benefit to the RtBC. Instead, funding for RtBC recovery should be directed towards the protection and replanting of individual paddock trees or trees at low density, particularly in areas dominated by *E. arenacea*.

During the 57-year period from 1947 to 2004, there was a 44.6% loss of stringybark tree cover recorded from sample sites covering 2% of the RtBC range. Losses were similar between stringybark species, but were far more severe on private than on public land, and for paddock trees than remnant woodlands. When measured over the period 1992/97–2004, annual rates of loss were 0.04% for *E. arenacea* and 0.02% for *E. baxteri*. Recent habitat loss was associated primarily with paddock trees (0.26% per annum). Greater emphasis on the protection of stringybark paddock trees appears warranted, considering the disproportionate importance of such trees.

RtBCs nest in hollows in live or dead eucalypt trees, and any gum eucalypt which forms large hollows is probably suitable as nesting habitat. We considered current and past availability of living, large old gum eucalypts within the RtBC range. A separate study is required to factor in the effects of changes in availability of dead hollow trees on overall nest hollow availability. Paddock gum eucalypts were typically very large, with trunk diameters of between 140 and 200 cm most frequently encountered, but less than 5% of trees in woodland patches and roadside met or exceeded the mean size of trees in which RtBC nests have been recorded.

As with Buloke, the vast majority of live gum eucalypt trees occur on roadsides or in woodland patches, are relatively small (< 60 cm DBH) and are estimated to be younger than 100 years. However, these young trees occur at very high densities, and only a few can be expected to mature into large, hollow-bearing trees suitable for the RtBC.

Between 1947 and 2004, 30% of gum eucalypt tree cover in the sample area was lost, all from private land. Areas of paddock trees were more than thirty times as likely to be lost as areas of denser woodland. These paddock tree losses are likely to have been of larger trees with consequent higher probabilities of containing suitable nesting hollows for the RtBC. However, losses of food resources over the same period appear to have been greater than losses of live gums. Between 1992/97–2004, the overall annual rate of loss for both public and private land tenures combined was 0.32%, due entirely to the loss of 3% of scattered paddock tree cover. Tree cover loss from grazing and non-irrigated cropping land was minor, but significant losses from intensive cropping landscapes suggested that clearing to allow agricultural intensification was an important contributing factor.

A series of future scenarios of resource availability were investigated utilising the information collected during this study and existing data. The current climatic envelope of Buloke was modelled, then projected shifts in this envelope under climate change scenarios was explored. The scenarios suggested a general southwards shift of the current climatic envelope of Buloke in the RtBC range. While an increase in temperature alone would not have a negative effect on Buloke distribution within the RtBC range, a 20% decrease in rainfall would substantially reduce the climatic suitability of almost the entire area in which Buloke currently occurs within the RtBC range.

Secondly, information gathered about the current state of feeding and nesting resources, recent rates of change in availability of these resources and future management options was compiled to demonstrate a suite of alternative future scenarios of resource availability. Projections of Buloke availability were characterised by a decline for 100 years followed by some recovery. The status quo scenario, which projected current rates of change into the future, was the worst for Buloke resource availability, with resource availability at only 65% of current levels at 100 years and subsequent increases from maturing revegetation being moderate. The same decline in resources was also expected for a scenario of continued clearing of paddock trees with offsets involving protection of Buloke trees in patches. Protection of other scattered paddock trees as offsets for clearing fared better, but was contingent on the magnitude of the reductions in death rates achieved for ‘protected’ trees. A general reduction in paddock tree death rates achieved the best outcome for resource availability. Revegetation was critical for recovery in resource levels after 100 years. Revegetation effort of 10,000–18,000 Buloke trees per year for 20 years needed to be included in the scenarios to restore Buloke availability to 2004 levels in the long term.

For stringybarks, scenarios with moderately increased wildfire extents reduced stringybark resource availability by up to 5%, while reducing canopy scorch in prescribed burns led to a projected 3% increase. A severe negative effect was caused by catastrophic wildfire where 85% of woodland blocks were burnt, which produced an overall reduction in stringybark seed resources of 49%. Simulating the loss of all paddock trees also produced a severe reduction in stringybark food availability of 11%. The most effective management action in terms of increasing overall stringybark seed availability was found to be the replanting of low density ‘paddock’ trees.

The scenarios of gum availability showed less substantial declines over time, as in most scenarios tree losses were partially offset by maturation of roadside trees. However, the rates of paddock tree loss may accelerate in the future, as trees age and land use intensifies. The scenarios examined varied little, and suggested that rates of loss of large old gums over time are not expected to be as great as those of Buloke, or the potential losses of stringybark food availability under several plausible scenarios. Therefore, it seems likely that food availability will remain the limiting factor for the RtBC population for the foreseeable future.

The findings of this report suggest several general recommendations:

1. Plant and protect scarce resources to maximise complementarity, specifically Buloke and *E. arenacea*;
2. Improve protection of paddock trees, which are the most valuable occurrences for all resource types;
3. Where clearing is unavoidable, ensure offsets for clearing paddock trees protect several other at-risk paddock trees for each tree cleared;
4. Maintain appropriate stem density when planting and manage woodlands to reduce competition.

More particularly, Buloke revegetation effort and survival rates should be substantially increased, and replanting should be focused in higher rainfall areas and on shallow A-horizons. Stringybark revegetation should focus on replanting *E. arenacea* ‘paddock’ (or low density) trees in appropriate areas (ideally, within 1–5 km of existing stringybark remnants). Continued emphasis on prescribed burns that minimise canopy scorch is also important for RtBC resource availability.

# 1. INTRODUCTION

## 1.1 BACKGROUND

The south-eastern Red-tailed Black-Cockatoo, *Calyptorhynchus banksii graptogyne*, is a nationally endangered taxon, restricted in occurrence to the far south-east of South Australia and south-western Victoria (Commonwealth of Australia 2007). Unlike other subspecies of *C. banksii*, the south-eastern Red-tailed Black-Cockatoo (hereafter RtBC) is an ecological specialist, feeding exclusively on the seeds of Brown Stringybark *Eucalyptus baxteri*, Desert Stringybark *E. arenacea* and Buloke *Allocasuarina luehmannii*. Through the work of the Red-tailed Black-Cockatoo Recovery Team and associated researchers the ongoing decline in quality and quantity of food resources has been identified as the major threat to the cockatoo (Garnett and Crowley, 2000; Hill, in prep; Burnard and Hill, 2002; Koch, 2003; Maron, 2005; Maron and Fitzsimons 2007). The availability of nesting hollows, although not currently a limiting factor, may potentially become important in the future as large old hollow-bearing trees are lost.

**Threats to key resources**

Buloke woodland has been heavily cleared for agriculture, and is now a nationally threatened plant community (EPBC 1999). Much of the Buloke that remains in the RtBC range occurs as scattered paddock trees. Recent research has documented one of the highest rates of loss of scattered paddock trees in Australia within the RtBC's range (Maron, 2005; Maron & Fitzsimons, 2007), linked in part to a high rate of agricultural intensification occurring in the north of the taxon's range (Maron & Fitzsimons, 2007). This is complicated by slow maturation of trees; replanted Buloke seedlings may take 100 years to reach a size at which they are utilized by the cockatoo, and considerably longer before they become a preferred resource (Maron, 2000; Maron & Lill, 2004). Threats to the stringybark resource include tree damage due to grazing (Hill, unpublished data) and canopy scorch from prescribed burns and wildfires which substantially reduce the amount of seed available for up to 10 years (Koch, 2003). The extent to which potentially hollow-bearing gum eucalypts are lost on private land has not been estimated.

**Investment in habitat protection and enhancement**

The protection and enhancement of RtBC habitat has attracted substantial investment throughout the cockatoo's range. In recent years, there have been considerable efforts by individual landholders to protect and enhance habitat within the cockatoo's range. These efforts have been supported by non-government organisations such as Greening Australia and Trust for Nature, as well as through the initiatives of the regional natural resource management organisations. In particular, through the Wimmera Catchment Management Authority, a market-based program disbursing over $700K to private landholders in exchange for habitat protection and enhancement is being implemented. Changes to policies surrounding applications to clear native vegetation have also taken into account the losses of RtBC habitat. The West Wimmera Shire has adopted an Environmental Significance Overlay to protect potential cockatoo nest trees, and the Victorian Department of Sustainability and Environment are developing interim operational guidelines in order to assist interpretation of the Native Vegetation Framework in the case of applications to remove RtBC habitat.

Given the significant investment in the protection of the RtBC's key resources, it is of critical importance that clear guidelines for the most efficient and effective targeting of investment are available. Both legislative and policy frameworks and landholder incentive schemes must be informed by the best quality science in order to ensure that such actions work to maintain and enhance the feeding and nesting habitat of the cockatoo, and the multitude of species which also rely on its habitats. This study aimed to bring together existing research and to fill important knowledge gaps to develop a series of future scenarios for RtBC resource availability under different combinations of permitted clearing, burning, protection and enhancement of habitat. It represents a tool to assist land managers and policy makers in determining likely large-scale and long-term effects on the RtBC of different strategies for native vegetation management in the RtBC region.

## *1.2 STUDY OBJECTIVES*

The research described herein had three main objectives:

**1) Identify the extent and condition of habitat currently available to south-eastern Red-tailed Black-Cockatoos**.

The current availability of all species of feeding and nesting resources was detailed, building on existing datasets by adding detail on condition from the perspective of the RtBC, such as tree age structure (as well as sex ratio for Bulokes) and tree health, as well as fire history data. This involved documenting:

- the current extent of patches of potential feeding and nesting habitat including roadside vegetation corridors;
- the current density of paddock trees relative to trees in patches (Buloke, Brown Stringybark, Desert Stringybark, Yellow Gum *E. leucoxylon* and Red Gum *E. camaldulensis*). Existing data for Bulokes were extended across the RtBC range using aerial photography analysis. For stringybarks, the relative food value of trees in paddocks was determined;
- the current age-class distribution and health status for Bulokes and gum eucalypts (the two resource types which must be of substantial age before they provide a useful resource for the RtBC). For Bulokes, sex ratio and degree of mistletoe parasitism was also determined;
- the impacts of grazing on the health and quality of stringybark woodlands and food value of stringybark trees (measurements of percentage dieback, recruitment and density of capsules).

**2. Identify how food availability has changed historically and more recently under various land management regimes**

This required building on existing research for Bulokes in Victoria and eucalypts in South Australia by documenting:

- losses of Buloke, stringybark and gum woodlands over the past 40–60 years (using existing spatial data and recent and historical aerial photography). This provided an historical context for interpreting the effect on the RtBC population of current and future resource availability;
- recent losses of Buloke, stringybark and gum eucalypt trees from landscapes dominated by different land uses over a period of 7–12 years to 2004 (using analysis of aerial photographs and some ground-truthing). This provided annual rates of change calculated over a recent time period to be used as estimates of current rates of change.

**3. Identify how habitat extent and condition would change in the future under different scenarios of protection, enhancement and restoration.**

This part of the project combined all available information with expert knowledge to develop a series of future scenarios of resource availability for the RtBC. Firstly, we used bioclimatic modelling to investigate the current climate envelope inhabited by Buloke trees and examine the potential for climate change to result in changes in Buloke distribution within the range of the RtBC. Secondly, we examined projections of resource availability over 250 years (as this is the minimum timeframe which allows incorporation of newly planted eucalypt trees becoming potential nesting resources). Availability of each resource type will change temporally in response to factors such as tree recruitment, maturation rates and death rates, as well as fire frequency, extent and intensity (important determinants of condition for stringybarks). These factors all vary according to changes in land use, climate change, policy and legislation, and investment in incentives. For this modelling, existing data on rates of change for each resource were used, including:

- estimated growth rates of Buloke and gums
- estimated age of gum eucalypt trees which form large hollows
- current fire regime
- impacts of fire and grazing on stringybark trees and food value
- likely future changes to wildfire frequency due to climate change
- annual rates of loss of Buloke, stringybark and gum eucalypt trees and woodlands in different land use types.

We developed a dynamic mathematical model for each resource (mature Buloke, stringybark, mature gum eucalypts) to simulate availability under different scenarios of land management and planning. Numerous scenarios are explored in this report but the model has the capacity for updating of parameter values to reflect changing knowledge and to investigate the potential impacts of alternative policy, planning and management options. A set of scenarios are presented as Bayesian Belief Network models in Netica v.3.24, which has a user-friendly interface that can be used to quickly identify outcomes of different combinations of planning and management options. These models also contain a ‘goal seek’ function, whereby a desired outcome can be specified and required changes to management and policy identified. The models are held by the Wimmera Catchment Management Authority.

# 2. GENERAL METHODS – RATES OF LOSS DATA

## 2.1 Assessing tree cover for stringybark and gums

Rates of loss for all habitat types (Buloke, stringybarks, gum eucalypts) were determined through analysis of aerial photographs. Aerial photography is the only means to obtain accurate historical tree cover records, particularly when tree cover resulting from individual paddock trees must be measured. Satellite imagery is generally too coarse in its resolution, particularly when historical imagery is involved. The distribution of the Buloke habitat type is restricted to the northernmost part of the RtBC range and thus it was possible to determine rates of loss across its whole extent. However, as the study area covers approximately 18,000 $km^2$ (Figure 2.1), it was not practical to measure directly rates of loss for stringybark and gum woodlands across the whole RtBC range. Therefore, rates of loss for a sample of the RtBC range were determined using the following sampling method.

**Sampling Method**

A 5 km buffer of current feeding habitat was created around the current extent of stringybark woodlands. This area covered 85% of the current known RtBC range, and maps the extent of the critical habitat of the taxon (with the exception of Buloke, important stands of which occur distant from stringybark woodlands). Almost all recorded nests are within 5 km of stringybark blocks. Sixteen 30 km long x 1 km wide transects were located within this buffer across the RtBC range, with 5 transects located in each NRM region (South-east NRM Board (SENRM), Glenelg-Hopkins CMA (GHCMA) and Wimmera CMA (WCMA) regions) and one additional strip to compensate for lost coverage due to aerial photo “blackspots”. Transects were haphazardly located within NRM regions, distributed along a north-south gradient in each, and all ran on an east-west axis (Figure 2.1). Transects were further divided into 1 x 1 km sample plots. In locations where the aerial photo data were unable to provide a 30 km transect, several 1 x 1 km plots were located randomly above and below the main transect. Each transect and sample plot was filled by a graphic fishnet of 100 x 1 ha squares to aid characterisation of features (see Figure 2.2).

Within the transects, tree cover was sampled from aerial photography from 2004, the 1990s (1997 for WCMA and SENRM and 1992 for GHCMA) and 1947 (for example sample areas, see Appendices 1–3). The total sample area was 47,400 ha for the 2004 and 1990s photos and 42,700 ha for the 1940s photos (some aerial photos were missing). This accounts for around 2% of the total mapped critical habitat area of the RtBC. The 2004 imagery was supplied by the South Australian Department of the Environment and Heritage and the Catchment Management Authorities. Other aerial photos were scanned and rectified so as to have a 2 m on-ground spatial resolution (which was adequate to identify individual paddock trees).

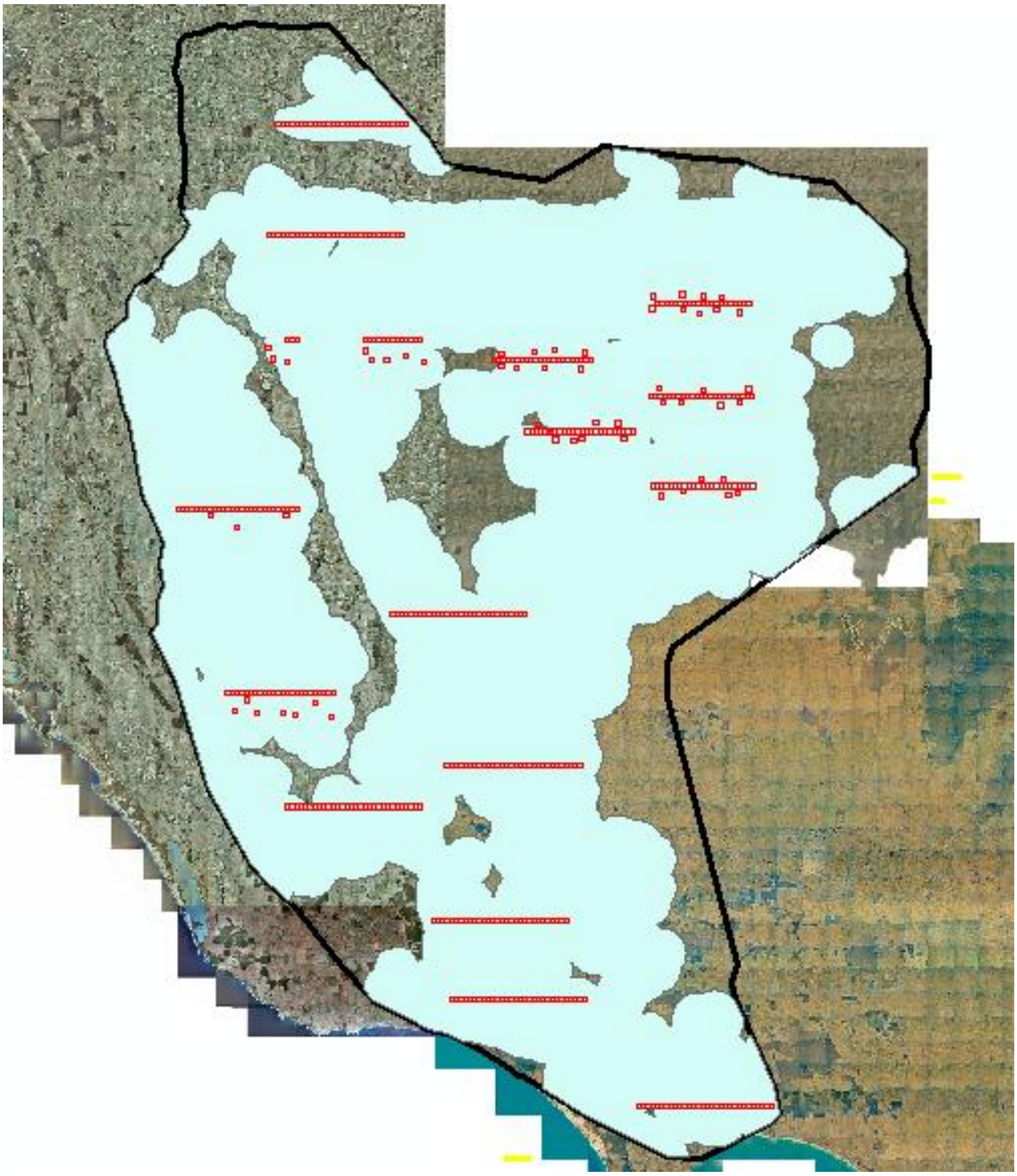


**Figure 2.1.** The location of the 30 km transects used for determination of change in tree cover of stringybarks and gums. Study area boundary is marked in black, 5 km buffer from stringybark blocks shown in pale blue.

Each 1 x 1 km sample plot was visually viewed at a consistent 1:12,000 scale to measure the area in hectares of native tree cover. At this scale it was possible to distinguish remnant native vegetation and paddock trees from plantations or exotic species used in windbreaks or homestead yards, which were not counted as tree cover. A hectare of tree cover was defined as over 75% canopy cover of the hectare plot. Half and quarter tree cover hectares were also counted to contribute to the total tree cover (see Figure 2.3).

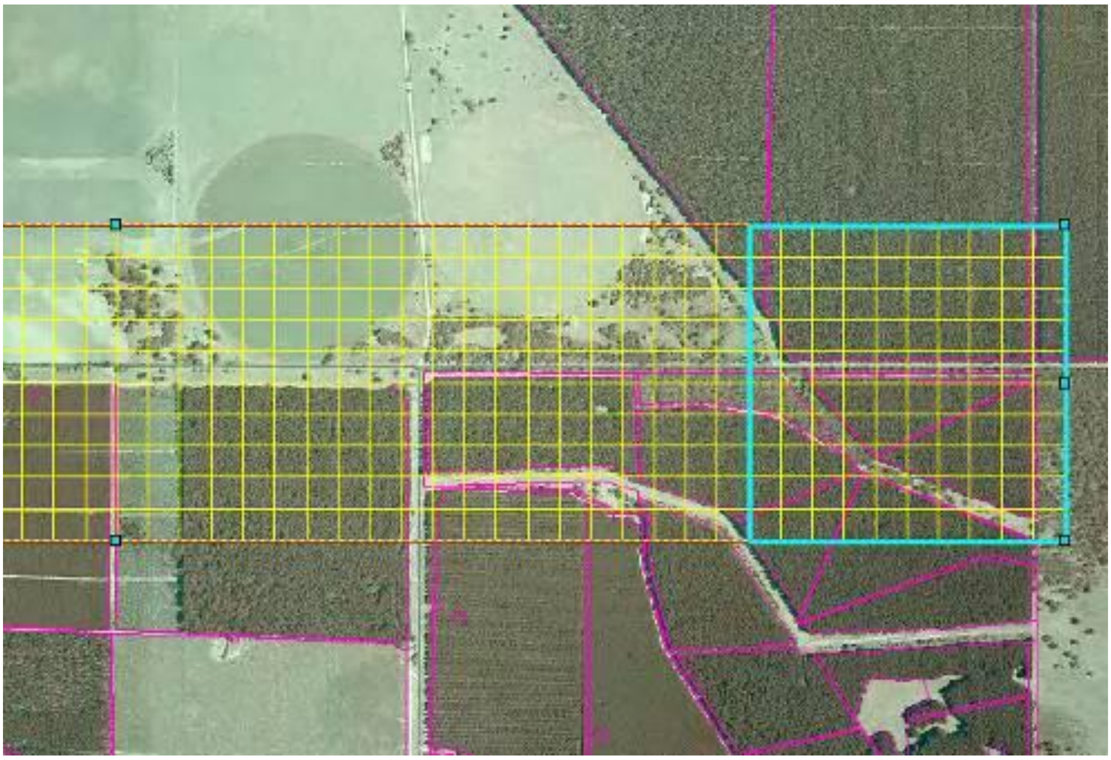


**Figure 2.2.** An example of a 1 x 1 km sample plot (blue polygon) with a 100 x 100 m fishnet grid used for recording tree cover.

Using existing mapping of the distribution of vegetation types dominated by each of the two stringybark species and gum eucalypts, the tree cover within each sample plot was attributed to gum eucalypts, *E. arenacea*, or *E. baxteri*. Mapping of pre-settlement vegetation was used so that paddock trees and other vegetation not covered by current extent mapping could be attributed to a habitat type. For each time point considered, the dominant (most extensive) land tenure and land use in each 1 ha square within the 1 x 1 km sample plots was identified using a combination of land use mapping (see Section 2.2) and visual assessment. Visual assessment was used for identifying areas of intensive agriculture such as centre pivots in areas where this was not mapped. Land use at a paddock scale can vary year by year, and so although this approach to mapping land use was fairly coarse, it nonetheless gave a useful snapshot in order for changes in the extent of vegetation over time to be compared among particular land tenures and land uses. As amounts of tree cover change were not available over the same period for the different regions when calculating the recent rates of change (1992–2004 for GHCMA; 1997–2004 for WCMA and SENRM) we calculated an average annual rate of loss weighted by the number of 1 x 1 km sample plots in each region (annual rates of loss per region were also reported).

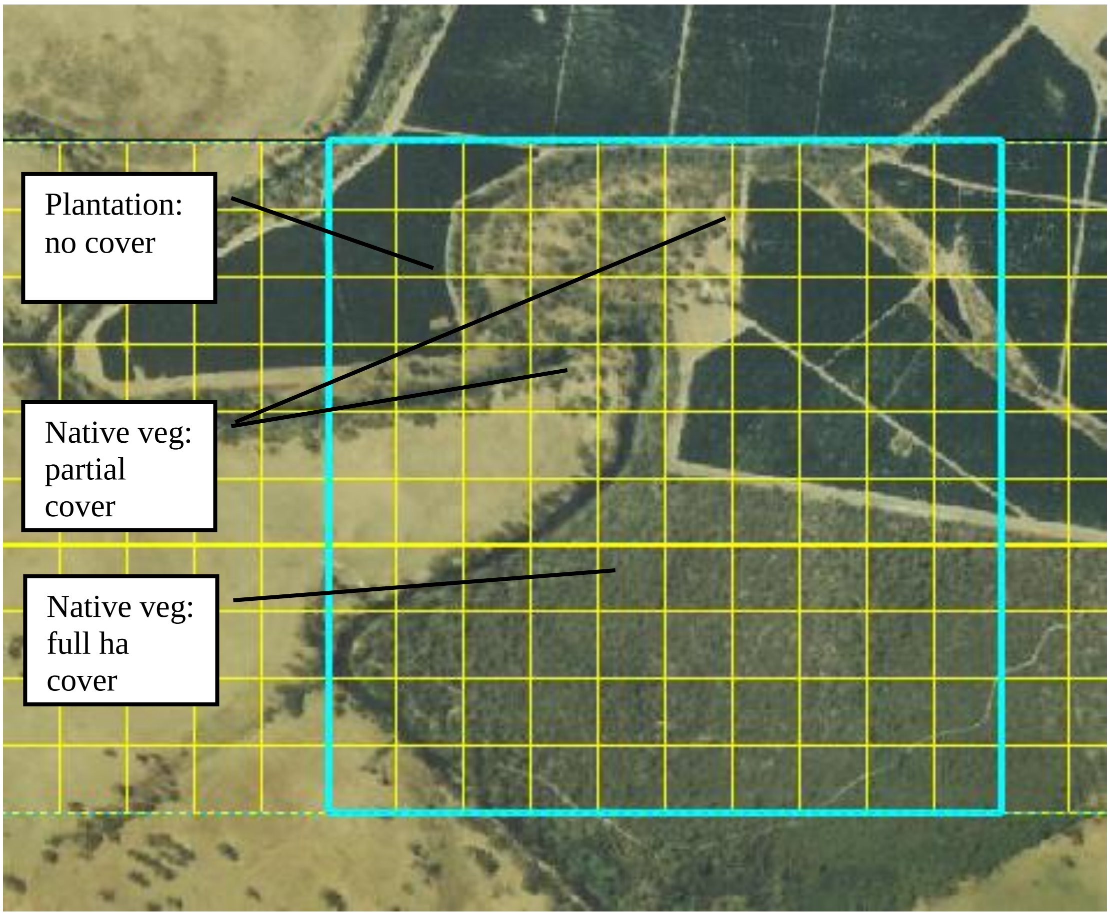



**Figure 2.3.** Example assessments of tree cover for stringybarks and gums.

## 2.2 Land use and land tenure mapping

An important consideration for the study was the impact of different agricultural practices on rates of loss for the various RtBC habitat types. Initially, a new land use layer was created from South Australian and Victorian land use data by amalgamating the data to the most simple common fields. Additional data on plantation locations and private conservation areas were used to enhance the dataset. Land use classifications were verified at each sample location where possible through interpretation of the aerial photography.

# 3. FACTORS INFLUENCING BULOKE AVAILABILITY

## *3.1 CURRENT STATE OF BULOKE RESOURCES*

### 3.1.1 Introduction

Buloke is a critically important food resource for the RtBC, providing high-quality seed in most years during summer and autumn (Maron and Lill 2004). Although widely distributed on a variety of soils types throughout a large area of eastern Australia, Buloke trees in the RtBC range historically occurred in the north, on the more fertile soils of the plains lying between the sandy dunes. Today, agriculture dominates these fertile soils, and as such, the distribution of Buloke woodland is reduced to only a few percent of its original extent in the Wimmera region. Buloke now occurs as scattered paddock trees and regeneration on roadsides, and in a few small patches of woodland. In the south-west Wimmera region, there have been substantial changes to the distribution and availability of Buloke over the past 50 years, with paddock trees declining and dense regeneration on roadsides increasing (Maron 2005; Maron and Fitzsimons 2007). Despite the increase in roadside Buloke vegetation, not all trees in these areas can be considered suitable feeding habitat, as RtBCs preferentially feed in trees larger than 30–40 cm diameter at breast height (DBH) and rarely feed in trees smaller than 19 cm DBH (Maron and Lill 2004). Recent work has suggested that Buloke growth rates are very slow, with trees of 19 cm DBH estimated at about 100 years of age (Macaulay, unpublished data). Therefore, the current availability of Buloke food resources for the RtBC is not the same as the number of live Buloke trees. An understanding of the current availability of mature Buloke trees is required in order to set initial conditions for any modelling of future scenarios.

To identify the amount of Buloke food resource currently available to the RtBC, we needed to a) map the distribution of remaining Buloke trees within their range, and b) identify what proportion of these trees is mature and thus producing a useful food source. Neither of these factors were known, and therefore this part of the project aimed to fill these knowledge gaps in order to determine current Buloke food availability across the RtBC range.

### 3.1.2 Methods

**Spatial distribution**

Aerial photography for the Wimmera and South East NRM regions captured in 2004 was used to map the current distribution of Buloke trees and Buloke-dominated woodlands within the Buloke foraging range of the RtBC (few trees are located in the Glenelg-Hopkins NRM region). This range was considered to be all Buloke south of the Little Desert National Park and west of Wail, within a total area of approximately 5,000 $km^2$. Although the distribution of Buloke continues north and east in Victoria, RtBCs are not known to feed in these trees and the vast majority of sightings of RtBCs foraging in Buloke are in the south-west of the area. All paddock Buloke trees were digitised in ArcMap v. 9.2. This digitisation was done manually as available automated procedures were unsuitable for identifying trees of different species, particularly when the imagery depicts trees against backgrounds which vary in color and contrast. An experienced observer conducted all species identification and digitisation by applying a 1 x 1 km fishnet across the study area and systematically searching each grid square. Species assignment was on the basis of canopy size, canopy color and proximity to other stands of Buloke. Individual paddock trees were identified as Bulokes through a combination of known distribution, greyish (as opposed to green) canopy color and small canopy size compared with other paddock trees such as eucalypts. In some cases where tree identity within an area was uncertain, ground-truthing was undertaken. Reference to other available aerial photography and, in the case of South Australia, mapping of the distribution of the Buloke woodland vegetation community, also assisted in species identification.

The distribution of Buloke in paddocks was used as a guide to the location of remnant patches of Buloke and Buloke-dominated roadsides, as well as Buloke-dominated regions within larger forests and reserves. Patches included areas of paddock trees where tree canopy was continuous or near-continuous, thus precluding mapping of individual trees. Roadside areas of Buloke were mapped separately to other patches. Again, all digitisation was done manually as judgements on species were too subtle to allow automation of the process, particularly across images taken at different times of the year.

Mapping of Buloke trees was conducted throughout the area described above, but detailed counts and measurements of trees and Buloke patches were conducted within a subset of this area which was comparable with that for which 1997 aerial photo coverage was available (see Section 3.3 below). This area (hereafter referred to as the 2004 study area) included the vast majority of mapped Bulokes but excluded a few outlying populations.

**Stem density and size class distribution**

Bulokes typically occur in one of three situations: as scattered paddock trees and small clumps in grazed and cropped paddocks; on roadsides, and in patches on public land or protected private land. Each of these occurrences has a different characteristic stem density and size class distribution. Therefore, field sampling to determine stem density and size class distribution was stratified by these categories.

For paddock trees and most small grazed clumps, tree density could most easily be determined using aerial photography, as individual trees could be distinguished. However, the determination of mean stem density for Bulokes occurring as patches on private land, public land or roadsides required field measurements. For these measurements, a series of sites in each category were randomly located throughout the Buloke study area in both South Australia and Victoria (Figure 3.2).

For patches on private and public land, 20 x 20 m quadrats were randomly located within areas of continuous or near-continuous canopy cover (corresponding with areas mapped as patches) and the stem density of Bulokes recorded. In some cases on public land young stems were too dense to permit accurate counting within time constraints in 20 x 20 m quadrats. In these cases, stems were counted in 5 x 5 m quadrats. A total of 11 sample sites were established in Buloke woodland patches.

For roadsides, due to their narrow, linear shape and very high stem density, quadrat dimensions generally had to be modified and were typically 3 x 3 m, and up to five quadrats were located at each of 18 roadside sites(a total of 79 quadrats in total). The mean stem density from all quadrats at a site was considered one replicate (N = 18),

In addition to estimating mean stem density of Bulokes in each category, we also recorded examples of particularly high stem density in cases where the majority of stems exceeded 30 cm DBH. This was because we needed to be able to estimate the maximum stem density likely to be attained by a woodland or stand of mature trees for the purposes of modelling future scenarios. During data collection for the above activities, dense stands of mature Buloke on both public and private land were sought out. Trees within a 50 x 50 m quadrat in the two densest parts of the densest patch of mature (>30 cm DBH) Buloke were counted and measured.

Size class determination for Buloke trees on roadsides and in patches and reserves was done in the same quadrats used for stem density measurements. For paddock trees, an additional

15 sites were located in both cropped and grazed paddocks that were accessible from the road (Figure 3.2). At each paddock tree site, the girths of the 20 trees nearest to a random location along the access road were measured. Other variables recorded for each tree included the number and species of any mistletoes growing on it, a canopy health score measured on a Lickert scale of 1–10 (with 1 representing a tree with very little live canopy and 10 representing a full, healthy canopy; see Fig 1a & b; Maron 2005) and its sex (most females still held cones at this time and males typically had early flowers).

a) 

b) 

**Figure 3.1.** Examples of paddock Buloke trees with canopy health scores of a) 10 and b) 3.

For roadsides and public and private land patches, measurements of girth and mistletoe presence were taken at the same sites that were used for stem density determination. For these trees, canopy health and sex determination was unreliable due to high stem densities resulting in difficulty in distinguishing individual canopies, and the fact that although many young trees showed no sign of fruit, they had probably not yet reached an age where fruiting would occur, and so males and females appeared similar.

In order to create a frequency histogram of Buloke age classes within the study area, the number of Buloke trees on roadsides and in remnant patches were estimated using the total mapped area of each of Buloke-dominated patches of these types and the mean Buloke stem density appropriate to that patch type. The mean size class distributions for each patch type were then applied to the derived estimate of total tree numbers for the study area. The same process was used to determine size-class distributions of the paddock Bulokes, which when combined with the data for the patch types gave an estimate of the total number of Bulokes within each size class across the study area.

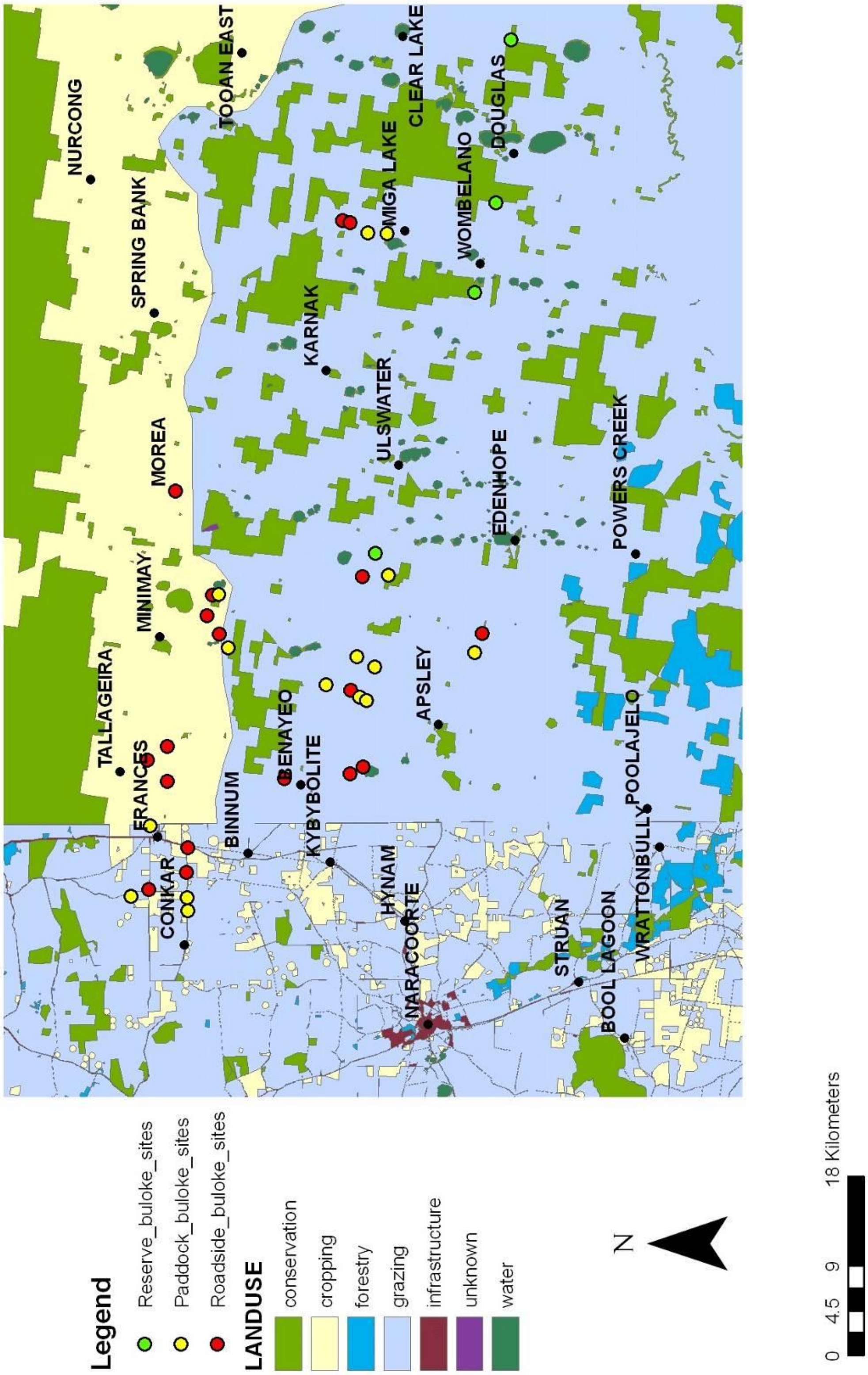



**Figure 3.2.** Location of field sites for Buloke size class and stem density measurements.

### 3.1.3 Results

**Spatial distribution**

The mapped distribution of scattered paddock Buloke trees is presented in Figure 3.3. Bulokes occurred extensively throughout the study area, and a total of 51,477 paddock trees were counted throughout the 2004 study area, with the vast majority (49,463) in Victoria. The distribution of trees was fairly evenly split between cropping and grazing landscapes, with a small number in conservation land (Table 3.1).

**Table 3.1.** Number of paddock Buloke trees in each state and each land use in 2004, within the 2004 study area.

| Land use | Vic | SA | Total |
|---|---|---|---|
| Cropping | 24738 | 384 | 25122 |
| Grazing | 24461 | 1604 | 26065 |
| Conservation | 166 | 0 | 166 |
| Other/unknown | 98 | 26 | 124 |
| **Total paddock trees** | 49463 | 2014 | 51477 |

The area of densely occurring Buloke trees on roadsides and patches was relatively small, with a total of 162 ha of roadsides and 201 ha of other patches within the 2004 Buloke study area (Table 3.2). Extrapolation from results of the stem density and size class distribution study (see below) suggested that approximately 40,000 mature trees occur in areas where tree canopy is continuous in patches and on roadsides.

**Table 3.2.** Area of densely occurring Buloke tree cover and estimated maximum number of mature trees in woodland in each state in 2004, within the 2004 study area.

| | Vic | | SA | | Total | |
|---|---|---|---|---|---|---|
| **Land use** | **Area** | **Est. trees** | **Area** | **Est. trees** | **Area** | **Est. trees** |
| Patches (ha) | 192.8 | 14845 | 8.3 | 639 | 201.1 | 15484 |
| Roadsides (ha) | 143.4 | 21223 | 19.2 | 2842 | 162.6 | 24065 |
| **Total** | **336.2** | **36068** | **27.5** | **3481** | **363.7** | **39549** |

**Stem density**

The stem density values for the quadrats in the highest stem density patch were 28 and 32 per 50 x 50 m quadrat, with mean DBHs of 39.6 and 32.1, respectively. Once trees smaller than 30 cm DBH were excluded, the total number of mature trees per quadrat was 22 and 20. Thus, the figure used for the maximum mature Buloke density was 21 per 2500 m2 or 84 per hectare.

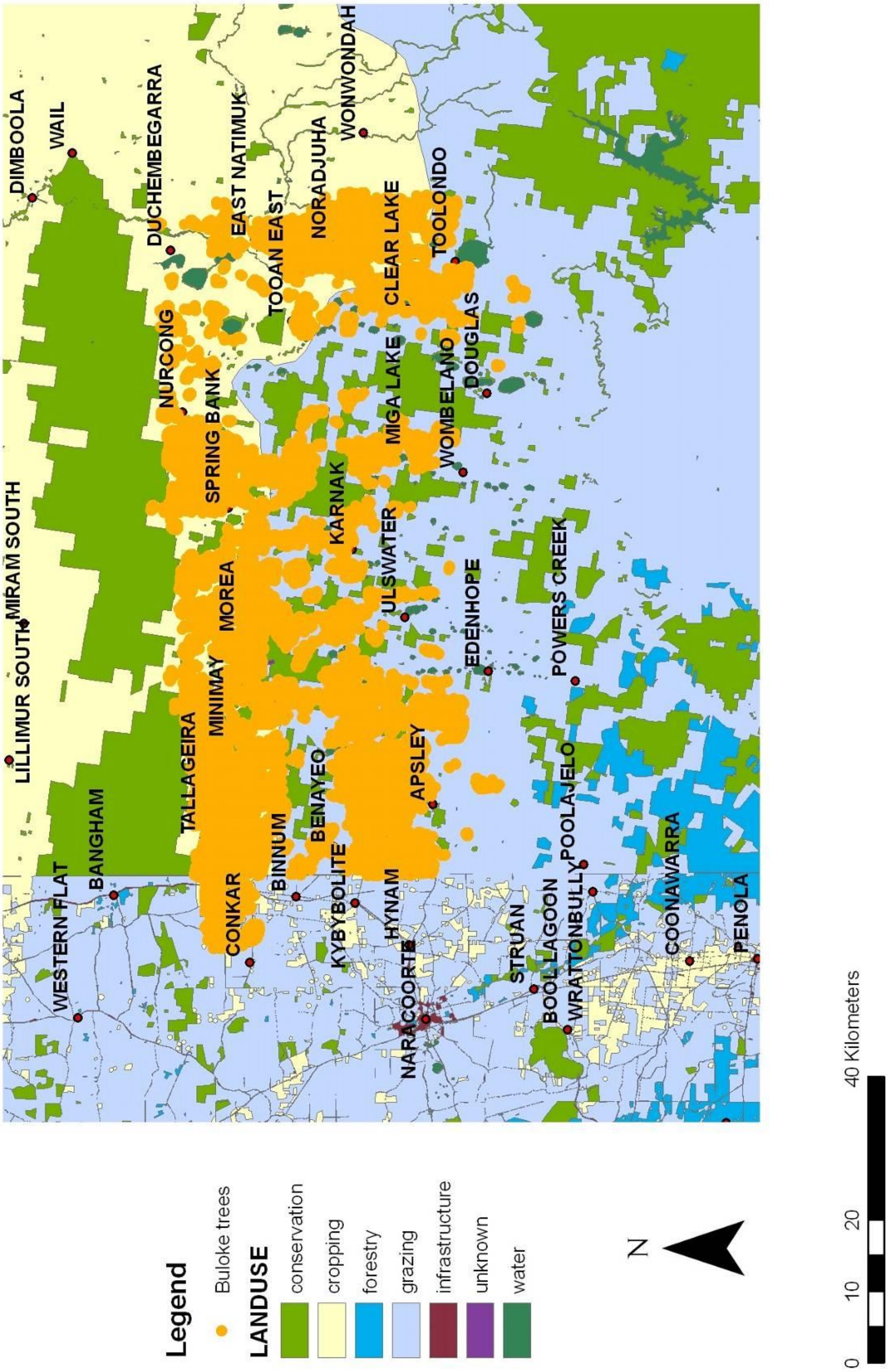



**Figure 3.3.** The distribution of Buloke trees in the study area in 2004.

The grand mean (across 18 sites) of Buloke stem density density on roadsides per 3 x 3 m quadrat was 7 ± 1 (mean ±1 s.e), equating to over 7700 stems per hectare (hereafter all errors reported are 1 s.e.). Density of Buloke trees in reserves or ungrazed patches was estimated from 20 x 20 m quadrats at the 11 sites. The mean was 72 ± 20 stems per quadrat, equating to 1800 stems per hectare.

**Size and age class distribution**

Paddock Buloke trees averaged 54 ± 1 cm DBH, with the majority between 40–70 cm DBH (Figure 3.4). There were very few small Buloke trees in paddocks and no seedlings or suckers <10 cm diameter were recorded. Mean canopy health score was 6.2 ± 0.1, with relatively few trees having a canopy health score of less than 5 (Figure 3.5). There was a weak but statistically significant positive relationship between tree DBH and canopy health scores (Figure 3.6). Two species of mistletoe were recorded parasitising the trees: Buloke mistletoe *Amyema linophylla* and the more abundant harlequin mistletoe *Lysiana exocarpi*. The number of mistletoe plants per tree averaged 12 ± 1. There was no relationship between canopy health score and the number of mistletoe plants on a tree ($R^2$ = 0.006). The ratio of female to male paddock Bulokes was 0.75:1.

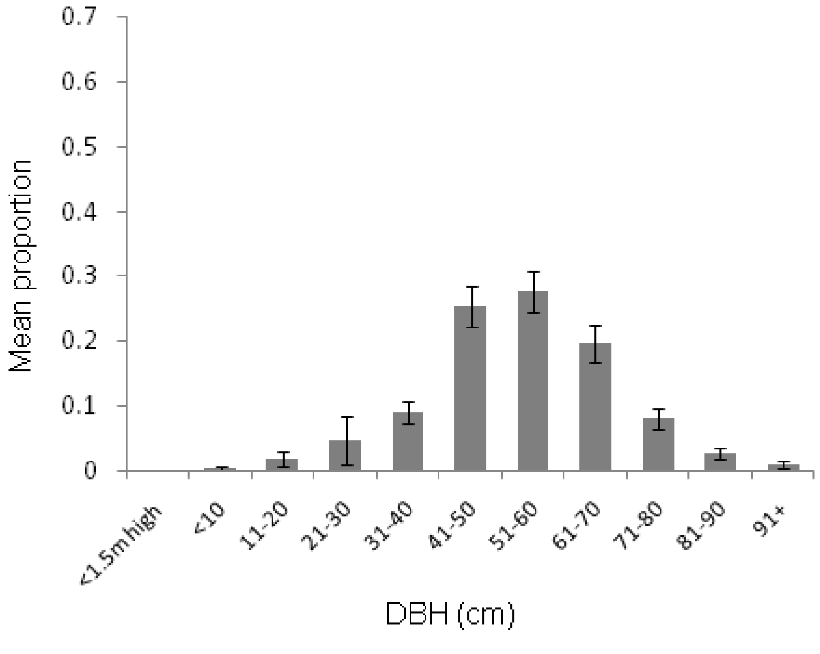



**Figure 3.4.** Mean size class distribution of paddock Buloke trees.

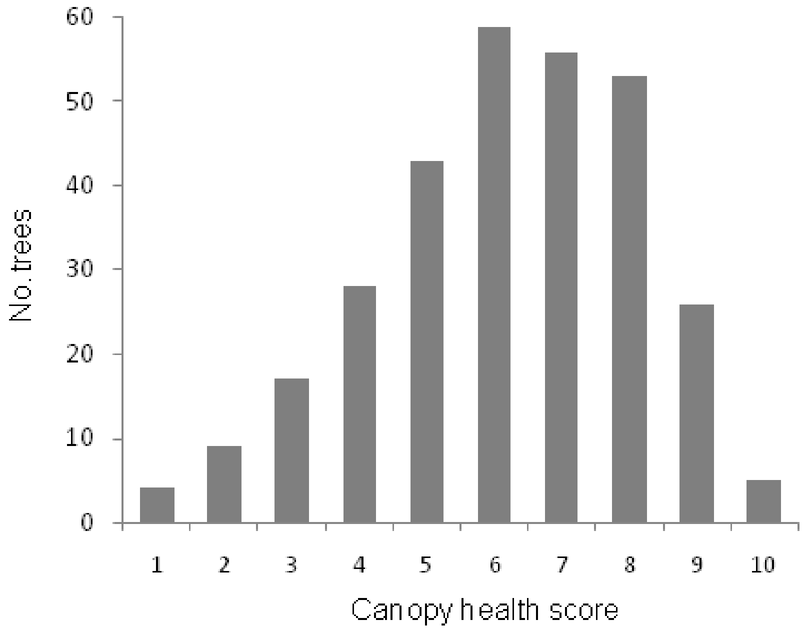



**Figure 3.5.** The frequency distribution of canopy health scores for the 300 paddock Bulokes assessed.

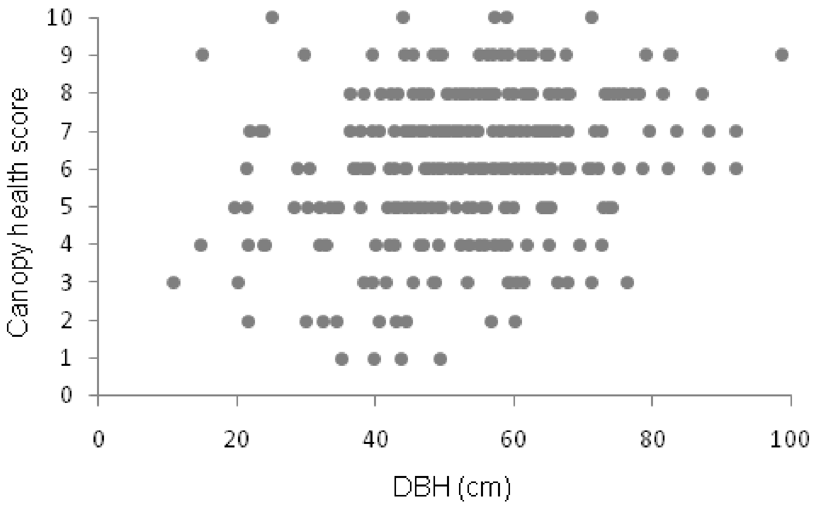



**Figure 3.6.** Relationship between canopy health and diameter at breast height of the 300 paddock trees assessed ($R^2$ = 0.07).

A total of 567 trees were measured in the 79 roadside quadrats. The size class distribution of Buloke trees on roadsides differed substantially from that of paddock trees, with roadside trees averaging 5.6 ± 0.3 cm DBH (Figure 3.7). Most trees fell within the 1–10 cm DBH size class with a substantial number of trees being smaller than 150 cm in height (and therefore with no DBH recorded). Mistletoes were far less likely to occur on Buloke trees on roadsides, with only 18 of the 567 trees being parasitised. The size class distribution of Buloke trees in ungrazed reserves and patches was very similar to that recorded for roadsides, although the stem density was lower (Figure 3.8).

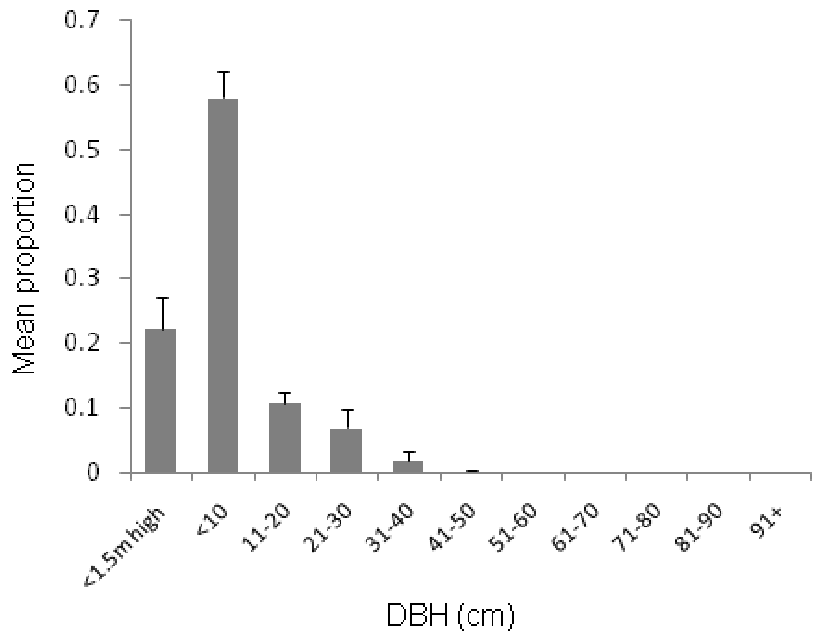



**Figure 3.7.** Mean (± 1 s.e.) size class distribution of Buloke trees on roadsides.

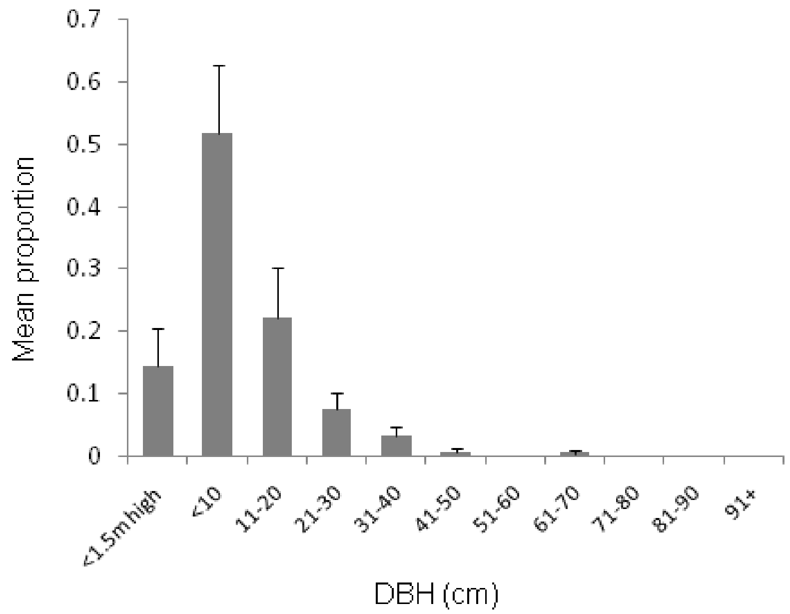



**Figure 3.8**. Mean (± 1 s.e.) size class distribution of Bulokes in reserves and patches.

By combining the mean stem density and age class information with the mapped area of Buloke patches and roadsides and number of paddock trees, a frequency histogram of the estimated current number of Bulokes in each size class in the study area was created (Fig 3.9). By far the majority of trees were smaller than 10 cm DBH.

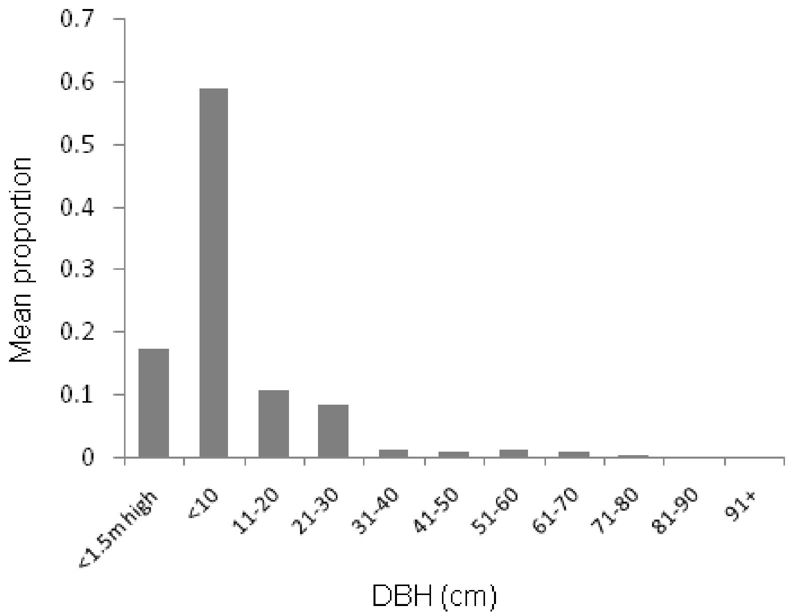



**Figure 3.9.** Estimated size class distribution of Buloke trees in the study area in 2004.

### 3.1.4 Discussion

Although trees were mapped as far east as East Natimuk, most sightings of RtBCs are made in the far west of the region. It is not known whether this is because the trees in the eastern half of the study area are less suitable for the RtBC, or whether there are fewer observers engaged in reporting sightings in those areas. Tree selection by the RtBC is sensitive to factors related to foraging profitability (Maron and Lill 2004), and it is possible that the heavier soils and higher rainfall typical of the far western part of the region result in more productive Buloke trees. The Buloke trees in this area appear much larger than those located further north in Australia, and produce larger cones with more seeds (M. Maron, personal observation). Even the Buloke trees occurring on the northern edge of the RtBC's range tend to be smaller and less productive.

The majority of the distribution of Buloke in the study area is on private land which is grazed or cropped. Both of these land uses prevent recruitment of Buloke and no young trees were observed in paddocks. Natural regeneration of Buloke is therefore restricted to roadsides, fenced patches and reserves. However, where this regeneration occurs, it is very dense and appears to be of sucker, rather than seedling, origin. The dense young trees on roadsides may have their origin as suckers which emerged after the cessation of droving along roads in the 1960s, or following roadworks around that time. Their parents are likely to be the few large trees which were present on the roadsides. Due to the extremely high density of these young trees, only a very small proportion might ever mature to trees of sufficient size to provide a useful resource for the RtBC. Interestingly, the density of trees >30 cm DBH on roadsides is estimated already to exceed the maximum density recorded for mature trees in non-linear patches. This is likely to be due to the linear nature of the roadside strips, which results in reduced competition due to the absence of other perennial vegetation either side of the strip.

The strongly skewed size class distribution of Buloke trees in the study area confirmed the strong preference of RtBCs for large Bulokes. Although approximately 95% of available Buloke trees are < 30 cm DBH and more than 85% are < 20 cm DBH, the majority of trees in which RtBCs have been observed feeding are > 30 cm DBH and almost all are > 20 cm DBH (Maron and Lill 2004). Through considering all Buloke trees available in the study area, this study has provided a more accurate picture of the strength of RtBC feeding preferences.

There was considerable difficulty in mapping Buloke-dominated areas within large blocks of other vegetation types, such as state forests. This may be due in part to problems with distinguishing the canopies of buloke trees from other vegetation types in the mosaic-like patches. However, it was clear that in the majority of cases land dominated by Buloke trees had been cleared up to the ecotone with another vegetation type. Furthermore, during extensive field work in the interior of large blocks of publicly-owned native vegetation (both as part of the Buloke field surveys and those for gum eucalypts), few areas of vegetation dominated by Buloke were located, with most having an emergent canopy of eucalypts, preventing their identification from aerial photography. Buloke trees in these areas also tended to be smaller and therefore less suitable for RtBC foraging.

The ratio of female to male paddock Buloke trees was fairly low, at 0.75:1, and no large-scale patchiness in the sex ratio was evident during surveying. Although RtBCs feed on the seeds produced by the female trees, both sexes are required for seed set, and so preservation of both males and females is important to resource availability for RtBCs. However, although the sex ratio departed from 50:50, the departure did not appear to be substantial. Without knowledge of the historical sex ratio of Buloke trees, it is difficult to conclude whether the current sex ratio is a cause for concern, although at a landscape scale, it suggests that males are not a factor limiting seed production.

Interestingly, mistletoe infestation appeared to have no effect on canopy health. Mistletoe is an important resource for many species of birds, mammals and invertebrates and despite its reputation as a tree killer, it rarely poses a lethal threat to trees (Watson 2001). Trees in paddocks varied in their canopy health but most appeared reasonably healthy and a few were exceptional. This work was carried out during a long drought and almost all trees in the study area appeared somewhat stressed. However, Bulokes are particularly resilient and are likely to recover rapidly from drought, although other pressures, such as ringbarking by stock, would continue to contribute to tree stress.

## *3.2 HISTORICAL CHANGE IN BULOKE AVAILABILITY (1963-2004)*

### 3.2.1 Introduction

In order to provide context for the current availability of Bulokes, it is important to quantify past availability. Information on historical availability of Bulokes can also provide insight into which parts of the landscape have undergone the most change and the land uses in which the greatest change occurred. Past work on changes in Buloke availability considered changes between 1981/82–1997 (Maron 2005) and 1997–2004 (Maron and Fitzsimons 2007). However, by the early 1980s, substantial landscape transformation had already occurred throughout the Wimmera and so although these studies provided insight into recent rates of change, assessment of resource availability in the context of longer-term landscape changes was not possible. This part of the project aimed to establish the amount of change in resource availability over a longer time period. The resources made available for this comparison were 1963 hard-copy aerial photo mosaics of the Wimmera CMA region.

### 3.2.2 Methods

Two main datasets were accessed for this component of the study: hard copy aerial photo mosaics (black and white) covering the Victorian part of the study area in 1963 and an existing georectified color photo mosaic captured in 2004.

The study area was as defined in Section 3.1.2. This area encapsulated the vast majority of sightings of RtBCs feeding in Buloke. Within this region, individual paddock Buloke trees, patches of Buloke, and Buloke-dominated roadsides present in 1963 were digitised using the methodology described above in Section 3.1.2, but with frequent reference to the 2004 color photo mosaic to assist in species identification. A sampling approach was taken to digitisation of Buloke for this comparison, as all digitisation was by hand and the number of individual trees present in 1963 was very large (> 100,000). A 1 x 1 km fishnet was placed over the photo mosaic in ArcMap, and mapping was undertaken in every fourth north-south 1 km-wide sample strip. This yielded a total sampled area of 946 $km^2$. The distribution of Buloke trees, patches and roadsides within this area was compared between 1963 and 2004.

### 3.2.3 Results

During the 41 year period from 1963 to 2004, there was a 48.3% loss of Buloke trees in paddocks, which equates to 1.2% loss per year and a total loss of 12,203 trees in the sample strips, which sampled approximately one-quarter of the Buloke study area (Table 3.3). The total area of Buloke woodland patches on private land decreased by 37.8%, but the area of roadside Buloke woodland more than doubled (Table 3.3).

**Table 3.3.** Changes in the number of Buloke paddock trees and area of Buloke woodland in sample strips within the Buloke study area between 1963 and 2004.

| Category | 1963 | 2004 | Change (ha) | % change |
|---|---|---|---|---|
| Patches (ha) | 53.7 | 33.4 | -20.3 | -37.8 |
| Roadside (ha) | 33.7 | 69.6 | 35.9 | 106.5 |
| **Total area** | 87.4 | 103.0 | 15.6 | 17.8 |
| **Trees:** | | | | |
| Cropping | 13909 | 5375 | -8534 | -61.4 |
| Grazing | 11325 | 7661 | -3664 | -32.4 |
| Conservation | 36 | 31 | -5 | -13.9 |
| Other/unknown | 0 | 0 | 0 | |
| **Total trees** | **25270** | **13067** | **-12203** | **-48.3** |

The majority of change occurred on land which is now mapped as cropping. Although the majority of Buloke trees mapped in 1963 occurred in these areas, they declined by 61.4% over the 41 years and by 2004 there were fewer Bulokes in cropping landscapes than in grazing landscapes. Trees in grazing landscapes declined by a substantially lower 32.4% over the same period.

### 3.2.4 Discussion

Since 1964, there have been substantial changes in the distribution and abundance of Buloke trees in the south west Wimmera. Bulokes which in 1963 occurred as scattered trees suffered the most substantial losses, although those occurring in patches also were lost due to clearing. These changes changed the landscape substantially from one which might be classed as variegated, with different parts of the landscape being modified to various degrees, to a fragmented landscape, consisting chiefly of a highly modified agricultural matrix surrounding patches of remnant vegetation. In particular, the increase in contrast between the agricultural areas and remnant vegetation is notable, and is likely to have had negative consequences for biodiversity. A matrix densely scattered with trees such as that evident in the 1963 imagery is likely to constitute suitable habitat for a larger number of native species than one with fewer, sparser trees and dominated by intensive land uses such as the 2004 landscape. Furthermore, movements through the matrix by species normally restricted to denser woodland vegetation are likely to become more difficult.

For the RtBC, although it is a highly mobile species, these changes in the distribution of resources in the landscape have several consequences. Firstly, the loss of resources has been substantial, with increases in the area of dense regeneration on roadsides not compensating for the substantial loss of scattered paddock Bulokes and Buloke woodland patches. The loss of the trees found in paddocks is of particular concern, as these trees are very large and therefore constitute a higher-quality resource for the RtBC. Secondly, a decrease in average density of Buloke trees in the landscape potentially has consequences for RtBC foraging efficiency. The more that birds need to move among areas of Bulokes to find suitable fruiting trees, the more energy they expend and the more profitable the individual trees must be to offset the search costs. Finally, the amount of Buloke food resources in close proximity to potential nest trees is likely to have reduced substantially since 1963. While Bulokes occur on the better soils adjacent to the sandier stringybark-dominated forest areas, River Red Gums *Eucalyptus camaldulensis* tend to occur on the lowest-lying soils of the region near drainage lines. Many nest trees are in hollows in live or dead Red Gums, and the reduction in availability of surrounding Buloke resources conceivably could affect nest success.

## *3.3 RECENT CHANGES IN BULOKE DISTRIBUTION*

### 3.3.1 Introduction

Before likely future trends can be explored, recent data on the rates of change of Buloke trees, patches and roadsides are required. In recent years, substantial changes in land use have occurred, particularly in the West Wimmera. The most notable changes have been increases in cropping on land which formerly was used predominantly for grazing, and conversion of extensively cropped land to irrigated cropping with the use of centre pivots. The drying climate has in part contributed to this, with areas which had in the past been too seasonally wet for cropping now becoming available. These changes in land use have driven substantial changes in the availability of mature Buloke trees in paddocks (Maron and Fitzsimons 2007). This part of the study expands previous work to assess recent rates of change in Buloke availability across the majority of the area of overlap between the RtBC and Buloke, and includes an assessment of rates of change in Buloke-dominated roadsides and remnants.

### 3.3.2 Methods

In order to prepare a ‘status quo’ scenario representing the likely trajectory of Buloke availability under recent rates of change, information was extracted from the digital maps of Buloke distribution in 1997 and 2004. As no extensive digitised aerial photography for 1997 was available, a new study area (the ‘recent comparison study area’) was identified covering the vast majority of the 2004 mapped Buloke (but excluding a few outlier populations). This area covered 151,800 ha. A total of 93 colour aerial photographs providing coverage of this new study area and taken in 1997 were purchased from Quasco Vic Image, scanned and rectified so as to have a 3 m resolution, adequate for identification of mature paddock trees. Rates of change of paddock trees, roadside Buloke and Buloke patches between 1997 and 2004 in South Australia and Victoria and in each land use category (intensive cropping, extensive cropping, grazing) were determined within the recent comparison study area.

### 3.3.3 Results

Between 1997 and 2004, the number of paddock trees within the recent comparison study area changed from 54,840 to 49,463 in Victoria (Table 3.4) and from 2,169 to 2,014 in South Australia (Table 3.5). These represented rates of loss of 9.8% and 7.1% over the seven years in Victoria and South Australia, respectively (Table 3.4 and 3.5). Death rates were similar between cropping and grazing, but were 100% in areas converted to intensive cropping (Table 3.6). The pattern of tree loss was strongly clumped, with most trees lost in the far north west and the south east of the study area. The clumping was also evident at a finer scale, with particular paddocks losing large numbers of trees while adjacent paddock lost none. In particular, for the western half of the region where most RtBC sightings of birds feeding in Buloke are made, tree loss had accelerated since 1997, with 17.6% of paddock Buloke trees lost in seven years.

**Table 3.4.** Changes in area of Buloke-dominated patches and roadsides and numbers of Buloke paddock trees in each land use type in Victoria in the recent comparison study area. Figures for "Pivots" and "Extensive cropping" sum to "All cropping".

| Category | 1997 | 2004 | change | % change | land use area |
|---|---|---|---|---|---|
| Patches (ha) | 192.9 | 192.8 | -0.1 | -0.1 | |
| Roadside (ha) | 142.8 | 143.4 | 0.6 | 0.4 | |
| **Trees:** | | | | | |
| *Pivots* | *1119* | *0* | *-1119* | *-100.0* | *7085* |
| *Extensive cropping* | *26258* | *24738* | *-1520* | *-5.8* | *68976* |
| All cropping | 27377 | 24738 | -2639 | -9.6 | 76061 |
| Grazing | 27199 | 24461 | -2738 | -10.1 | 73772 |
| Conservation | 166 | 166 | 0 | 0.0 | 1846 |
| Other/unknown | 98 | 98 | 0 | 0.0 | 182 |
| **Total** | **54840** | **49463** | **-5377** | **-9.8** | 151861 |

**Table3.5.** Changes in area of Buloke-dominated patches and roadsides and numbers of Buloke paddock trees in each land use type in South Australia. Figures for "Pivots" and "Extensive cropping" sum to "All cropping".

| Category | 1997 | 2004 | change | % change | land use area |
|---|---|---|---|---|---|
| Patches (ha) | 8.3 | 8.3 | 0 | 0.0 | |
| Roadside (ha) | 19.2 | 19.2 | 0 | 0.0 | |
| **Trees:** | | | | | |
| *Pivots* | *106* | *0* | *-106* | *-100.0* | *521* |
| *Extensive cropping* | *406* | *384* | *-22* | *-5.4* | *2250* |
| All cropping | 512 | 384 | -128 | -25.0 | 2771 |
| Grazing | 1631 | 1604 | -27 | -1.7 | 5660 |
| Conservation | 0 | 0 | 0 | 0.0 | 0 |
| Other/unknown | 26 | 26 | 0 | 0.0 | 16 |
| **Total** | **2169** | **2014** | **-155** | **-7.1** | |

**Table 3.6.** Changes in the amount of Buloke-dominated roadside and patches and the number of scattered Buloke trees between 1997 and 2004 in the study area including both South Australia and Victoria combined.

| Category | 1997 | 2004 | change | % change | % change p.a. | land use area |
|---|---|---|---|---|---|---|
| Patches (ha) | 201.2 | 201.1 | 0.1 | 0.0 | 0 | |
| Roadside (ha) | 162.0 | 162.6 | 0.6 | 0.0 | 0 | |
| **Trees:** | | | | | | |
| Pivots | 1225 | 0 | 1225 | -100.0 | — | 7606 |
| Extensive cropping | 26664 | 25122 | 1542 | -5.8 | -0.83 | 71226 |
| All cropping | 27889 | 25122 | 2767 | -9.9 | -1.42 | 78832 |
| Grazing | 28830 | 26065 | 2765 | -9.6 | -1.37 | 79432 |
| Conservation | 166 | 166 | 0 | 0.0 | 0 | 1846 |
| Other/unknown | 124 | 124 | 0 | 0.0 | 0 | 182 |
| **Total** | **57009** | **51477** | **-5532** | **-9.7** | **-1.4** | **239124** |

## 3.3.4 Discussion

The amount of change in area occupied by Buloke-dominated roadside vegetation and Buloke woodland patches between 1997–2004 was negligible, suggesting that neither clearing nor expansion of these areas was occurring. However, the annual percentage loss of paddock Buloke trees was higher during the period 1997–2004 than it was when calculated for the period 1963–2004. There was a loss of approximately 790 trees per year across the recent comparison study area between 1997–2004. It is not known how much of this was due to permit or other deliberate clearing and how much was caused by deaths due to other factors such as stubble fires and blowovers. However, it can reasonably be assumed that those trees lost from areas which centre pivots occupied by the end of the period were mostly cleared for that purpose. Although centre pivots occupied only 3% of the recent comparison study area, they accounted for 22% of paddock tree losses between 1997 and 2004.

The land use mapping for Victoria is coarse-grained and the true extent of land which is regularly cropped is greater than that which is mapped as such. Most cropping is in the northern half of the recent comparison study area, yet in recent years drier weather has driven an increase in cropping activity further south, as far as the southern boundary of the distribution of Buloke. Therefore, it is difficult to attribute the high death rates observed in land currently mapped as grazing to that practice alone, and caution should be exercised in the interpretation of results for different land use categories.

Tree losses due to clearing for centre pivots aside, the spatial distribution of trees lost between 1997 and 2004 was clumped. While many grid squares examined lost no or very few trees, others lost large numbers. Notable differences were also evident between adjacent paddocks. This would suggest that deliberate tree removal or different land management practices, such as stubble burning, contributed substantially to the losses of trees from cropping and grazing landscapes.

It is important to note that although every effort was made to map all Buloke within the recent comparison study area, a small percentage of the distribution of Buloke within the RtBC range will have been missed, due either to incomplete aerial photo coverage or incorrect classification. The lack of mapping for Buloke woodland in Victoria made identification of areas of Buloke within large patches difficult, although field examination revealed few patches of Buloke-dominated woodland with large trees in such patches. Detailed on-ground mapping is required to determine the full extent of Buloke in patches, and the degree to which such trees are utilised by RtBCs.

# 4. FACTORS INFLUENCING STRINGYBARK SEED AVAILABILITY

## 4.1 IMPACT OF FIRE ON STRINGYBARK SEED RESOURCES

### 4.1.1 Introduction

Stringybark woodland occurs in patches throughout the RtBC range, and seed is available year-round, although availability and food value is spatially and temporally variable. The distribution of *E. arenacea* and *E. baxteri* is likely to ultimately limit the range of the RtBC, as although Buloke is distributed much more widely than the RtBC, its fruit is available for only part of the year. The suitability of *E. arenacea* and *E. baxteri* is similar, but the two species differ in flowering phenology, with fruit of only one or the other species available in some years. However, fruiting of both species is affected negatively by scorching of the canopy.

Deliberate burning of stringybark woodlands has been widely practiced within the RtBC range over the last 100 years. It is likely that indigenous people performed deliberate burns for hunting and to clear a path through dense vegetation for travel, but the extent and frequency of these burns in stringybark vegetation is a matter of speculation. From the 1900s to the 1920s, burns were performed predominantly by farmers to reduce the risk of wildfire and increase the palatability of plants for cattle (Luke and MacArthur 1977). These burns are reported to have been performed every few years, or “as often as a fire would carry through the scrub” (Cleary pers. comm.). A fire exclusion policy was implemented in the 1920s, but was terminated 35 years later following a series of intense wildfires.

Prescribed burns (performed by fire management authorities) were introduced in the mid 1950s. The objective of these burns was essentially to reduce the extent of wildfires and aid in wildfire suppression efforts. Broad-acre burning was practised throughout the region until 1989, when a moratorium on block burns was introduced for the Horsham region. Block burns were replaced by strip burns: the burning of strips along the perimeters of remnants and fire tracks to reduce fuel loads in strategic areas. This moratorium was introduced specifically to address concerns about food supply for the RtBC.

More recently, the moratorium was lifted in favour of block burns that produced low levels of canopy scorch. This followed a study by the Victorian Department of Sustainability and Environment (DSE) (Rudolph 2006) trialling a series of cool burns designed to minimise canopy scorch, while also documenting seed fall (% of capsules opening) and bud scorch pre and post fire. The trial burns were successful in that they met both fuel reduction objectives and a self-imposed target of <30% seed release after fire. This study followed an earlier study (Koch 2003) comparing capsule availability and habitat use by the RtBC between burnt areas (time since fire age classes: 3, 5, 6, 7, 9 and 11 years post-fire) and nearby long unburnt areas (>25 years since fire). The study indicated that prescribed burns significantly reduce seed capsule availability in stringybark woodlands over an approximate 10 year post-fire period and confirmed that rates of habitat use increase sharply with increased time since fire. The study also suggested that food supply limits RtBC population size and indicated that reducing the intensity of burns, or the total area burnt, was an effective way to increase food supply to the RtBC (Koch 2003).

There are currently three fire management authorities operating in the range of the RtBC: the DSE (Portland and Horsham Fire Management Districts, corresponding to the southern and northern Victorian parts of the range, respectively), the South Australian Department of Environment and Heritage (DEH, South Australia) and Forestry SA. DEH and Forestry SA regularly perform prescribed burns in parks and reserves of South Australia according to a schedule of prescribed burns that varies depending on the area being managed. However, the majority of stringybark habitat occurring within the RtBC range is present as large blocks of crown land within Victoria. Correspondingly, the majority of prescribed burns are conducted in Victoria by DSE. The DSE is obliged under the Forests Act 1958 to reduce fuel hazards to help prevent wildfire.

Burns occurring on private land are a minor component of the total area burnt, particularly within South Australia where burns on private land are classed as vegetation clearance under the Native Vegetation Act (1991) and require a planning permit and approval by the Native Vegetation Council. Within SA, the vast majority (80%) of the stringybark habitat occurs on private land and burns on private land make up a relatively minor component of the total area burnt.

### 4.1.2 Methods

Existing fire history data in a GIS were used to document time since fire in stringybark woodlands across the region. Datasets used comprised the Fire100_year dataset supplied by DSE for Victoria and additional separate fire history datasets from DEH and Forestry SA in South Australia. These data were overlaid with a vegetation layer depicting stringybark-dominated woodlands within the range of the RtBC and intersected to create a new layer of burnt stringybark woodlands. The area of stringybark burnt in the past 10 years and the area of stringybark burnt in the total period of recorded fire history (1939–2006) were calculated. Areas were also identified as either prescribed burnt or burnt by wildfire. Fires were already categorised in this way for DSE and DEH datasets but not for ForestrySA data. For ForestrySA data, all burns were assumed to be prescribed burns unless they occurred in a year when extensive wildfires were known to occur in the region (eg. 1939, 1944, 1962), in which case they were assumed to be wildfires. Although there doubtless was a degree of error in this categorisation, the area of stringybark burnt in lands managed by ForestrySA makes up a small proportion of the total, so the overall error margin is likely to be very small.

The impact of prescribed burns on food supply across the range was then calculated as the proportion of the total area of stringybark prescribed burnt in the past 10 years multiplied by the average reduction in crop size attributable to prescribed burns over a 10 year post-fire period. These latter data were taken directly from a comprehensive study by Koch (2003) which determined the impact of time since the last prescribed burn on stringybark seed availability to the RtBC.

### 4.1.3 Results

**Time since fire across the RtBC range**

Table 4.1 shows the total area prescribed burnt and burnt by wildfire in the 10 years to 2007 and for the 67 year period for which fire history data has been collected. Approximately 11.1% of stringybark woodlands have been prescribed burnt in the past 10 years. An additional 15.9% have been burnt by wildfire. Therefore, approximately 27% of all stringybark woodlands in the range of the RtBC have been burnt in the past 10 years.

Approximately 345% of stringybark woodlands have been burnt over the past 67 years, giving a minimum average fire frequency of 19.4 years. The area of stringybark burnt by wildfire or prescribed burns in South Australia is a very small proportion of the total area of stringybark burnt.

**Table 4.1.** Area (ha) of stringybark woodland prescribed burnt and burnt by wildfire in the last 10 years and in the total period of mapped fire history (1939-2006).

| **Fire Authority** | **Prescribed Burns (ha)** | | **Wildfire (ha)** | | **Total (ha)** | |
|---|---|---|---|---|---|---|
| | Last 10 years | All years mapped | Last 10 years | All years mapped | Last 10 years | All years mapped |
| **DSE (Vic Total)** | 30329 | 255017 | 44276 | 678864 | 75071 | 960746 |
| **% of Vic** | 13 | 107 | 19 | 286 | 32 | 404 |
| DEH (SA) | 58 | 58 | 42 | 4988 | 133 | 5143 |
| Forestry SA | 955 | 5828 | 564 | 1072 | 18 | 5383 |
| **SA Total** | 1013 | 5886 | 606 | 6061 | 151 | 10526 |
| **% of SA** | 2 | 13 | 1 | 14 | <1 | 24 |
| **TOTAL** | 31342 | 260903 | 44882 | 684925 | 75223 | 971273 |
| **SA % of total stringybark area burnt** | 3.23 | 2.26 | 1.35 | 0.88 | 0.20 | 1.08 |
| **VIC % of total stringybark area burnt** | 96.77 | 97.74 | 98.65 | 99.12 | 99.80 | 98.92 |
| **% of total stringybark area** | **11.13** | 92.66 | **15.94** | 243.26 | **26.72** | 344.96 |

**Impact of fire on food availability**

On average, trees in areas that have been prescribed burnt within the past 10 yearsproduce 49% as many seed capsules as those in long-unburnt areas (Koch 2003). The area of stringybark prescribed burnt in the past 10 years makes up 11.13% of the total area of stringybark feeding habitat. Therefore, the total reduction in stringybark seed availability in woodland blocks across the range of the RtBC attributable to prescribed burns is 0.11 x 0.49 = 0.0539 or 5.4%. Note, however, that these calculations do not take into account paddock trees which provide additional seed resources (the modelling in Section 6.4 provides a more comprehensive estimate based on realistic scenarios of improved fire management).

Assuming that wildfire produces a similar reduction in stringybark seed availability over a 10 year post-fire period, the reduction in seed capsule availability in woodland blocks attributable to wildfire is 0.16 x 0.49 = 0.078 or 7.8%, across the range of the RtBC.

### 4.1.4 Discussion

The results collectively indicate that the total reduction in food availability in woodlands attributable to prescribed burns is 5.4%. This estimate is simplistic, because the extent to which burnt areas are used by cockatoos is likely to depend on temporal aspects of food availability (Koch 2003), and consequently, food availability is a relative term. Furthermore, this does not take into account the greater food value of paddock trees, which are rarely burnt (see Section 4.2). Nevertheless, it provides a rough estimate of what can realistically be expected from changes in fire management. The overall increase in food supply we can expect from improvements to fire management (eg. reducing levels of canopy scorch, reducing the total area burnt) appears to be much lower than target increases outlined in the Draft RtBC Recovery Plan (50%, Burnard and Hill 2002). Therefore, improved fire management will need to be supplemented with revegetation or replacement of individual "paddock" trees on a broad scale to achieve higher percentage increases. These scenarios are discussed further in Section 6.4.

Having said this, there are three important reasons why improved fire management is an important tool for improving the conservation status of this bird: (1) improved fire management is the only way to increase food supply to the cockatoos in the short term, because the planted trees would probably take at least 15 years to reach levels of capsule availability comparable to those of established trees (Koch personal observation); and (2) prescribed burn management can potentially be adapted to minimise losses to seed availability during periods of temporary food shortage, such as when birds are feeding exclusively in *E. arenacea* or when capsule availability is generally low across the range. The RtBC Recovery Team recently begun a monitoring program for stringybark seed availability across the cockatoo's range and DSE fire management authorities in Portland and Horsham have started to factor seed availability for RtBCs into their prescribed burn regimes (Tucker pers. comm., McGuire pers. comm.). This is something that could potentially be improved when the results from the monitoring program become available. In practice, this would mean that fire management authorities concentrate prescribed burning for a given year in either *E. arenacea* or *E. baxteri* depending on which species is likely to be favoured by the RtBC.

A further 13.2% reduction in stringybark seed availability within woodland blocks is attributable to wildfire, assuming that wildfire produces a similar post-fire tree response. This finding indicates that wildfire currently plays a substantially greater role in reducing seed availability than prescribed burns. The frequency, intensity and extent of wildfires is likely to increase as our climate becomes hotter and drier, increasing the influence of wildfire on seed availability and narrowing the annual climatic window during which prescribed burns can safely be performed. The most catastrophic wildfires (large proportion of the stringybark remnants burnt) would likely produce a situation in which isolated paddock trees become crucial for the continued persistence of this bird, because seed availability would be virtually nonexistent in burnt areas. Therefore, protecting existing trees on private land and increasing the extent of paddock trees may be an important strategy to reduce the risk of a fire-induced population crash.

The actual reduction in food supply attributable to wildfire may be much greater (a more prolonged period of reduced seed production) than that due to prescribed burning, due to the higher fire intensities involved. More intense fires produce a greater terminal twig diameter (Cheney, pers. comm., Koch personal observation); i.e., larger branches are killed by fire. This increased dieback typically results in longer periods required to rebuild the canopy. In theory, increased dieback after fire will directly influence seed production because trees with more crown dieback need to invest more resources in rebuilding the canopy, leaving fewer resources available for reproductive functions (Reekie and Bazzaz 1987).

## *4.2 IMPACT OF GRAZING ON STRINGYBARK SEED RESOURCES*

### 4.2.1 Introduction

Grazing of native vegetation by livestock can impact on many aspects of vegetation condition and is widely known to reduce understorey diversity, suppress plant recruitment, reduce canopy cover and encourage weed invasion. Protection of native vegetation from grazing by livestock is generally accepted as being fundamental to the long term persistence of intact vegetation communities.

The RtBC feeds almost entirely on seed stored in the canopy and is commonly observed to feed on paddock trees (Hill pers. comm., Koch pers. obs.). Therefore, habitat complexity in terms of species and structural diversity does not appear to influence foraging site selection. The RtBC is unlikely to show a preference for intact vegetation communities *per se,* but it is known to select foraging sites that are high in capsule availability characteristics such as crop size (total number of capsules per tree) and capsule density (number of capsules per branch; Koch 2003). Consequently, the short to medium term impacts of grazing on food supply to the RtBC are likely to depend almost entirely on the productivity (capsule availability) of trees and canopy cover within grazed patches. The productivity of trees is influenced by both temporal factors (eg. natural periodicities between seed crops, time since fire) and spatial factors (eg. fire intensity; Koch 2003). However, studies have indicated that trees of both species tend to be highly synchronous within a site with regard to years of substantial seed production and years of negligible seed production (Koch 2003). The interaction between temporal and spatial factors no doubt contributes to variability in tree responses, but tree responses to spatial factors such as fire intensity are nonetheless discernable (Koch 2003). Longer term impacts on food supply are likely to result from incremental habitat loss associated with tree mortality and poor recruitment.

A large proportion of stringybark within the range of the RtBC is located on private land, and much of this is susceptible to the effects of grazing by livestock. This study aimed firstly to determine the proportion of stringybark in the range of the RtBC susceptible to grazing (not known to be fenced or actively protected); and secondly to assess the impact of grazing on the availability of stringybark seed to the RtBC in the short to medium term. Longer term impacts of grazing and other factors on stringybark food availability are considered in Section 4.3 and 4.4 in relation to rates of habitat loss.

## 4.2.2 Methods

**Study sites**

The study focussed on remnants of stringybark forest near Edenhope and Casterton in south-west Victoria and near Penola and Mt Gambier in south-east South Australia. The study sites were characterised by dry sclerophyll forest of predominantly *E. baxteri* or *E. arenacea*, in association with a heathy understorey. The stringybark of this region is restricted to infertile, well-drained, aeolian sands that occur as irregular, low dunes over fertile clay. The understorey is typically dominated by small shrubs such as *Leptospermum* spp., *Leucopogon* spp., *Astroloma* spp. and *Hibbertia* spp., with occasional tall shrubs such as *Banksia marginata* and *B. ornata* (see Purdie & Slayter 1976 for a detailed description of the understorey). *Xanthorrhoea australis* and *X. caespitosa* are prominent in some areas.

Ten sites were selected across the range of the RtBC, comprising five *E. baxteri* sites and five *E. arenacea* sites. Site selection was based on access to private land and the presence of a grazed patch of stringybark located within a 1 km radius of a suitable reference or ungrazed area. Grazed patches of stringybark ranged from 2–20 ha and were defined as groups of 5 or more trees where tree canopy cover (projective foliage cover) was at least 20%. The canopy cover guideline was based on the native vegetation remnant definition used by the DSE (DSE 2004). Ungrazed patches were typically large, publicly-managed remnants, usually Crown Land in Victoria and Conservation Parks in South Australia. All sites were dominated by a stringybark overstorey and were judged to represent either EVC 48 Heathy Woodland or EVC 179 Heathy Herb-rich Woodland.

Sampling points within patches were selected haphazardly by walking 50 m from the remnant edge (area cleared of vegetation for at least 20 m) into the interior of the remnant. In the case of small patches, the approximate centre of the remnant was usually selected for sampling. At each sampling point, 15 trees were haphazardly selected for estimates of capsule availability, canopy health and projective foliage cover. Other (site-level) measurements were made at sampling points to estimate recruitment, tree density and gap width.

**Tree measurements**

The canopy of the trees studied is modular in architecture and is made up of clumps of foliage held on branches of approximately 10–20 mm diameter, that are easily recognised (Koch personal observation). Each of these branches or clumps of canopy usually bears a similar quantity of foliage (Koch personal observation), and so these branches were used as the unit of measurement for estimates of habitat use and capsule availability.

Capsule density was estimated by counting the number of capsules per branch for three typical branches (branches bearing seed crops judged to be representative of most foliage), and calculating the average. Binoculars were used to count capsules where necessary. Crop size (size of the standing crop of capsules) was estimated by counting the number of fruiting branches per tree and multiplying by the mean capsule density per branch. Similar methods have been used by Koch (2003) for *E. baxteri* and *E. arenacea*, Maron and Lill (2004) for

*Allocasuarina luehmannii* and Yates et al. (1994) for *Eucalyptus salmonophloia* to estimate crop size and fruit density.

Canopy health was estimated according to the method used for Habitat Hectare assessments in Victoria (described in DSE 2004). The method involves estimating the percentage of healthy foliage present compared to that expected using a visual guide provided in the assessment manual. Projective foliage cover (also described in DSE 2004) was estimated for each tree.

**Site measurements**

At each sampling point, estimates were made of recruitment, tree density and canopy cover. Recruitment (the number of seedlings and saplings below 5 m height) and tree density (the number of living stringybark stems greater than 5 m height) were counted within a 20 x 20 m quadrat (measured with a flexible measuring tape and marked at each corner with flagging tape). This was repeated for three quadrats and the results averaged. The method for estimating canopy cover is also described in the Habitat Hectare Assessment Manual (DSE 2004).

**Food value of paddock trees**

An additional sample of paddock trees were scored for capsule density and crop size. A tree was defined as a paddock tree if it occurred as either a single tree or it occurred within a group of trees made up of fewer than five individual trees that were surrounded by cleared land on all sides for at least 20 m. A sub-set of two of the *E. baxteri* sites and two of the *E. arenacea* sites were selected for assessment of nearby paddock trees. All accessible paddock trees at the sites were scored. This produced a sample of approximately 10 trees per site.

**Statistical analysis**

The two species of stringybark were considered separately because they appeared to respond differently to grazing in terms of capsule density, crop size and recruitment (see results). Two-way (factorial) ANOVAs were used to test for the effects of grazing status (grazed and ungrazed groups), site and the interaction term for the response variables: capsule density, crop size and canopy health. Trees were replicates for these variables. One way ANOVAs were used to test for the effects of grazing status on canopy cover, recruitment and tree density, using sites as replicates.

Two-way ANOVAs were used to test for the effects of status (paddock and ungrazed groups), site and the interaction term on capsule density and crop size. Sample sizes for paddock and ungrazed groups were uneven (the test requires a balanced design) so samples were randomly deleted where necessary for the analysis, as suggested by Zar (1984).

The data for capsule density, crop size and recruitment were positively skewed but there was sufficient normality for these variables following log transformation. The ANOVA test is considered to be sufficiently robust to tolerate small departures from normality (Underwood 1981). In situations where the data are highly positively skewed, medians are a more accurate measure of central tendency than means. Medians and interquartile ranges (the 75$^{th}$ percentile) are given instead of means and standard errors where appropriate.

### 4.2.3 Results

**Capsule density, crop size and canopy health**

The effect of grazing depended upon which species of Stringybark was being considered. *E. arenacea* values were comparable between grazed and ungrazed areas for capsule density, crop size and canopy health (Table 4.2). The effect of grazing status was not significant for either capsule density ($F_{149, 140}$ = 0.71, $P$ = 0.4), crop size ($F_{149, 140}$ = 0.36, $P$ = 0.55) or canopy health ($F_{149, 140}$ = 1.5, $P$ = 0.20). The effect of site was also not significant for all variables.

For *E. baxteri*, trees in grazed sites were significantly higher than ungrazed sites in both capsule density and crop size. Comparison of median values (the most accurate measure of central tendency when the data are highly positively skewed) showed that grazed sites were twice as high in capsule density and substantially higher in crop size. The effect of site was also significant for capsule density ($F_{149, 140}$ = 37.2, $P$ <0.0001) and crop size ($F_{149, 140}$ = 17.8, $P$ <0.0001) but not for canopy health. The interaction term (between grazing status and site) was significant for crop size, indicating that the effect of grazing for this variable depended upon which site was being considered.

**Table 4.2.** Mean (± SE) capsule density, crop size and canopy health for trees in ungrazed and grazed sites. Significant differences based on 2-Way ANOVA tests are shown with asterisks (* P<0.05, ** P<0.01, *** P<0.001, ****P<0.0001; n, no. of trees = 75). Medians and Inter-quartile ranges are also given in parentheses for highly positively skewed variables.

| | Ungrazed | Grazed |
|---|---|---|
| ***E. arenacea*** | | |
| Capsule Density | 21 ± 3 | 21 ± 4 |
| | *(12 ± 90)* | *(8 ± 193)* |
| Crop Size | 1139 ± 217 | 969 ± 202 |
| | *(280 ± 11083)* | *(266 ± 11200)* |
| Canopy Health | 37 ± 1 | 39 ± 2 |
| ***E. baxteri*** | | |
| Capsule Density | 28 ± 4 | 32 ± 4** |
| | *(10 ± 143)* | *(20 ± 43)* |
| Crop Size | 1888 ± 498 | 091 ± 338*** |
| | *(933 ± 12600)* | *(1733 ± 51333)* |
| Canopy Health | 41 ± 1 | 43 ± 2 |

**Canopy cover, recruitment and tree density**

For *E. arenacea* sites, recruitment and tree density values were comparatively higher at grazed sites than ungrazed sites (Table 4.3). Values for canopy cover at grazed and ungrazed sites were comparable. The effect of grazing status was not significant for any variable.

For *E. baxteri* sites, recruitment was significantly lower at grazed sites than ungrazed sites. Values for canopy cover and tree density were comparable between grazed and ungrazed sites and the differences were not significant.

**Table 4.3.** Mean (± SE) canopy cover, recruitment and tree density for ungrazed and grazed sites. Significant differences based on 1-Way ANOVA tests are shown with asterisks (Levels: * P<0.05, ** P<0.01, *** P<0.001, ****P<0.0001; n, no. of sites = 5).

| | Ungrazed | Grazed |
|---|---|---|
| ***E. arenacea*** | | |
| Canopy Cover | 13 ± 1 | 12 ± 2 |
| Recruitment | 2.7 ± 1.5 | 4.4 ± 2.5 |
| Tree Density | 13.6 ± 4.4 | 18.6 ± 5.2 |
| ***E. baxteri*** | | |
| Canopy Cover | 11 ± 1 | 13 ± 2 |
| Recruitment | 3.2 ± 1.3 | 0.2 ± 0.1* |
| Tree Density | 9.5 ± 1.7 | 9 ± 2.1 |

**Food value of paddock trees**

Paddock trees were substantially higher than ungrazed trees in both capsule density and crop size for both species of stringybark (Table 4.4). For *E. arenacea*, capsule density for paddock trees was more than twice as high as that of trees in ungrazed areas and crop size for paddock trees was more than 27 times as high (based on comparisons of medians). For *E. baxteri*, capsule density for paddock trees was slightly greater (1.3 times) than that of trees in ungrazed areas while crop size for paddock trees was more than 12 times as high. Differences were significant for both species. The effect of site and the interaction term was also significant for all variables, indicating that the effect of status (ungrazed or paddock trees) was dependent on which site was being considered.

**Table 4.4.** Mean (± SE) capsule density and crop size for ungrazed trees and paddock trees. Significant differences based on 2-way ANOVA tests are shown with asterisks (Levels: * P<0.05, ** P<0.01, *** P<0.001, ****P<0.0001; n, no. of trees per species = 32). Medians and inter-quartile ranges are given in parentheses.

| | Ungrazed | Paddock |
|---|---|---|
| ***E. arenacea*** | | |
| Capsule Density | 32 ± 4 | 68 ± 9*** |
| | *(31 ± 47)* | *(60 ± 91)* |
| Crop Size | 1660 ± 443 | 19482 ± 4764**** |
| | *(633 ± 1691)* | *(17333 ± 20708)* |
| ***E. baxteri*** | | |
| Capsule Density | 40 ± 9 | 51 ± 14** |
| | *(5 ± 68)* | *(17 ± 77)* |
| Crop Size | 2649 ± 897 | 6397 ± 2506*** |
| | *(140 ± 1733)* | *(1733 ± 6233)* |

## 4.2.4 Discussion

**Effect of grazing on remnant woodlands**

The results of this study have some important implications for management of Red-tailed Black Cockatoo habitat on private land. The results suggest that in general, grazing of Stringybark woodlands has a limited impact on both capsule density and crop size (as well as canopy health), both of which have been demonstrated by Koch (2003) to be key aspects of food supply. Capsule availability (measured in terms of capsule density and crop size) was significantly greater in grazed *E. baxteri* woodlands, indicating that there may be some advantage conveyed to trees in grazed remnants on private land. However, this effect depended upon which site was being considered (the interaction between site and grazing status was significant), so it is not certain whether this effect was directly caused by grazing or some other effect associated with soil quality (since areas of private land are likely to be associated with more fertile soils), fire regime or landscape context. Alternatively, the increased productivity of *E. baxteri* trees in grazed patches may be associated with reduced competition for resources. *E. baxteri* woodlands are often associated with dense bracken in the understorey and stringybarks are relatively shallow-rooted trees compared with, say, Red

Gums (Koch pers. obsv.). Grazing may substantially reduce the cover of bracken and hence reduce competition for resources, increasing the productivity of trees.

Site-based measures of food availability such as tree density and canopy cover were also surprisingly similar between grazed and ungrazed remnants for both species. These results indicate that, despite lower levels of recruitment in terms of number of seedlings (in *E. baxteri* at least), woodlands in grazed areas nevertheless maintain a comparable stem density. Of course, the age structure of trees may be different in grazed areas, with a higher proportion of older trees. An alternative explanation is that recruitment still occurs in grazed patches, but it tends to be episodic. Relatively high levels of recruitment may occur at times when pastures are in good condition and stock spend less time in remnant vegetation.

**Food value of paddock trees**

Paddock trees were remarkably productive when compared with trees in adjacent woodlands on public land. *E. arenacea* paddock trees were approximately 27 times higher and *E. baxteri* paddock trees approximately 12 times higher in crop size. These results were based on a low sample size (2 sites per species, 8 trees per treatment per site) and the statistical analysis indicated that the effect was site-dependent. The findings are therefore preliminary and need to be interpreted as such. Nevetheless, they are consistent with the authors' observations that paddock trees are generally far more productive than trees in forested areas.

Studies of edge trees in remnant woodlands suggest that reduced competition for light is a key factor driving this enhanced productivity (Koch 2003). However, other factors such as soil nutrient and moisture availability or even superphosphate application may be involved and further research is clearly warranted, particularly if replanting of stringybarks is deemed to be an important means of increasing stringybark seed availability (discussed further in Section 6.3).

**Implications for management**

Obviously it would be unwise to recommend grazing as a management action for remnants on private land due to probable negative effects on overall biodiversity, but the findings do suggest that fencing off of stringybark remnants will have little or no benefit in terms of improving food supply to the RtBC. Instead, funding for RtBC recovery should be directed towards the protection and replanting of individual paddock trees or trees at low density, particularly in areas dominated by *E. arenacea*. The planting of individual trees, potentially within double fencelines, represents a real possibility to enhance food supply on private land. Large tree canopies with large and dense seed crops are likely to enhance rate of seed intake due to lower search times and flights between trees. Foraging observations suggest that, over the course of a day, this can have a large impact on rate of seed intake. Paddock trees may also be important during the breeding season when rate of seed intake is likely to be critically important for nesting success. On the other hand, the benefits of paddock trees for foraging cockatoos are likely to diminish with increased distance from core habitat areas due to the increased energy expenditure associated with flying long distances. Saunders (1989) demonstrated the importance of feeding habitat proximity on the breeding success and conservation status of Carnaby's Cockatoo *Calyptorhynchus latirostris*. Observations also suggest that the RtBC rarely flies across cleared land unless necessary for nesting or feeding, possibly due to increased exposure to predators such as Wedge-tailed Eagles (Hill pers. comm.). Where possible, therefore, stringybarks should be planted near a vegetated roadside, state forest or conservation park to minimise flight times and travel across open areas. See Appendix 5 for priority areas for planting stringybark trees.

## *4.3 HISTORICAL CHANGE IN STRINGYBARK SEED AVAILABILITY (1947-2004)*

### 4.3.1 Introduction

Stringybark seed availability is now considered to be one of the most important factors limiting the recovery of the RtBC population (Koch 2003, Burnard & Hill 2003). Field observations, phenological information for the two species of stringybark and rates of habitat use data suggests that food availability is generally low across the RtBC range, particularly in years when only one species of stringybark has recently produced a mature seed crop (Koch 2003). It is therefore important to determine how historical rates of change data vary in different parts of the RtBC range and to identify the land uses in which the greatest rates of change occurred. In particular, historical change data can help to identify long term trends in habitat loss that may not be apparent in more recent change data. This section documents rates of stringybark habitat loss between 1947 and 2004, stratified separately by land use, NRM region, habitat type (remnant woodlands or paddock trees), stringybark species and land tenure.

### 4.3.2 Methods

The methods used for determining rates of loss for the two stringybark species are described in Section 2.

### 4.3.3 Results

During the 58 year period from 1947 to 2004, there was a 44.6% total loss of stringybark tree cover recorded from sample aerial photo strips (Table 4.5). Losses were comparable between species, but losses on private land were far more severe than losses on public land.

**Table 4.5.** Changes in the area of *E. arenacea* and *E. baxteri* tree cover on public and private land between 1947 and 2004, across the range of the RtBC. (N = no. of 1 x 1 km samples).

| Stringybark Species | Tenure | N | 1947 (ha) | Change to 2004 (ha) | % Change | % Annual Change |
|---|---|---|---|---|---|---|
| *E. arenacea* | Private | 69 | 3497 | -2414 | -69.03 | -1.19 |
| *E. arenacea* | Public | 30 | 2539 | -176 | -6.93 | -0.12 |
| **all *E. arenacea*** | | 99 | **6036** | **-2590** | -42.91 | -0.74 |
| *E. baxteri* | Private | 122 | 5612 | -3698 | -65.89 | -1.14 |
| *E. baxteri* | Public | 41 | 2465 | -9 | -0.37 | -0.01 |
| **all *E. baxteri*** | | 163 | **8077** | **-3707** | -45.90 | -0.79 |
| **All stringybark** | | 262 | 14113 | -6297 | -44.62 | -0.77 |

Rates of habitat loss are compared between remnant woodlands and paddock trees in Table 4.6. Rates of habitat loss were much higher for paddock trees than remnant woodlands for both species of stringybark. A small net gain was recorded for tree cover associated with *E. baxteri.*

**Table 4.6.** Comparison of losses to vegetation categorised as either remnant woodland or paddock trees for *E. arenacea* and *E. baxteri* between 1947 and 2004, across the range of the RtBC. (N = no. of 1 x 1 km samples). Samples that were characterised as "mixed" comprised a mixture of paddock trees and remnant woodlands.

| Stringybark Species | Habitat Type | N | 1947 (ha) | Change to 2004 (ha) | % Change | % Annual Change |
|---|---|---|---|---|---|---|
| *E. arenacea* | Remnant | 24 | 2061 | -208 | -10.09 | -0.17 |
| | Paddock | 48 | 2099 | -1742 | -82.99 | -1.43 |
| | Mixed | 27 | 1876 | -640 | -34.12 | -0.59 |
| *E. baxteri* | Remnant | 27 | 1819 | 138 | 7.59 | 0.13 |
| | Paddock | 85 | 3447 | -2859 | -82.94 | -1.43 |
| | Mixed | 51 | 2811 | -986 | -35.08 | -0.60 |
| All stringybark | Remnant | 51 | 3880 | -70 | -1.80 | -0.03 |
| | Paddock | 133 | 5546 | -4601 | -82.96 | -1.43 |
| | Mixed | 78 | 4687 | -1626 | -34.69 | -0.60 |

Rates of change data for stringybark woodlands stratified by land use type are given in Table 4.7. The highest losses were recorded for areas dominated by centre pivot agriculture, but this made up a very small number of samples. Losses in cropping areas were also comparatively higher than losses in grazing and plantation areas, which were comparable.

**Table 4.7.** Changes in the area of stringybark tree cover across the range of the RtBC between 1947 and 2004, stratified by land use type. Values for *E. baxteri* and *E. arenacea* were similar and so were combined for this summary (N = no. of 1 x1 km samples).

| Land Use Type | N | 1947 (ha) | Change to 2004 (ha) | % Change | % Annual Change |
|---|---|---|---|---|---|
| Conservation | 72 | 5094 | -200 | -3.93 | -0.07 |
| Cropping | 27 | 1263 | -963 | -76.25 | -1.31 |
| Grazing | 113 | 5717 | -3912 | -68.43 | -1.18 |
| Pivot | 2 | 160 | -142 | -88.75 | -1.53 |
| Plantation | 48 | 1879 | -1080 | -57.48 | -0.99 |

Table 4.8 shows how losses of stringybark food resources are divided between natural resource management regions. The highest losses were recorded within the SENRM Board region (South Australian part of the RtBC range). The most striking difference, however, was between the WCMA and GHCMA regions, which correspond to the northern Victorian and southern Victorian parts of the RtBC range, respectively. Losses within the WCMA region were very small in comparison to those from the GHCMA region.

**Table 4.8.** Changes in the area of stringybark tree cover across the range of the RtBC between 1947 and 2004, stratified by NRM region and stringybark species (N = no. of 1 x1 km samples).

| Region | Stringybark Species | N | 1947 (ha) | Change to 2004 (ha) | % Change | % Annual Change |
|---|---|---|---|---|---|---|
| **WCMA** | *E. arenacea* | 45 | 3262 | -755 | -23.15 | -0.40 |
| | *E. baxteri* | 35 | 1300 | -7 | -0.54 | -0.01 |
| | All stringybark | 80 | 4562 | -762 | -16.70 | -0.29 |
| **GHCMA** | *E. arenacea* | 17 | 1193 | -505 | -42.33 | -0.73 |
| | *E. baxteri* | 66 | 3202 | -1263 | -39.44 | -0.68 |
| | All stringybark | 83 | 4395 | -1768 | -40.23 | -0.69 |
| **SENRM** | *E. arenacea* | 37 | 1581 | -1330 | -84.12 | -1.45 |
| | *E. baxteri* | 62 | 3575 | -2437 | -68.17 | -1.18 |
| | All stringybark | 99 | 5156 | -3767 | -73.06 | -1.26 |

## 4.3.4 Discussion

**Overall rates of loss**

The overall percentage change in area of habitat was similar for *E. arenacea* (43%) and *E. baxteri* (46%) woodlands, with an overall annual rate of loss of 0.77 for the two species of stringybark combined. The percentage of habitat loss in terms of area was much higher for paddock trees than for remnant woodlands, with “mixed” samples having an intermediate percentage loss. There are likely to be several reasons for the extensive loss of habitat during this time. Firstly, clearing of stringybark for grazing purposes was more likely to occur in the later part of the twentieth century due to a series of government policies demanding increased agricultural productivity (Hedditch 2007). Clearing of land for grazing on the poorer soils associated with stringybark woodlands was much more likely as European settlement increased and grazing practices intensified. Secondly, stringybark felling for fence posts was widely practised during this time. Thirdly, plantation forestry gained a strong foothold in the region during this period and large areas were cleared and converted to softwood plantations. Fourthly, agricultural practices intensified during this period from simple grazing across large areas to centre pivot agriculture and cropping, resulting in further losses.

**Habitat loss for paddock trees vs remnant woodlands**

Total percentage habitat loss was much higher for paddock trees (-83%) than for remnant woodland (-1.8%). This somewhat surprising result indicates that the vast majority of clearing occurring during the period 1947–2004 was associated with paddock trees, suggesting that broad-scale clearing of stringybark was relatively uncommon in the post-war period. Increased death rates may have contributed to the high rates of tree loss. Paddock trees are subject to a range of stresses not present in remnant woodlands that are likely to increase death rates. Some of these stresses include: cattle rubbing (Hill unpublished data) and bark eating by cattle (Scotty pers. comm.), wind damage (Koch pers. obsv.) and possibly the use of synthetic fertilizers. On the other hand, those paddock trees that do survive generally produce much larger seed crops on much larger canopies than trees in remnant woodlands (particularly *E. arenacea;* see Section 4.2), suggesting that to some extent they benefit from the reduced competition and perhaps from increased soil fertility. Stringybarks are currently considered to be messy and undesirable as paddock trees by some farmers (Brady pers. comm.) and were probably more likely to be cleared or neglected than gum species. It was not possible to determine the relative impacts of deliberate clearing and increased tree stress on paddock tree loss from this study.

In any case, the extensive loss of paddock trees recorded in this study is an important new finding, particularly given that the pre-European EVC mapping (on which previous studies of habitat loss have been based eg. Koch 2003, Burnard and Hill 2002) does not pick up paddock trees. Paddock trees are an important food resource because they tend to be much higher in food value per tree (although they occur at far fewer trees per hectare than woodland). A single paddock tree can support a small flock of cockatoos for many hours (Koch pers. obsv.), which would likely increase foraging efficiency through reduced search times and flight energy requirements. Paddock trees may be a particularly valuable resource during the breeding season, when the energy demands on the breeding pair are greatest.

**Habitat loss for different land use types**

Total percentage habitat loss was highest in centre pivot agriculture (89% loss) land use areas, followed by: cropping (76%), grazing (68%) or plantation (57%) land use areas. The present study indicates that historical rates of loss in areas of agricultural intensification (centre pivot agriculture and cropping) are substantially greater than rates of loss in areas dominated by plantation forestry. This is surprising because softwood plantations are generally planted on sandy soils associated with stringybark vegetation communities, whereas agricultural activities such as cropping are generally better suited to more fertile soil types. On the other hand, most of the habitat loss for stringybark was associated with paddock trees. Paddock trees are likely to be more susceptible to clearing in areas of agricultural intensification, and death rates associated with increased tree stress may also be higher. However, it is important to remember that centre pivot agriculture and cropping areas made up only 0.8% and 10% of samples, respectively (Table 4.7), indicating that they are a relatively minor component of land use types associated with stringybark vegetation communities. Furthermore, recent change data (Section 4.3) indicates that ongoing habitat loss attributable to cropping is negligible.

**Habitat loss for different NRM regions**

Total percentage habitat loss within the Wimmera CMA region (17%) was substantially lower than that of both the Glenelg-Hopkins CMA (40%) and South-east NRM (73%) regions. This result contrasts markedly with rates of loss data for Buloke, which was associated with high rates of loss in the Wimmera region over the historical period 1963-2004. One possible explanation is that a higher proportion of stringybark occurs on public land within the Wimmera CMA region. However, numbers of samples occurring on public and private land were comparable between Wimmera and Glenelg-Hopkins CMA regions of Victoria (unpublished data). A more likely explanation is that clearing for plantations, a major factor contributing to historical losses of stringybark woodlands, is virtually nonexistent in the Wimmera region (where no samples were classified as plantation) but was widespread in the Glenelg-Hopkins CMA (and South-east NRM) regions.

## 4.4 RECENT CHANGES IN STRINGYBARK DISTRIBUTION

### 4.4.1 Introduction

In the preceding section, it was determined that all the major land use types (grazing, plantations, cropping) occurring on soils associated with stringybark woodlands have been associated with losses of stringybark tree cover over the past 60 years. A somewhat unexpected finding was that historical rates of habitat loss in the Wimmera CMA region were markedly lower than those of both the Glenelg-Hopkins CMA and South-east NRM regions. This section seeks to determine the recent rates of habitat loss in the RtBC range during the period 1997 to 2004, stratified separately by land use, NRM region, habitat type (remnant woodlands or paddock trees), stringybark species and land tenure.

### 4.4.2 Methods

The methods used for determining rates of loss for the two stringybark species are described in Section 2.

### 4.4.3 Results

Recent rates of habitat loss were minor across the range of the RtBC, with less than 1% total stringybark loss recorded in all three NRM regions (Table 4.9). Annual rates of habitat loss were much higher for *E. arenacea* than *E. baxteri* within the GHCMA region, and were much higher for *E. baxteri* than *E. arenacea* in the SENRM region. Zero habitat loss was recorded within the WCMA region. The mean (±SE) percentage change across all samples was -1.04 ± 0.43 for *E. arenacea* and -1.85 ± 0.98 for *E. baxteri.* These figures indicate that in general the variability among samples across the RtBC range was substantial.

**Table 4.9.** Changes in the area of stringybark woodlands between 1997/1992 and 2004, stratified by NRM region and stringybark species (N = no. of 1 x 1 km samples). Annual rates of loss for all regions combined were calculated as weighted averages (weighted by the number of samples in each region).

| Region | Stringybark Species | N | 1992/1997 (ha) | Change to 2004 (ha) | % Change | % Annual Change |
|---|---|---|---|---|---|---|
| WCMA | *E. arenacea* | 49 | 2775 | 0 | 0.00 | 0.00 |
| | *E. baxteri* | 37 | 1340 | 0 | 0.00 | 0.00 |
| | All stringybark | 86 | 4115 | 0 | 0.00 | 0.00 |
| GHCMA | *E. arenacea* | 17 | 708 | -20 | -2.82 | -0.24 |
| | *E. baxteri* | 69 | 2045 | 17 | 0.83 | 0.07 |
| | All stringybark | 86 | 2753 | -3 | -0.11 | -0.01 |
| SENRM | *E. arenacea* | 40 | 346 | 0 | 0.00 | 0.00 |
| | *E. baxteri* | 73 | 1381 | -11 | -0.80 | -0.11 |
| | All stringybark | 113 | 1727 | -11 | -0.64 | -0.09 |
| All Regions | *E. arenacea* | 106 | 3829 | -20.00 | -0.52 | -0.04 |
| | *E. baxteri* | 179 | 4766 | 6.00 | 0.13 | -0.02 |
| | All stringybark | 285 | 8595 | -14.00 | -0.16 | -0.04 |

Rates of habitat loss were generally higher on private land than public land (Table 4.10). However, a net gain was recorded for private land within the WCMA region (compared to a loss on public land for this region). In contrast, a small net gain was recorded for the GHCMA region on public land and a loss recorded on private land.

**Table 4.10.** Changes in the area of *E. arenacea* and *E. baxteri* woodlands on public and private land between 1997/1992 and 2004, across the range of the RtBC. (N = no. of 1 x 1 km samples). Annual rates of loss for all regions combined were calculated as weighted averages (weighted by the number of samples in each region).

| Region | Tenure | N | 1992/1997 (ha) | Change to 2004 (ha) | % Change | % Annual Change |
|---|---|---|---|---|---|---|
| WCMA | Private | 41 | 1087 | 4 | 0.37 | 0.05 |
| | Public | 45 | 3028 | -4 | -0.13 | -0.02 |
| GHCMA | Private | 59 | 876 | -28 | -3.20 | -0.27 |
| | Public | 27 | 1877 | 25 | 1.33 | 0.11 |
| SENRM | Private | 109 | 1447 | -11 | -0.76 | -0.11 |
| | Public | 4 | 280 | 0 | 0.00 | 0.00 |
| All regions | Private | 209 | 3410 | -35 | -1.03 | -0.12 |
| | Public | 76 | 5185 | 21 | 0.41 | 0.03 |

Rates of habitat loss are compared between habitat types (remnant woodland, paddock trees or “mixed” - a combination of both) for the three NRM regions in Table 4.11. Losses were associated only with paddock trees and were highest in the GHCMA region, followed by the SENRM region and then by the WCMA region. Net gains in stringybark tree cover were recorded for the WCMA and GHCMA regions.

**Table 4.11.** Comparison of losses to vegetation categorised as either remnant woodland, paddock trees or "mixed" (a combination of both) between 1997/1992 and 2004, stratified by NRM region (N = no. of 1 x 1 km samples). Annual rates of loss for all regions combined were calculated as weighted averages (weighted by the number of samples in each region).

| Region | Habitat Type | N | 1992/1997 (ha) | Change to 2004 (ha) | % Change | % Annual Change |
|---|---|---|---|---|---|---|
| WCMA | Remnant | 33 | 2611 | 1 | 0.04 | 0.01 |
| | Mixed | 31 | 1376 | 0 | 0.00 | 0.00 |
| | Paddock | 22 | 128 | -1 | -0.78 | -0.11 |
| GHCMA | Remnant | 22 | 1501 | 9 | 0.60 | 0.05 |
| | Mixed | 33 | 1013 | 0 | 0.00 | 0.00 |
| | Paddock | 31 | 239 | -12 | -5.02 | -0.42 |
| SENRM | Remnant | 2 | 155 | 0 | 0.00 | 0.00 |
| | Mixed | 21 | 920 | 0 | 0.00 | 0.00 |
| | Paddock | 90 | 652 | -11 | -1.69 | -0.24 |
| All Regions | Remnant | 57 | 4267 | 10 | 0.23 | 0.02 |
| | Mixed | 85 | 3309 | 0 | 0.00 | 0.00 |
| | Paddock | 143 | 1019 | -24 | -2.36 | -0.26 |

Recent stringybark habitat losses are divided between existing land use types in Table 4.12. The highest annual rates of loss were recorded for areas dominated by centre pivot agriculture, although note that this made up a small number of samples. Interestingly, losses in plantation-dominated areas were relatively high for both the GHCMA and SENRM regions. Grazing produced substantial habitat losses in the GHCMA region, but relatively minor losses elsewhere. Zero losses were recorded within cropping areas and an annual net gain of 0.04% per annum was recorded for areas classified as conservation (state parks/reserves and national parks).

**Table 4.12.** Changes in the area of stringybark woodland across the range of the RtBC between 1997/1992 and 2004, stratified by land use type (N = no. of 1 x 1 km samples). Annual rates of loss for all regions combined were calculated as weighted averages (weighted by the number of samples in each region).

| Region | Land Use Type | N | 1992/1997 (ha) | Change to 2004 (ha) | % Change | % Annual Change |
|---|---|---|---|---|---|---|
| WCMA | Conservation | 46 | 3098 | 1 | 0.03 | 0.00 |
| | Cropping | 6 | 97 | 0 | 0.00 | 0.00 |
| | Grazing | 34 | 920 | -1 | -0.11 | -0.02 |
| GHCMA | Conservation | 27 | 1877 | 25 | 1.33 | 0.11 |
| | Grazing | 30 | 351 | -6 | -1.71 | -0.14 |
| | Plantation | 29 | 525 | -22 | -4.19 | -0.35 |
| SENRM | Conservation | 4 | 280 | 0 | 0.00 | 0.00 |
| | Cropping | 22 | 206 | 0 | 0.00 | 0.00 |
| | Grazing | 55 | 806 | 0 | 0.00 | 0.00 |
| | Pivot | 4 | 27 | -4 | -14.81 | -2.12 |
| | Plantation | 28 | 408 | -7 | -1.72 | -0.25 |
| All Regions | Conservation | 77 | 5255 | 26 | 0.49 | 0.04 |
| | Cropping | 28 | 303 | 0 | 0.00 | 0.00 |
| | Grazing | 119 | 2077 | -7 | -0.34 | -0.04 |
| | Pivot | 4 | 27 | -4 | -14.81 | -2.12 |
| | Plantation | 57 | 933 | -29 | -3.11 | -0.30 |

## 4.4.4 .Discussion

**Overall rates of loss**

Annual rates of loss were 0.04% for *E. arenacea*, 0.02% for *E. baxteri* and 0.04% for both species of stringybark combined. If current circumstances continue, we would lose 4% of stringybark woodlands within the range of the RtBC over the next 100 years. This rate of loss is substantially lower than that recorded for Buloke woodlands (1.2% per year, Section 3.3), but it is still unacceptable considering that stringybark seed availability is considered to be a primary factor limiting the recovery of the RtBC population (Koch 2003, Burnard and Hill

2003). Increasing the area of stringybark woodlands within the range of the RtBC is considered to be of fundamental importance to the recovery of this species.

**Rates of loss for paddock trees vs remnant woodlands**

Recent habitat loss was associated primarily with paddock trees (0.26% loss per annum). By contrast, remnant woodlands were associated with an overall net gain of 0.02% per annum. This is not surprising, considering that paddock trees are subject to a range of additional stresses (eg. cattle rubbing, wind exposure, felling for firewood etc.) that would likely accelerate rates of loss in the absence of natural regeneration and that remnant woodlands are better protected by vegetation clearance regulations.

The finding also confirms that significant advances in habitat protection on private land are required before overall gains in stringybark feeding habitat can be made. Rates of paddock tree loss were of particular concern for the South-east NRM (0.24% loss per annum) and Glenelg Hopkins CMA regions (0.42% loss per annum). Greater emphasis on the protection of paddock trees is required both in policy and planning, particularly considering the importance of this food resource in terms of seed availability (Section 4.2). It is also recommended that offset requirements for stringybarks in the range of the RtBC be increased. Currently, offset requirements within Victoria depend on the tree diameter, which is not necessarily the best indicator of food value for the RtBC (Koch 2003).

**Rates of loss for different land use types**

The highest rates of habitat loss were associated with centre pivot irrigation (2.12% loss per annum), followed by plantations (0.3% loss per annum), then grazing (0.04% loss per annum). Zero rates of habitat loss were recorded for both stringybark species for the cropping land use. This latter result is encouraging and suggests that the decline of stringybarks on private land can be negligible in the absence of deliberate clearing, at least over the measured time scales (7 to 12 years).

On the other hand, the high rates of loss associated with the plantation industry are of particular concern given the rapid expansion of hardwood plantations in the region. Further investigation of the causes of losses in plantation areas is warranted, as it is not clear from this study whether losses are associated with increased mortality of individual trees (possibly associated with water availability) or with deliberate clearing. The high rates of loss associated with centre pivot irrigation are of less concern because it is a relatively minor land use associated with stringybark vegetation in the range of the RtBC (4 samples, Table 4.12). Nevertheless, all paddock tree losses will have a substantial impact on overall food supply (see Section 6.3) and the high rates of habitat loss associated with this land use highlight the need for more stringent clearing regulations and offset requirements for paddock trees.

# 5. FACTORS INFLUENCING NEST SITE AVAILABILITY

## 5.1 CURRENT STATE OF NESTING RESOURCES

### 5.1.1 Introduction

RtBCs nest in hollows in live or dead eucalypt trees. The majority of nesting occurs in River Red Gum, but has also been recorded in Yellow Gum *E. leucoxylon*, Manna Gum *E. viminalis* and Pink Gum *E. fasciculosa* as well as *E. baxteri* and *E. arenacea*. Any eucalypt which forms large hollows (entrance dimensions approx 19 x 18 cm; R. Hill unpublished data) is probably suitable as nesting habitat (R. Hill, pers. comm.). This section focuses primarily on gum woodlands and trees, particularly the widespread Red Gum and Yellow Gum, although is also includes some box eucalypts which occur in amongst areas of gum eucalypts (typically Grey Box *E. microcarpa*). There have been substantial changes to the availability of gum and box eucalypts over the past 100 years, as they occurred on more fertile soils and were extensively cleared and ringbarked. Many of these ringbarked trees formed suitable hollows for RtBCs as they decayed, but such trees are subject to high rates of loss due to blowovers in high winds and firewood collection. The West Wimmera Shire has introduced an Environmental Significance Overlay to protect these dead hollow trees.

A large proportion of RtBC nests are located in dead hollow trees, which can not be reliably identified using aerial photography. Therefore, assessment of these resources was not possible in this project. However, these dead hollow trees are being lost from the landscape, possibly at a rate more rapid than that affecting live paddock trees. Estimates of rates of loss as high as 4–7% per annum (Hill and Burnard 2001) suggest that existing dead trees in paddocks will be largely gone within 30 years. Future trends in tree maturation and hollow formation in live gums will therefore become important in the medium- to long-term and the assessment of the current availability of gums at different life stages will be critical to assessing these. The focus of this section, therefore, was to consider current living gum eucalypt resources and obtain information required to estimate the degree to which their availability may be expected to change over the next few hundred years.

Most areas where large old remnant trees persist are grazed, hampering recruitment. Yet in some areas, notably blocks of ungrazed public land and on roadsides, fairly dense growth of young trees can be observed. The growth rates of these eucalypts are much greater than those of Bulokes, but hollows suitable for occupancy by RtBCs are unlikely to form until the trees are at least 220 years old (Gibbons and Lindenmayer, 2002). The proportion of trees with hollows tends to increase linearly with tree size and age (Gibbons et al. 2000). Trees in which RtBCs nest average (± s.d.) 103 ± 41 cm DBH (Hill, unpublished data), and eucalypts of this size in other parts of temperate southern Australia are estimated at over 140 years old, with older trees most likely to contain hollows (Gibbons et al. 2000). Therefore, this part of the study aimed to document the spatial and size class distributions of gum eucalypt trees within the RtBC range.

### 5.1.2 Methods

As for Bulokes, gum and box eucalypts were considered to occur primarily as paddock trees, patches and reserves, and roadside woodland. Although this categorisation does not cover all occurrences of gums in the region, as areas of dense young regeneration such as that which may occur on dry lakebeds are excluded, it is inclusive of the vast majority. Furthermore, aerial photography and existing mapping was unreliable in distinguishing these regenerating areas.

Field surveys were undertaken to gather data on the size class distribution of trees in each of the selected categories. For paddock trees, the methodology followed that described in Section 2.1 for Bulokes, with the diameters of up to 20 trees in each of 16 randomly located paddocks recorded. For trees on roadsides and in patches and reserves, trees were counted and their diameters recorded within 20 x 20 m quadrats. The mean proportion of trees falling within each size class was calculated for each category of tree occurrence.

### 5.1.3 Results

Eucalypt species recorded in paddocks and quadrat were most frequently either River Red Gum or Yellow Gum. The stem density of trees in roadside quadrats averaged 19 ± 5 per 20 x

20 m, which equates to 468 per hectare. For quadrats in reserves and patches, stem density was lower at 5 ± 1, or 131 per hectare.

Paddock gum and box eucalypts were typically very large, with diameters of between 140 and 200 cm most frequently encountered (Figure 4.1). No young trees (< 40 cm DBH) or regeneration was evident in the surveyed paddocks. However, trees of large diameter were rarely recorded on roadsides and in patches or reserves, with less than 5% of trees in woodland patches and no recorded trees on roadsides reaching the mean size of trees in which RtBC nests have been identified. Most trees in woodland patches were < 60 cm diameter, and most roadside trees were < 40 cm diameter (Figures 4.2 and 4.3). Large gums (>100 cm DBH) occurred at low densities in both woodland and roadside sites. Woodland patches averaged 10 large old trees per hectare, while roadsides averaged just 4.6.

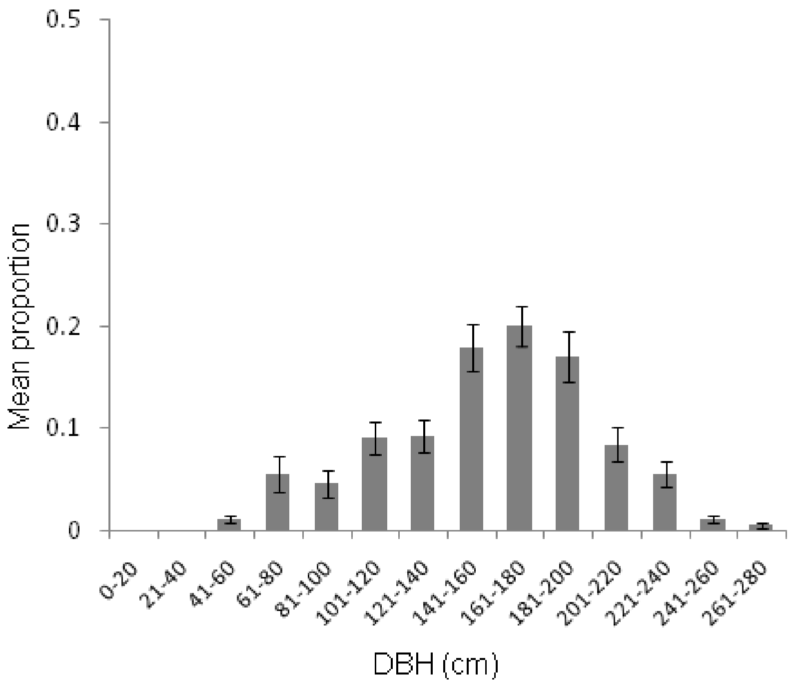



**Figure 4.1.** Mean (±1 s.e.) size class distribution of box and gum eucalypt paddock trees in the study area.

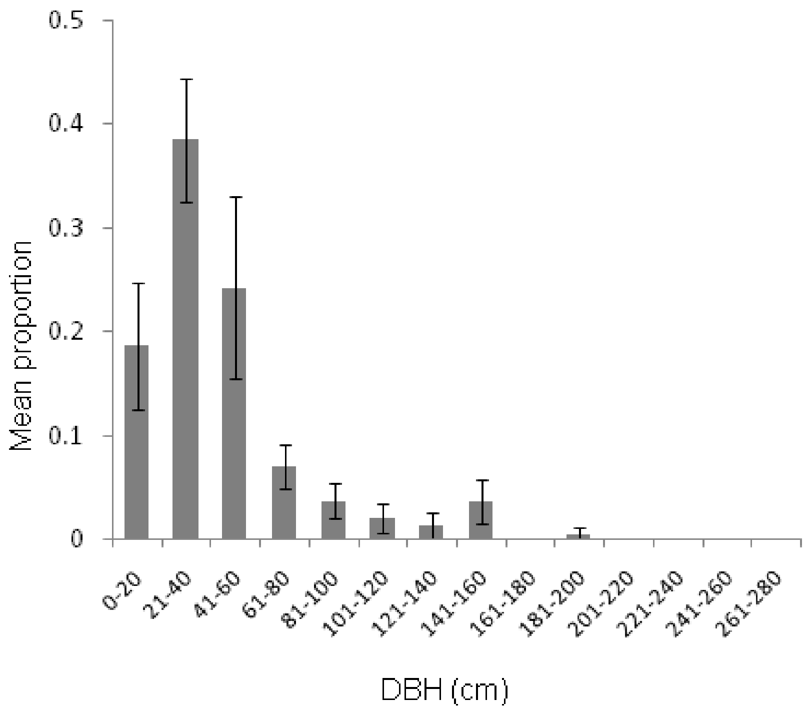



**Figure 4.2.** Mean (± 1 s.e.) size class distribution of box and gum eucalypts in woodland patches and reserves in the study area.

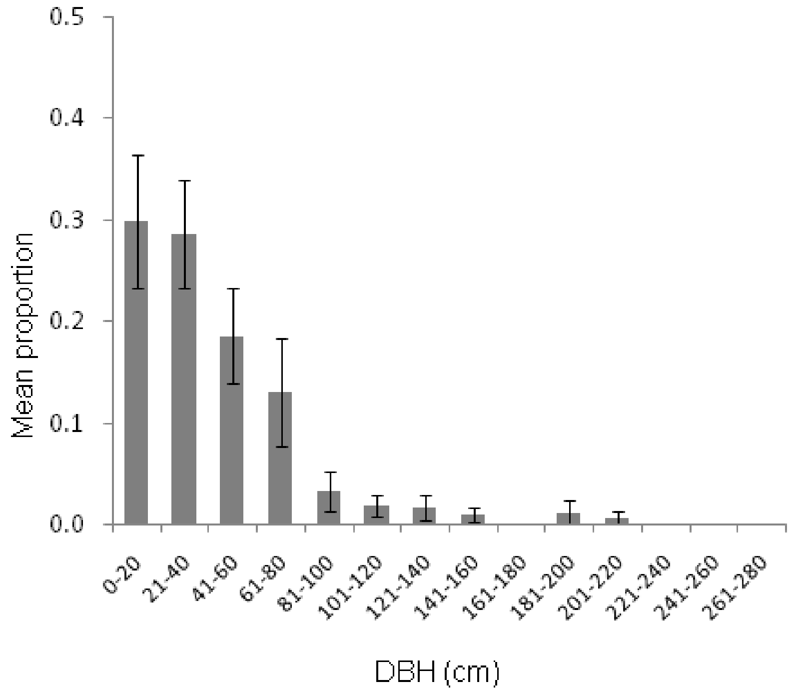



**Figure 4.3.** Mean (± 1 s.e.) size class distribution of box and gum eucalypts in roadside woodland in the study area.

By combining the size class distributions of the three stand categories weighted by the contribution of each category to the tree cover in the region, the overall size class distribution of gum eucalypts in the study area could be estimated (Figure 4.4). The much higher densities of trees on roadsides and in patches and reserves mean that small trees dominate, with about 75% of trees < 60 cm DBH. The distribution is slightly bimodal with a small peak representing the contribution of paddock trees in the 141-160 cm DBH size class, although this was only about 5% of trees.

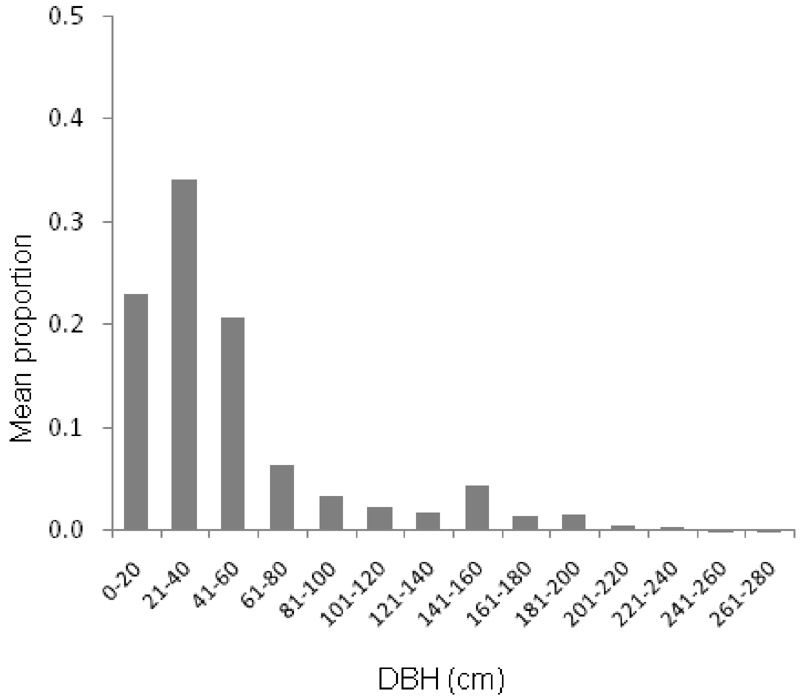



**Figure 4.4.** Estimated size class distribution of box and gum eucalypts across the study area.

### 5.1.4 Discussion

Gibbons et al. (2000) in a study of hollows in mixed eucalypt forest in eastern Victoria found that 50% of trees of 150 cm DBH had large (>10 cm diameter) hollows, compared to only 10% of 50 cm diameter trees. Age estimates for trees of 150 cm DBH for two species, *E. fastigata* and *E. obliqua* were 180–240 years. If a similar age to size class relationship occurs for potential RtBC nest tree species in the study area, we would conclude that only a few percent of trees occurring in reserves and on roadsides are likely to have large hollows, but the majority of paddock trees would be likely to contain such hollows. Although the study species and climatic conditions differ between Gibbons et al.'s (2000) study and the RtBC range, the relationships are likely to be similar, with the very large old trees typical of RtBC nest trees occurring frequently in paddocks but rarely in reserves and on roadsides.

The majority of known RtBC nest trees are located in agricultural land (Joseph et al. 1991; R. Hill, pers. comm.) although it is not known the degree to which this is a sampling artefact arising from the higher visibility of trees and nesting birds in open paddocks compared with denser and less-visited forest and woodland. However, given their much larger average diameter, paddock trees are more likely to be suitable than trees located in forests, so it might be expected that birds show a preference for paddock trees. The increased visibility of paddock trees may also make it easier for the cockatoos themselves to locate suitable hollows.

Paddock trees constitute a very small component of the estimated number of live gums in the study area. As with Buloke, the vast majority of live gum eucalypt trees occur on roadsides or in woodland patches, are relatively small (< 60 cm DBH) and are estimated to be younger than 100 years. However, these young trees occur at very high densities, and only a subset can be expected to mature into large, hollow-bearing trees suitable for the RtBC. In remnant gum woodland in the study area, trees >100 cm DBH occur at a very low density of approximately 10 per hectare, and no particularly dense stands of large eucalypts were located. For trees to attain a large diameter and thus have a high probability of producing large hollows, it seems likely that they would need to grow in a relatively low-competition environment, restricting densities of large trees to these low numbers. Therefore, very few of the densely occurring young trees on roadsides can be expected to mature to large trees. However, as large (>100 cm DBH) trees currently occur at a relatively low density of 4.6 per hectare on roadsides, there appears to be some opportunity for increased numbers of hollow-bearing trees in these areas over the next 100 years.

## *5.2 HISTORICAL CHANGE IN NESTING RESOURCE AVAILABILITY*

### 5.2.1 Introduction

Eucalypt woodlands throughout temperate Australia have been extensively cleared since European settlement, and several woodland types have been almost eliminated from parts of the wheatbelt (Yates and Hobbs 1997). As gum eucalypt woodlands in the study area occur on more fertile soils and were typically more open and grassy, they have suffered disproportionately from clearing. In order to place the current availability of gums in the context of historical change, this part of the study uses recent and historical aerial photographs to quantify the extent of losses between 1947–2004.

### 5.2.2 Methods

See Section 2 for a description of the methods used to determine change in the extent of gum woodlands.

### 5.2.3 Results

When averaged across 1 x 1 km samples, the mean area of gum eucalypt tree cover lost per sample was 5.4±1.1 ha. The average tree cover in 1947 per sample was 18.1±2.2 ha. In the following tables, the figures are based on total change in tree cover within the sampled area (165 $km^2$).

Between 1947 and 2004, 30% of gum eucalypt tree cover has been lost from the RtBC range (Table 5.1). This loss was entirely due to changes in tree cover on private land, with half the tree cover on private land disappearing over the 57 year period. This loss of tree cover from private land was largely due to high rates of loss of paddock trees, with 54% loss of paddock tree cover lost, compared with just 1.5% of tree cover in remnant patches (Table 5.2). A large proportion of trees was lost from all land use categories, except conservation, for which a slight gain in tree cover was recorded (Table 5.3). The greatest proportional loss was from areas currently under plantation forestry, with 69% of tree cover lost over the 57 years. Centre pivot irrigation and dryland cropping were the next most significant proportional contributors to tree cover loss, followed by grazing at 40%.

**Table 5.1.** Changes in the area (ha) of gum tree cover on public and private land between 1947 and 2004, across the range of the RtBC (N = no. of 1 x 1 km samples).

| Tenure | N | 1947 | 2004 | Change | % change | % annual change |
|---|---|---|---|---|---|---|
| Private | 152 | 1781 | 889 | -892 | -50.10 | -0.88 |
| Public | 13 | 1204 | 1208 | 4 | 0.33 | <0.01 |
| **All** | **165** | **2985** | **2097** | **-888** | **-29.75** | **-0.52** |

**Table 5.2.** Changes in the area (ha) of remnant gum woodlands and paddock trees between 1947 and 2004, across the range of the RtBC (N = no. of 1 x 1 km samples). Mixed = both paddock trees and remnant woodland in sample.

| Habitat Type | N | 1947 | 2004 | Change | % change | % annual change |
|---|---|---|---|---|---|---|
| Remnant | 13 | 1151 | 1169 | -18 | -1.56 | -0.03 |
| Mixed | 11 | 500 | 644 | -144 | -28.80 | -0.51 |
| Paddock | 141 | 1334 | 2060 | -726 | -54.42 | -0.95 |

**Table 5.3.** Changes in the area (ha) of gum woodland across the range of the RtBC between 1947 and 2004, summarised by land use type (N = no. of 1 x 1 km samples).

| Land Use Type | N | 1947 | 2004 | Change | % change | % annual change |
|---|---|---|---|---|---|---|
| Conservation | 13 | 1204 | 1208 | 4 | <0.01 | <0.01 |
| Cropping | 38 | 222 | 113 | -109 | -49.10 | -0.86 |
| Grazing | 92 | 968 | 585 | -383 | -39.57 | -0.69 |
| Pivot | 11 | 47 | 22 | -25 | -53.19 | -0.93 |
| Plantation | 11 | 544 | 169 | -375 | -68.93 | -1.21 |

The vast majority of gum eucalypt tree cover observed in sample areas was located in samples within the GHCMA region. Rates of gum eucalypt tree cover change in this region were the lowest of the three regions, with 21% decrease in tree cover evident in the sampled areas (Table 5.4). In the WCMA region, rates of change were slightly higher, at 26%. However the greatest proportional loss of tree cover was in the SENRM region, with 60% of gum tree cover which was evident in 1947 gone by 2004 (Table 5.4).

**Table 5.4.** Changes in the area of gum eucalypt tree cover across the range of the RtBC between 1947 and 2004, summarised by NRM region. (N = no. of 1 x 1 km samples).

| Region | N | 1947 | 2004 | Change | % change | % annual change |
|---|---|---|---|---|---|---|
| WCMA | 54 | 356 | 263 | -93 | -26.12 | -0.46 |
| GHCMA | 52 | 2025 | 1590 | -435 | -21.48 | -0.38 |
| SENRM | 59 | 604 | 244 | -360 | -59.60 | -1.05 |

## 5.2.4 Discussion

Losses of gum woodlands and paddock trees across the study area since 1947 were substantial, with 30% of all gum tree cover in the sample areas lost over 57 years. Both gum eucalypts and Buloke occur on the more fertile soils and have been preferentially cleared. Nevertheless, this figure is not as high as that representing clearance of stringybark tree cover over the same period (see Section 4.3). However, areas dominated by gum eucalypts also suffered from substantial clearing before 1947, so the extent of remnant gum woodland in 2004 covers only a few percent of its original distribution within the study area.

Losses in the sampled areas were all from private land, with the area of gum eucalypt tree cover on public land showing a small increase. Areas of paddock trees were more than thirty times as likely to be lost since 1947 as areas of denser woodland. These paddock tree losses are likely to have been of much larger trees with consequent higher probabilities of containing suitable nesting hollows for the RtBC (see Section 5.1). As described in Section 3.2 above, these landscape changes from a diverse variegated landscape to a fragmented landscape have consequences for biodiversity beyond the evident reduction in nesting resource availability for RtBCs.

Tree hollows suitable for RtBC nesting are currently not thought to be limiting the population. Nest success for the south-eastern RtBC is similar to that of other RtBC subspecies, and the rate of re-nesting in previously used hollows is low. Furthermore, populations of other species in the area with similar requirements for nest hollows, such as the Yellow-tailed Black-Cockatoo *Calyptorhynchus funereus*, are increasing in the presence of increased food supply (*Pinus radiata*). Although tree cover of live gums decreased substantially since 1947, the availability of hollows in dead trees may have increased due to ringbarking which was common practice for reducing tree cover in the early part of the 20th century. Without factoring in the contribution of these dead trees, it is difficult to conclude the effect of a reduction in gum eucalypt trees and woodlands over the 57-year period on RtBC nesting availability. However, it seems evident that a reduction in hollow availability has occurred given the substantial amount of loss of tree cover.

The loss of almost a third of the tree cover attributed to paddock trees in the 57 years to 2004 raises concerns about trends in nest hollow availability. However, losses of food resources over the same period have been greater than losses of nesting resources. Although at this time, there is more evidence that food availability limits the RtBC population rather than availability of nest hollows, if the trend in loss of nesting resources increases in severity, it may overtake food availability as a limiting factor in the future.

## *5.3 RECENT CHANGES IN GUM AND BOX EUCALYPT DISTRIBUTION*

### 5.3.1 Introduction

Although losses of gum eucalypt woodland and paddock trees have been substantial when measured over the 57 years to 2004, substantial changes in vegetation clearing legislation in both states mean that recent trends in the extent of gum eucalypt tree cover are likely to differ. In order to estimate future changes in nest hollow availability for RtBCs, it is necessary to gain an idea of the current rates of loss of tree cover. This section considers rates of change of paddock tree cover and woodland patches in the study area between 1992 and 2004 (GHCMA) and 1997 and 2004 (WCMA and SENRM).

### 5.3.2 Methods

Methods for determining change in availability of gum woodlands are as described in Section 2.

### 5.3.3 Results

The rates of change of tree cover on private land remained fairly high between 1992/97–2004, with annual rates of tree cover loss estimated at 0.34%. This was in stark contrast to publicly-owned land, which showed a small increase of 0.01% per annum. However, as the majority of gum eucalypt tree cover occurs on private land, the overall annual rate of loss for both land tenures combined was 0.32% (Table 5.5). Tree cover losses during this period were due entirely to losses of scattered paddock trees. Three percent of paddock tree cover was lost in the 7-12 year period.

**Table 5.5.** Changes in the area of Gum eucalypt tree cover on public and private land between 1992/1997 and 2004, across the range of the RtBC (N = no. of 1 x 1 km samples).

| Tenure | N | 1992/1997 | 2004 | change | % change | % annual change |
|---|---|---|---|---|---|---|
| Private | 176 | 1015 | 995 | -20 | -1.97 | -0.34 |
| Public | 13 | 1206 | 1208 | 2 | 0.17 | 0.01 |
| All | 189 | 2221 | 2203 | -18 | -0.81 | -0.32 |

**Table 5.6.** Changes in the area of Gum eucalypt tree cover attributed to paddock trees and remnant woodland between 1992/1997, across the range of the RtBC (N = no. of 1 x 1 km samples). Mixed = both remnant woodland and paddock trees in sample. Annual change represents weighted average across regions.

| Habitat Type | N | 1992/1997 | 2004 | change | % change | % annual change |
|---|---|---|---|---|---|---|
| Remnant | 13 | 1128 | 1132 | 5 | 0.44 | 0.04 |
| Mixed | 11 | 356 | 356 | 0 | 0 | 0 |
| Paddock Trees | 165 | 737 | 714 | -23 | -3.12 | -0.43 |

Although by far the greatest proportion of tree loss between 1947-2004 as from areas now dominated by plantation forestry, no tree cover was lost in those areas between 1992/97 and

2004. Instead, the only land use from which a notable proportion of tree cover was lost was centre pivot irrigation, with 27% of tree cover lost from those areas over the 7-12 year period. As almost all centre pivot irrigation is practiced in the WCMA and SENRM regions, these were the regions where the majority of tree loss occurred: nearly 4% over seven years in the WCMA and 2% over the same period in the SENRM region.

**Table 5.7.** Changes in the area of Gum woodlands on land dominated by different land use activities between 1947 and 2004, across the range of the RtBC (N = no. of 1 x 1 km samples). Annual change represents weighted average across regions.

| Land use type | N | 1992/1997 | 2004 | Change | % change | % annual change |
|---|---|---|---|---|---|---|
| Conservation | 9 | 819 | 819 | 0 | 0 | 0 |
| Cropping | 48 | 158 | 147 | -11 | -6.96 | -0.99 |
| Grazing | 102 | 636 | 637 | 1 | 0.16 | 0.05 |
| Pivot | 14 | 37 | 27 | -10 | -27.03 | -3.86 |
| Plantation | 12 | 184 | 184 | 0 | 0 | 0 |

**Table 5.4.** Changes in the area of Gum woodland across the range of the RtBC between 1992/97 and 2004, summarised by NRM region. (N = no. of 1 x 1 km samples). Annual change represents weighted average across regions.

| NRM Region | N | 1992/1997 | 2004 | Change | %change | % annual change |
|---|---|---|---|---|---|---|
| WCMA | 64 | 308 | 296 | -12 | -3.90 | -0.56 |
| GHCMA | 52 | 1589 | 1588 | 1 | 0.06 | <0.01 |
| SENRM | 73 | 324 | 317 | -7 | -2.16 | -0.31 |

## 5.3.4 Discussion

Annual rates of change calculated over a period of 7 (WCMA and SENRM) or 12 (GHCMA) years remained high, but were contributed to most by different land uses and different regions than changes in tree cover over the 57 year period to 2004. The losses over this period were concentrated in the SENRM and WCMA regions, both areas undergoing considerable agricultural intensification. Tree cover losses from grazing and non-irrigated cropping land were minor in the sample regions over this period, but significant losses from intensive cropping landscapes are suggestive that clearing to allow agricultural intensification is an important factor contributing to the loss of gum eucalypts. The land uses from which gum eucalypt tree cover was lost over the 7–12 year period to 2004 contrasted with those that were important over the 41-year period, as no tree cover in the sampled areas was lost from plantation forestry areas. The clearing for plantation forestry had therefore basically ceased in the sample areas by 1997.

Dead trees may be expected to become rare in the landscape over the next few decades, as ringbarking practices effectively ceased during the 20th century with the introduction of more effective methods of tree clearing. These dead trees are being lost from the landscape, and hollows from trees which die of causes other than ringbarking or which form in live trees will be the source of future nest hollows for the RtBC. Therefore, examining the dynamics of existing live gums can give an insight into the future availability of RtBC nest hollows. These results suggest that there may be a future bottleneck in availability of large gums, which are more likely to contain large hollows. In particular, available nest hollows will become less available in agricultural land and more concentrated in patches of remnant woodland.

# 6. MODELLING OF FUTURE SCENARIOS

## *6.1 BULOKE DISTRIBUTION AND CLIMATE CHANGE*

### 6.1.1 Introduction

**The climatic range of Buloke**

Buloke occurs in a wide range of climatic situations from northern Queensland to south east South Australia. It occurs in areas that experience a mean annual temperature from 14ºC to 25ºC (minimum temperature to 0.9ºC; maximum temperature to 36ºC). Annual precipitation within its range varies from 350mm to 2100mm. Figure 6.1 illustrates the distribution of Buloke in Australia and the climatic extremes of its distribution.

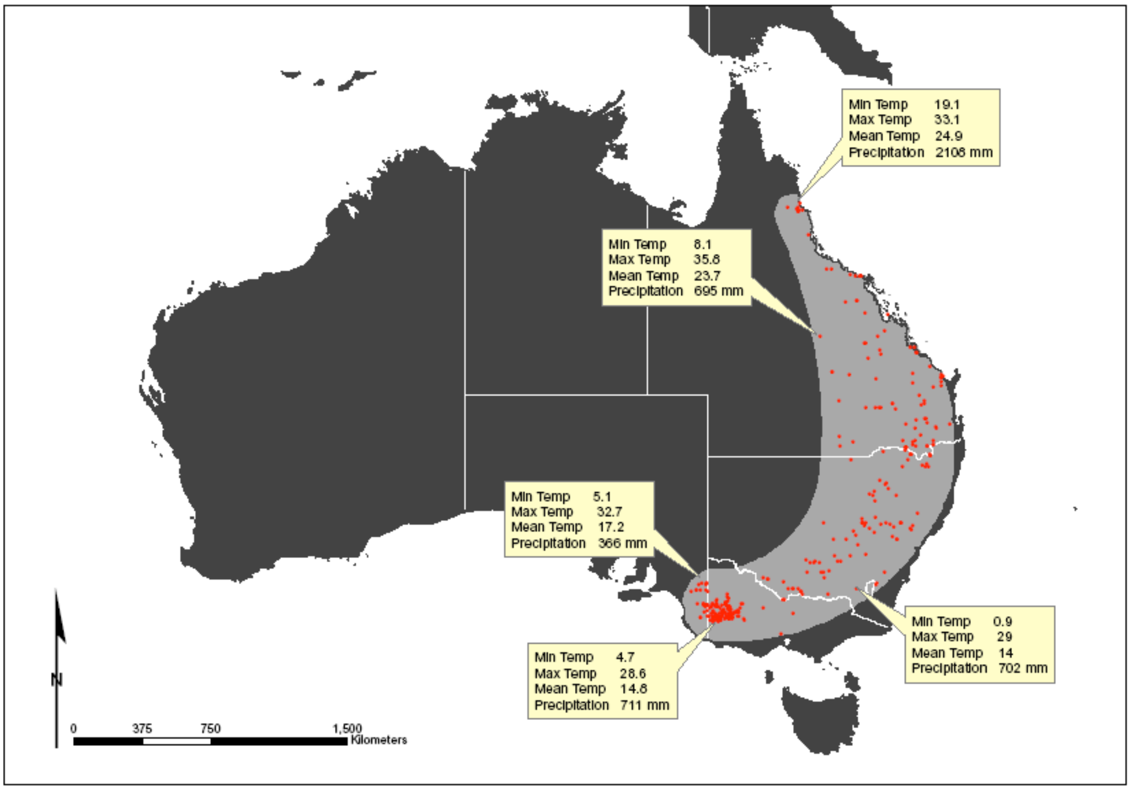



**Figure 6.1.** Distribution of Buloke *Allocasuarina luehmannii.*

**Climate within the RtBC range**

Within the RtBC's range, Buloke is predominantly located in the Wimmera IBRA subregion. Typically, the Wimmera region has warm to hot summers with average maximum temperatures of approximately 27ºC to 30ºC. In winter, average maximum temperatures are between 13ºC and 15ºC and frosts are common. Annual rainfall across the region averages 490mm. Regional rainfall is subject to significant inter-annual variability.

**General considerations**

Climate change is recognized as a major threat to biodiversity worldwide. Changing conditions may force species to move over large distances within a relatively short period of time to follow suitable climate. Although it is possible to model expected climatic changes over a region, it is very difficult to predict the actual response of plants (and plant-dependent animals) to these climatic changes as they are expected to respond to multiple variables. While species that occur over a wide variety of climates tend not to be vulnerable to climate change-induced extinction, their range may alter. Populations at the extremes of their current climatic range are likely to be more sensitive to changing climate.

Figure 6.1 outlines some climatic gradients in the area occupied by Bulokes. Figure 6.2 suggests that an increase in temperature is unlikely to have a significant effect on the distribution of Buloke within the RtBC's range, as Buloke tolerates much higher temperatures elsewhere in its current distribution. In contrast, a decrease in precipitation is likely to make the Wimmera region less suitable for Bulokes, as the area has the lowest rainfall in its current geographical range.

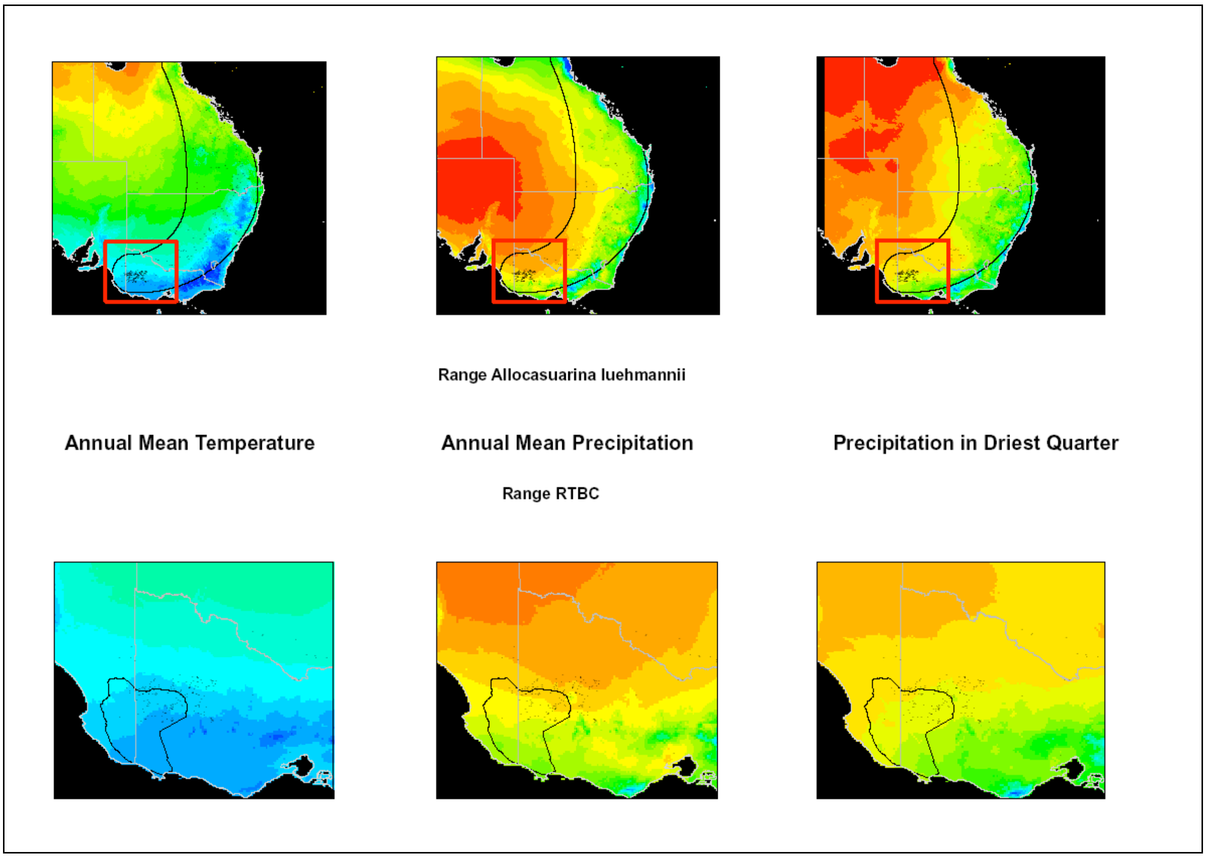



**Figure 6.2** Climate gradients in eastern Australia.

### Climate change in the wimmera

CSIRO predicts that the future climate in the Wimmera will be warmer and drier. Annual temperatures are expected to increase by 0.2ºC – 1.4ºC by 2030 and 0.7 – 4.3ºC by 2070. Predictions of changes in precipitation are less precise; however, an annual decrease in precipitation is likely (changes of +3% to -15% by 2030 and +10% to -40% by 2070) in all seasons and extreme heavy rainfall events may become more intense. Table 6.1 outlines expected seasonal changes.

**Table 6.1.** Projected changes in climate for the Wimmera bioregion. Source: CSIRO (Atmospheric Research)

| | 2030 | 2070 |
|---|---|---|
| **Spring** | warmer by 0.3 to 1.6º<br>drier by 0 to 20% | warmer by 0.8 to 5.0ºC<br>drier by 5 to 60% |
| **Summer** | warmer by 0.3 to 1.6ºC<br>precipitation change of +10 to -15% | warmer by 0.8 to 5.0ºC<br>precipitation change of +20 to -40% |
| **Autumn** | warmer by 0.2 to 1.4ºC<br>precipitation decrease likely (+3 to -10%) | warmer by 0.7 to 4.3ºC<br>precipitation decrease likely (+10 to -25%) |
| **Winter** | warmer by 0.2 to 1.4ºC<br>precipitation decrease likely (+3 to -10%) | warmer by 0.7 to 4.3º<br>precipitation decrease likely (+10 to -25% |

**Modelling species response to climate change**

Whilst great effort has been made to understand the potential impact of climate change on the distribution of species, the models currently used are rough and deliver questionable results. In general, two methods (or a combination of both) are used to determine effects of changing climate on species distribution:

1. Mechanistic models, in which species distribution is modelled based on the species' physiology.
2. Bioclimatic envelope models, in which the current species distribution is used to create an climatic envelope

Whilst we acknowledge that it would be ideal to utilize both methods in the project we currently do not have sufficient knowledge of the physiology of Buloke to apply mechanistic models. As a result our model uses species location data from state herbariums and BIOCLIM Data combined with *DIVA*-GIS software.

## 6.1.2 Methods

To determine the current climatic envelope of Buloke we obtained geo-referenced points of known occurrence of the species from the state herbariums of South Australia, Victoria, Canberra, New South Wales and Queensland.

Using BIOCLIM data and DIVA-GIS software we predicted the potential geographic distribution of Buloke using the climate characteristics of its current distribution. For the first prediction we used two climate variables (annual mean temperature and annual mean precipitation) For the second we used six variables (annual mean temperature, mean temperature of the coldest quarter, mean temperature of the warmest quarter, annual mean precipitation, precipitation of driest quarter and precipitation of wettest quarter) The more variables used in a model, the smaller the climatic envelope of a species (Figure 6.3).

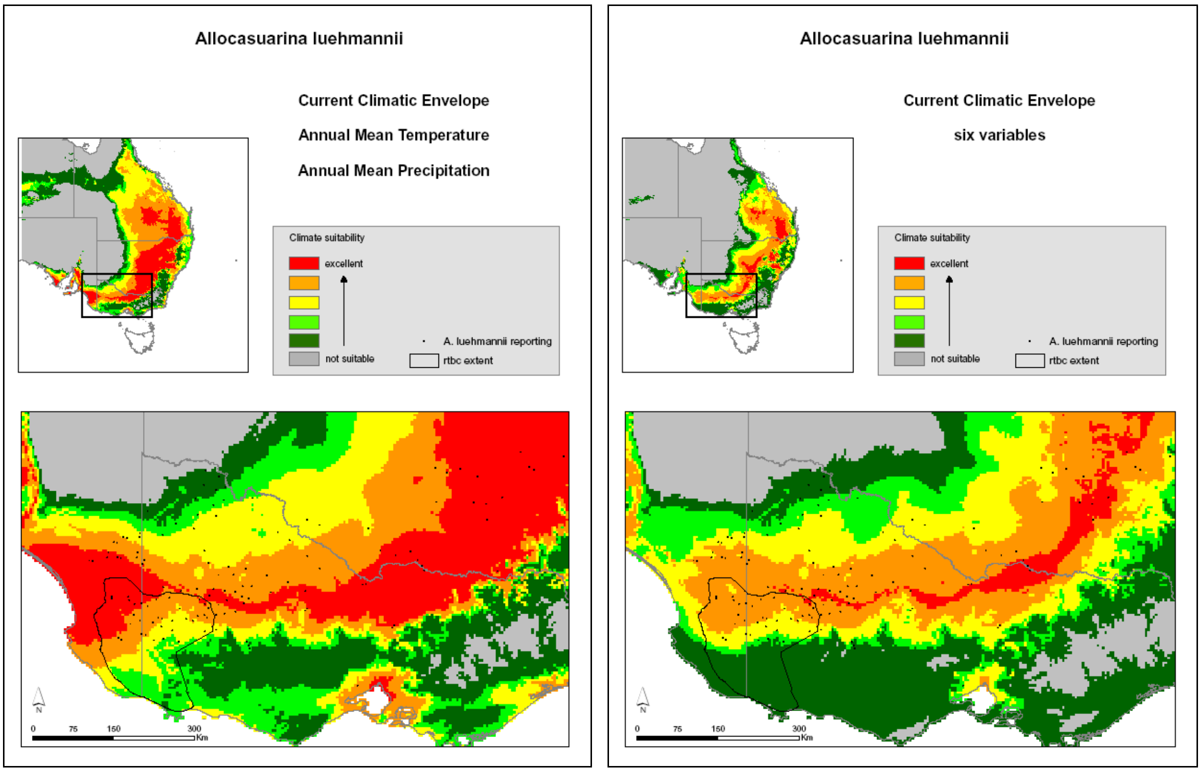



**Figure 6.3.** The predicted climate envelope of Buloke using two and six climate variables.

A reasonable expectation for climate change would be 3ºC warming by 2070 for the Australian continent. Unlike temperature, predicted changes in rainfall vary considerably across the current Buloke distribution. As the main objective of this project is to model future scenarios of this species as food source for the RtBC, we used rainfall predictions for the Wimmera in our scenarios. We created two scenarios, one with a decrease of 10% annual rainfall and one with a decrease of 20% annual rainfall, both at a temperature increase of 3ºC. Figure 6.4 illustrates the shift of the current climatic envelope in these scenarios.

### 6.1.3 Results

Under the scenarios we investigated, a general southwards shift of the current climatic envelope of Buloke in the RtBC range area is expected. While an increase in temperature alone would not have a negative effect on Buloke distribution within the RtBC range, a decrease in rainfall is likely to be a limiting factor. With a minor decrease of rainfall (-10%) suitable climate should still be found over the entire RtBC range. However, with a decrease of 20% rainfall, the northern parts of the current RtBC range will be faced with a far less suitable climate.

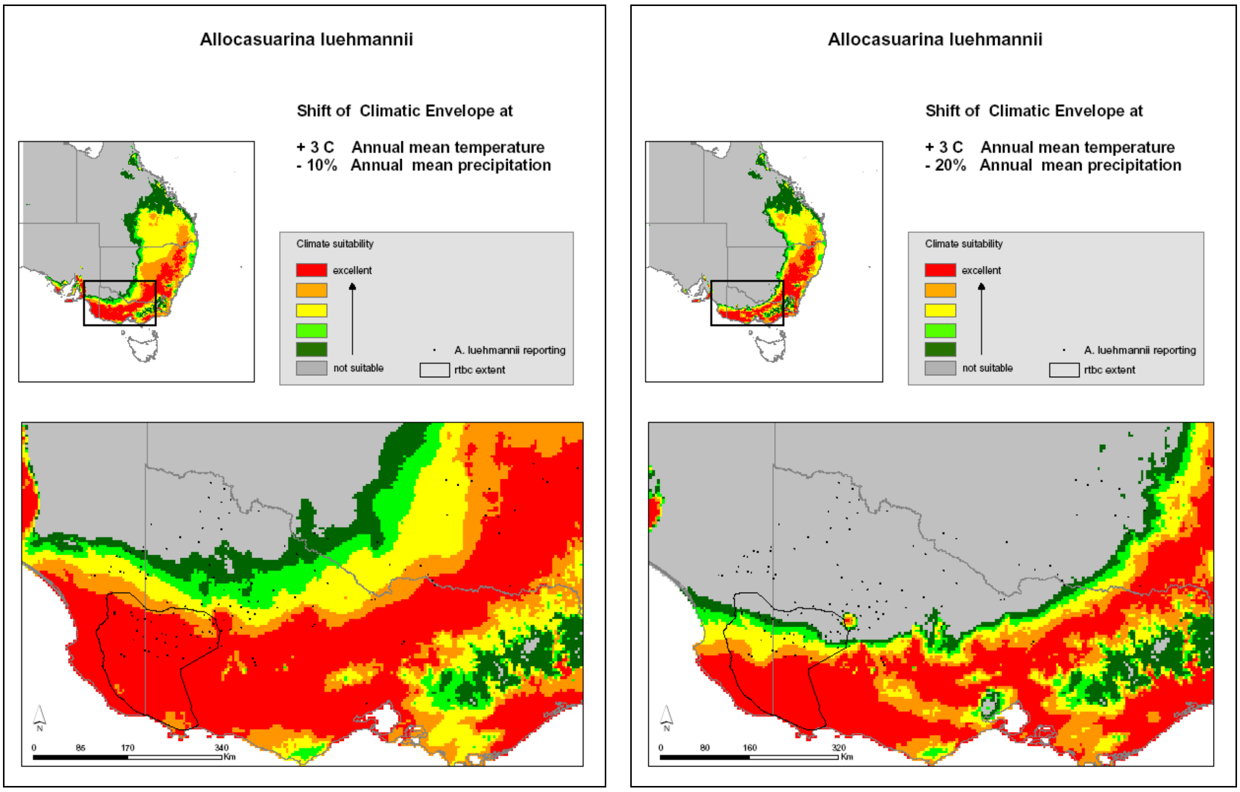



**Figure 6.4.** Future scenarios of suitable climatic conditions for Buloke within the RtBC range.

### 6.1.4 Discussion

These predictions are based on climatic boundaries derived from present day distributions and these boundaries might be different from the climatic boundaries in which Buloke physiologically could survive (the fundamental climatic envelope). Therefore these results should not be used to predict the future distribution of Buloke. Instead the results provide a **first approximation** of the potential impacts of climate change on this species and can provide some management guidance as long as the limitations of the model are considered. While a more sophisticated approach to modelling the impact of climate change on Buloke (researching physiology of the species and taking biotic interactions and dispersal abilities etc. into account) would be ideal it was not practicable for this project.

**Limitations of climate envelope modelling**

The most significant limitation of using the climatic envelope modelling we have applied is that it assumes species are unable to survive in conditions which are different to those they experience in their current distribution. The climate envelope model used here only takes the actual distribution of species in account and not the distribution in which they may physiologically be able to survive.

There are also many other factors determining species distribution under changing climate, which are not factored into the model. These include:

- Biotic interactions: How species interact competitively or in a predator-prey association
- Evolutionary changes: How species (or populations) adapt genetically to a changing environment
- Species dispersal: How “mobile” a species is; how fast a species can migrate and which dispersal mechanisms are used
- Increased CO2 levels: What effects the increased atmospheric CO2 level might have on plant growth

The results of the model we have used **do not** attempt to predict the shift of the Buloke distribution due to changing climate, but predicts the shift of its current climatic envelope due to change in temperature and precipitation. However, there is evidence to suggest that climate change will reduce the climatic suitability of areas where Buloke currently occurs within the RtBC range. A precautionary response to this evidence would include a greater focus on revegetation efforts towards the more southern and western of the areas in the RtBC range which currently support Buloke (See Appendix 4 for suggested priority planting areas). Such areas may be more likely to continue to have suitable climatic conditions than areas closer to Horsham and north of the southern boundary of the Little Desert.

## 6.2 FUTURE SCENARIOS OF BULOKE AVAILABILITY

### 6.2.1 Introduction

Should Buloke be able to survive in the Wimmera under future climate changes, numerous additional factors will influence the future availability of Buloke resources for the RtBC. This part of the project synthesises information gathered about the current state of Buloke resources, recent rates of change in availability of these resources and future management options to demonstrate a suite of alternative future scenarios of Buloke availability. By extrapolating recent rates of change to Buloke resource availability, a 'status quo' scenario can be generated to project the likely impact of a 'business as usual' approach to management of Bulokes. There are several variables over which land managers and policy makers may have some control, and by manipulating these variables in a series of scenarios we can identify which are likely to have the greatest impact, positive or negative, on future Buloke availability.

### 6.2.2 Methods

Buloke availability was modelled as the number of mature, live Buloke trees greater than 30 cm in diameter. Trees of this age were assumed to be 100 years old. This assumption is conservative as available research suggests that a tree of 19cm DBH is approximately 100 years (L. Macauley, pers. comm.). A conceptual model relating various factors to future mature Buloke availability was developed (Fig 6.5). This was used as a basis for developing a dynamic mathematical model which allowed calculation of the number of available Buloke trees over a period of several hundred years, given a set of parameter values.

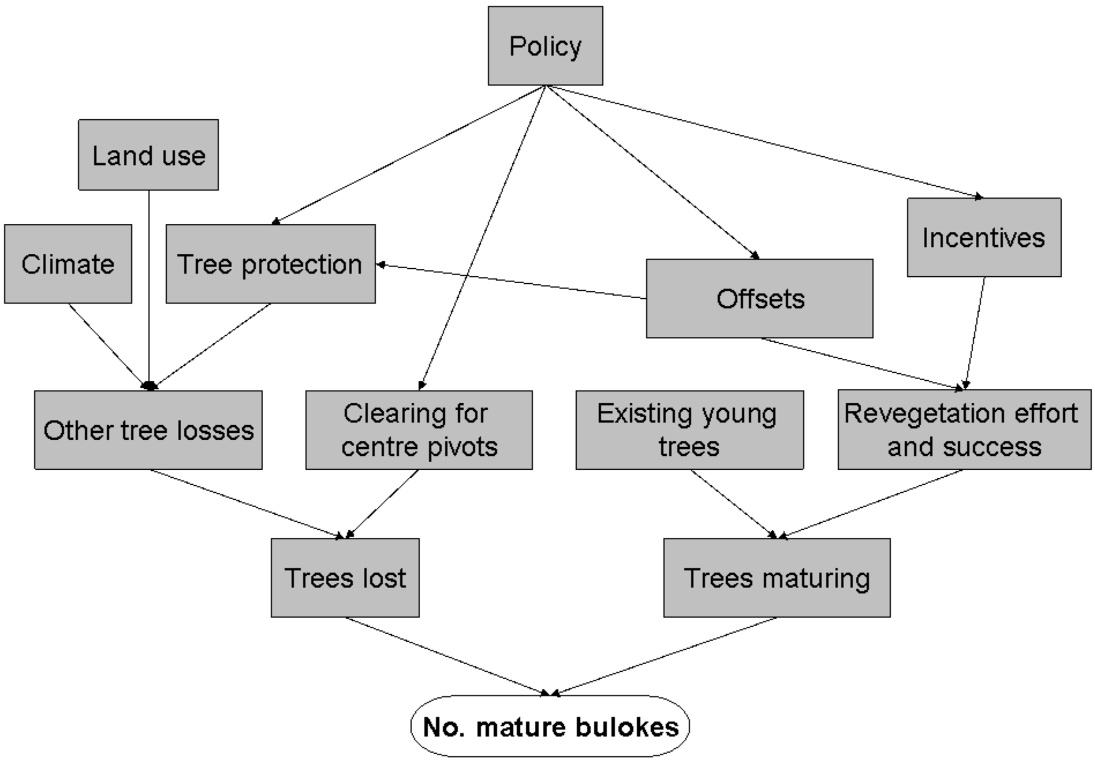



**Figure 6.5.** Conceptual model of Buloke availability

The model changes Buloke availability as a function of tree losses and tree gains. Losses were due to the application of landscape-specific death rates as well as direct losses due to clearing for centre-pivot irrigation. Gains were due to the maturation of trees from <100 years old to ≥ 100 years old. These maturing trees could be trees that were already present in the starting scenario but were not yet 30 cm DBH, or trees which were planted during the modelled period 100 years earlier.

An assumption was made that death rates were modified by climate: in dry years, death rates of trees in all land uses increased by 20% and in wet years death rates decreased by 20%. Although the true impact of annual precipitation on tree death rates is not known, these values represented a realistic scenario. The proportion of dry years could be manipulated. Death rates could also be modified by moving mature scattered trees in extensive cropping or grazing landscapes into a 'protected' landscape, as might occur under an offset scenario. The change in death rates for trees which become 'protected' in this way is not known, so several scenarios were explored with different reductions in death rates. Losses due to clearing for centre pivots could be increased or decreased and death rates of trees in extensive cropping and grazing landscapes could be altered.

Gains were due to trees being planted as part of offsets or voluntarily. Trees were input as the number of trees planted that were expected to survive for 100 years. It was assumed that planted trees entered the ‘reserve’ landscape—i.e., they would not be cleared or suffer death rates associated with cropping or grazing.

Increases in tree numbers were limited so that the number of trees could not exceed the number likely to be supported by the available land area. For example, although roadsides contain over 7,000 young Buloke trees per hectare, only a tiny proportion of these could be expected to reach maturity. Maximum densities of mature trees for most land uses were set at 84 trees per hectare, the maximum density of large trees recorded during the fieldwork component of this project (see Section 2.1). However, the maximum density of trees on roadsides was set somewhat higher, at 140 mature trees per hectare, as the reduced competition due to their linear shape is likely to allow a greater stem density.

Starting conditions for all scenarios were set as the 2004 distribution and availability of Buloke trees, patches and roadsides within the RtBC range as described in Section 2. The data on Buloke availability and rates of change from South Australia and Victoria were combined for the Buloke modelling as the rates of change were similar and relatively small number of Buloke trees were recorded for South Australia. The proportion of dry years for most scenarios was set to 0.5.

The ‘status quo’ scenario (**Scenario 1**) included the following inputs and assumptions:

- rates of change for extensive cropping, grazing, patches/reserves and roadsides were the annual rates of change calculated for the period 1997-2004;
- 160 trees were lost to centre pivots each year for the first 20 years, and offsets included 10 trees planted for each tree removed. 2000 trees were planted in other or voluntary revegetation per year for 20 years. Survival at 100 years for planted trees was set at 10%;
- It was assumed that Bulokes occurring on roadsides were not limited in their growth by competition (until mature trees reached maximum density) and would grow at the same rate as planted trees;
- Trees in mapped ‘patches’ of Buloke were assumed to be already at maximum mature tree density as these were defined as areas with a closed canopy, thus with

limited opportunity for additional revegetation. More open areas of Buloke woodland were mapped as individual scattered trees.

A series of amendments to the ‘status quo’ scenario were modelled.
**Extra revegetation.** This scenario included a 3 x greater rate of voluntary Buloke revegetation, plus a doubling of survival rates of planted trees such as might occur with increased effort and investment in planting and maintenance techniques.

**Additional offset requirements.** These scenarios represented the effects of adding the requirement that for each tree removed under permit for a centre pivot, a certain number of other mature scattered trees would be protected. Scenarios investigated were for four and eight trees protected per tree removed. As the change in death rate as a result of this protection is unknown, a series of alternative outcomes was modelled, including a 25%, 50% and 75% reduction in death rates for protected trees compared to the death rates to which they were previously subject in either grazing or cropping landscapes. The proportion of protected trees to come from grazing and cropping landscapes was set at 50:50.

**Low death rate, no clearing for centre pivots.** In this scenario, no trees were cleared for centre pivots, and the death rates of trees in cropping and grazing were halved, representing a possible scenario in which almost all deliberate and incidental (e.g. through stubble fires) clearing ceased.

**No clearing for centre pivots.** These scenarios were based on a cessation of clearing Buloke trees for centre pivots (although clearing of other types is factored into the landscape-specific death rates). These scenarios also included additional revegetation efforts as described above.

**Status quo in dry conditions.** These scenarios were identical to scenarios 1 and 2 but simulated an increase in dry years from 50% to 70%.

**Offsets as patches.** Clearing for centre pivots continues as before in these scenarios, but offsets for clearing include only revegetation and protection of Buloke trees in patches.

**Increased dry years.** These scenarios demonstrated the potential impacts of increasing the number of dry years to 70% on the above scenarios.

Finally, we investigated the required revegetation effort to restore Buloke availability to 2004 and 1963 levels for three representative scenarios: **Status quo**, **Additional offset (8 trees) moderate death rate**, **and low death rate**. The number of trees which would need to be planted per year for 20 years, then per year thereafter, assuming a 10% survival to maturity, was calculated. 1963 numbers were based on estimated rates of change as described in Section 3 and an assumption that age class distribution and stem density in different occurrence types were similar to that in 2004.

### 6.2.3 Results

In all scenarios, Buloke availability declined (to a greater or lesser degree) for 100 years before increasing due to revegetation maturing. The example scenarios which included replanting of Bulokes either voluntarily or as an offset simulated replanting over a 20-year period. Therefore, 120 years from time 0, there was a second peak in Buloke availability. Thus, there are two points in the time series of particular interest: 100 years (the point at which the resource bottleneck is most severe), and 120 years (future peak). The summary of resource availability in the different scenarios shows results for 100, 150 and 200 years (Table 6.2). Ongoing revegetation effort was not factored in to these scenarios and so food availability generally began to decline again after 120 years.

The model demonstrates future scenarios only and does not show them in the context of past change. This context is important in order to appreciate the magnitude of future changes in resource availability relative to past availability. Figure 6.6 below demonstrates Scenario ‘Status quo extra revegetation” in the context of change since 1960.

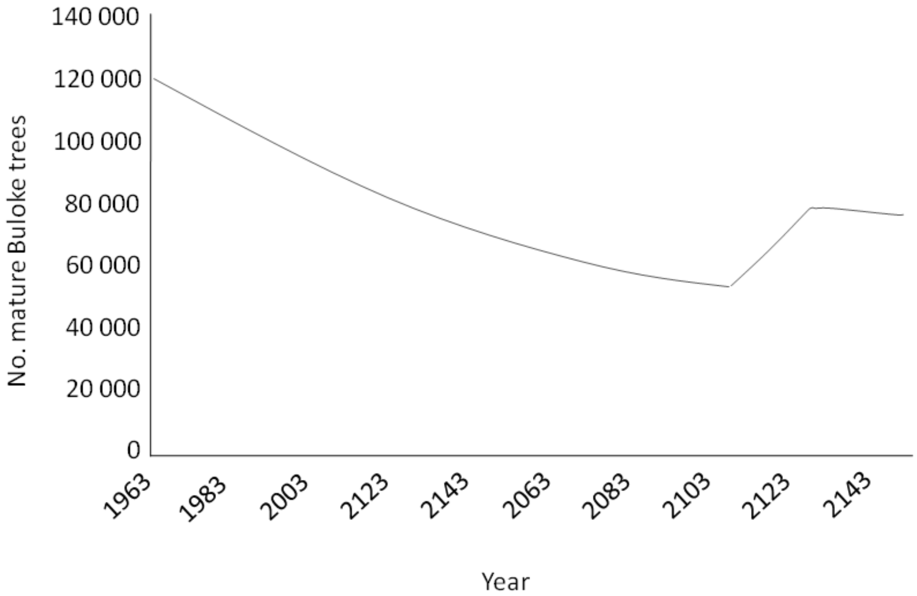



**Figure 6.6.** Future scenario 'Status quo extra revegetation' in the context of historical change in Buloke availability.

**Status quo and Extra revegetation**

Under the assumptions of the status quo scenario, the number of available Bulokes in the recent comparison study area declined from a starting level of 87,000 to a minimum bottleneck of 56,000, 41% of the number estimated to be available in 1962. After that point, maturing revegetation increased tree numbers slightly, after which they gradually began to decline again. However, with added effort and success in revegetation, the trees increased again to a peak of 84,000. In all scenarios pictured, the dark line at the top represents total mature trees, while the lines below represent changes over time to numbers of mature trees in different landscape categories (steady lines are trees on roadsides and reserves; declining lines are trees in cropped and grazed paddocks; line which increased after 100 years is revegetated trees).

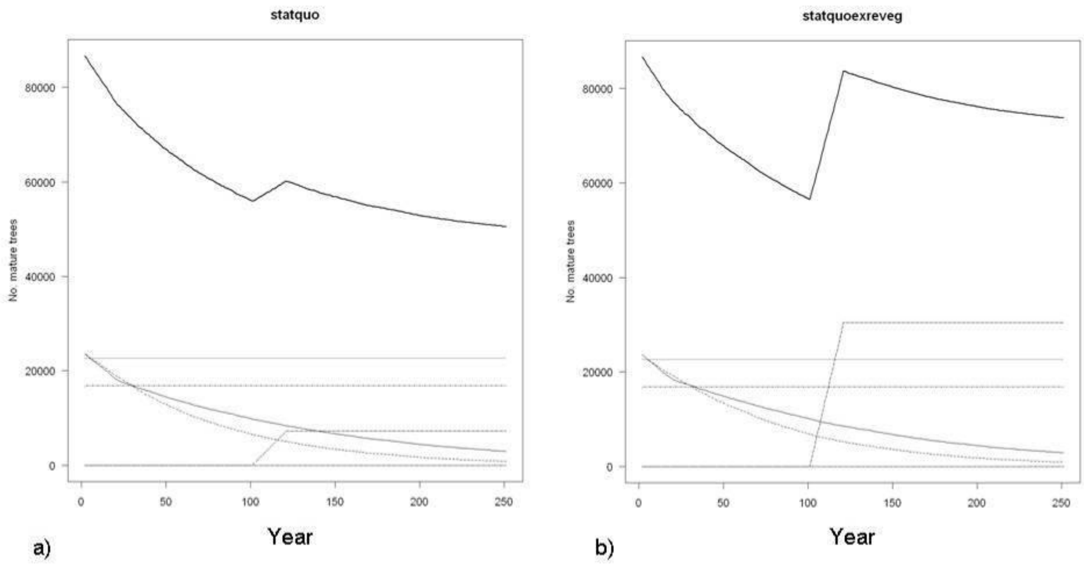



**Figure 6.7.** Buloke availability through time (bold line) in a) the status quo scenario and b) status quo with increased revegetation effort. Lines below represent contributions to the total from (top to bottom from right hand side of graph): trees in reserves, trees on roadsides, revegetated trees, trees in grazed paddocks, trees in extensively cropped paddocks, and trees in centre pivot paddocks (always zero).

**Additional offset requirements**

By introducing offset requirements to the subset of trees which are cleared for centre pivot irrigation, the severity of the 100-year bottleneck was reduced. Offsets for clearing which require four mature scattered Buloke trees to be protected for each tree cleared reduced the bottleneck to 57,000-62,000 trees at its narrowest point, depending on the reduction in death rates achieved by the protection. By increasing the offset requirement to eight trees protected per tree removed, the severity of the bottleneck further reduced to 59,000 - 68,000 trees, again depending on the effect of protection on death rates.

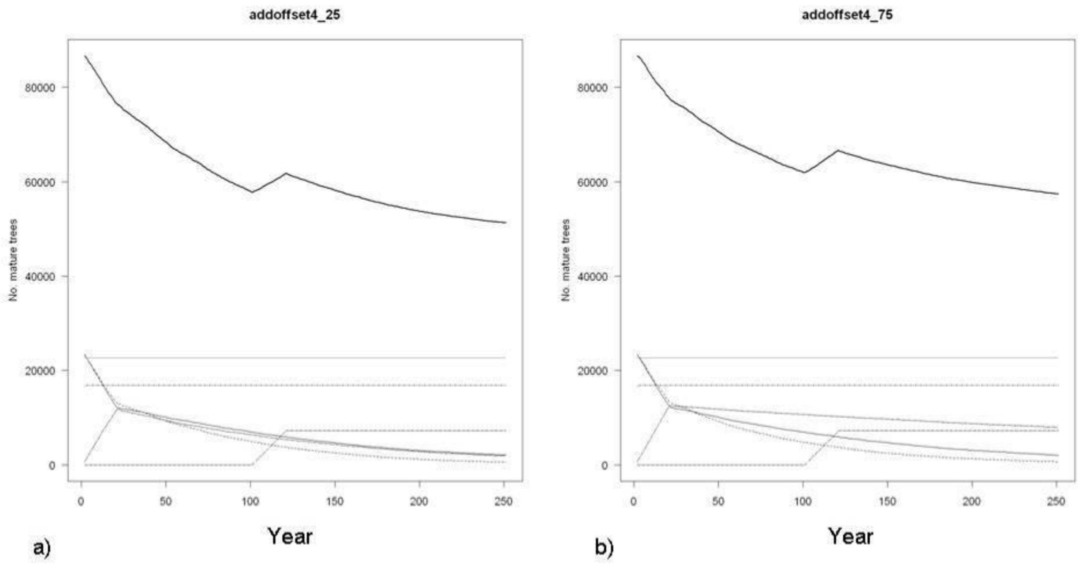



**Figure 6.8.** Future scenarios of Buloke availability assuming additional offset requirements of 4 mature scattered trees protected per tree cleared for centre pivots. Scenarios show reduction in death rates of protected trees of a) 25% and b) 75%. The line which increases sharply for the first 20 years represents trees entering the ‘protected’ landscape as they are protected as offsets.

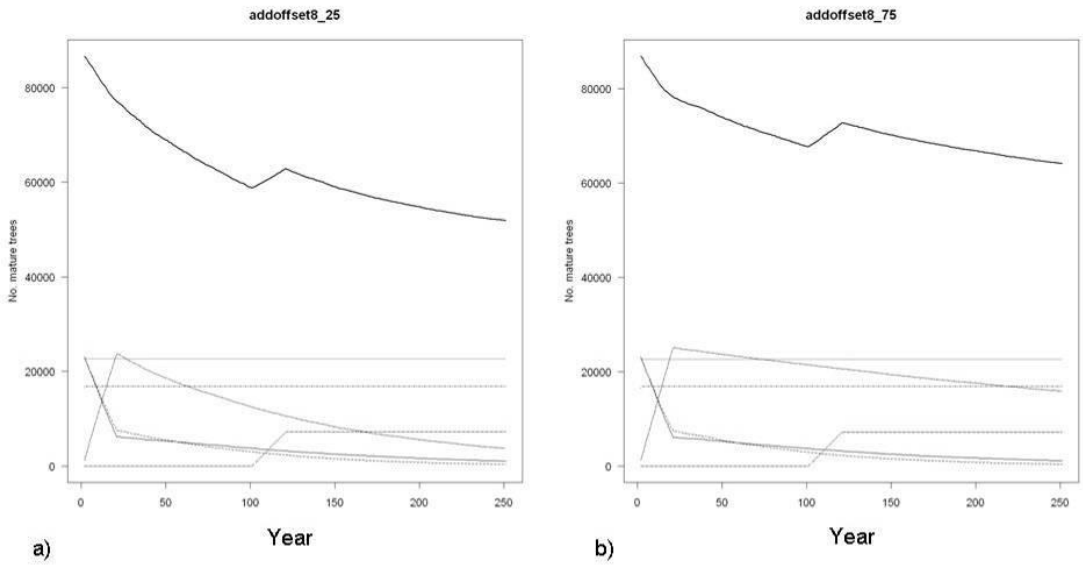



**Figure 6.8.** Future scenarios of Buloke availability assuming additional offset requirements of 8 mature scattered trees protected per tree cleared for centre pivots. Scenarios show reduction in death rate of protected trees of a) 25% and b) 75%.

**Increased dry years.** An increased incidence of drought modified the scenarios, but only slightly. For example, the increased offset 4 low death rate scenario in a drier climate differed from the same scenario in normal conditions by reducing resources by less than 1% (Table 6.2).

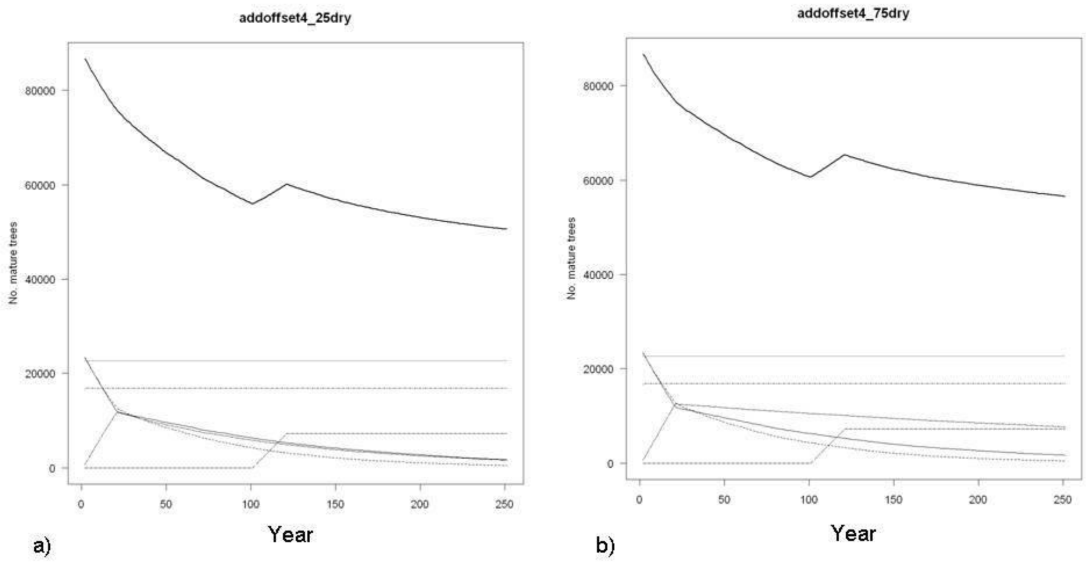



**Figure 6.9.** Future scenarios of Buloke availability with increased incidence of dry years and assuming additional offset requirements of 4 mature scattered trees protected per tree cleared for centre pivots at reductions of 25% and 75% in death rates of protected trees.

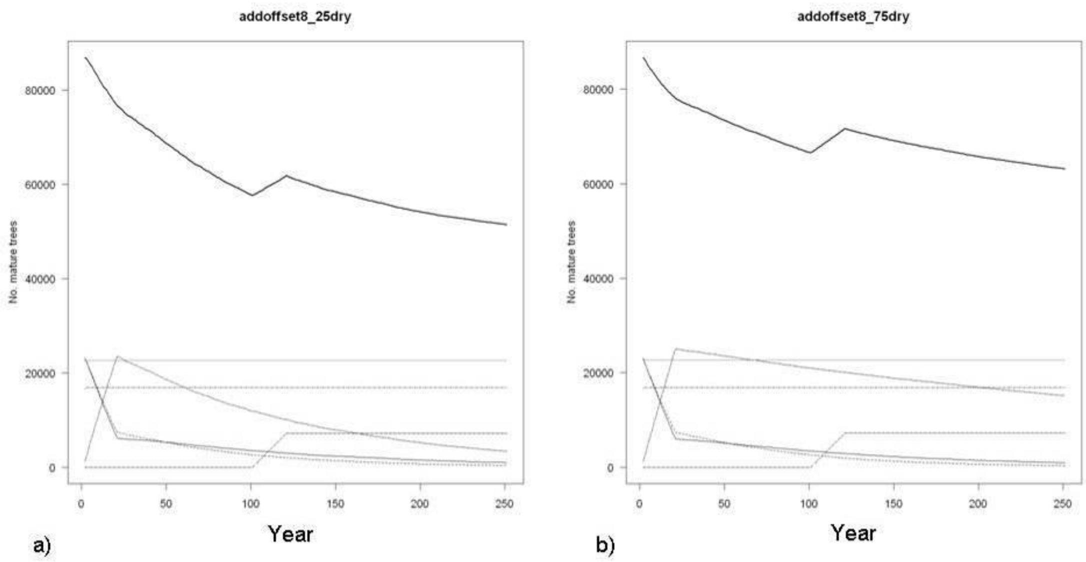



**Figure 6.10.** Future scenarios of Buloke availability with increased incidence of dry years and assuming additional offset requirements of 8 mature scattered trees protected per tree cleared for centre pivots. Scenarios show reduction in death rate of protected trees of a) 25%; and b) 75%.

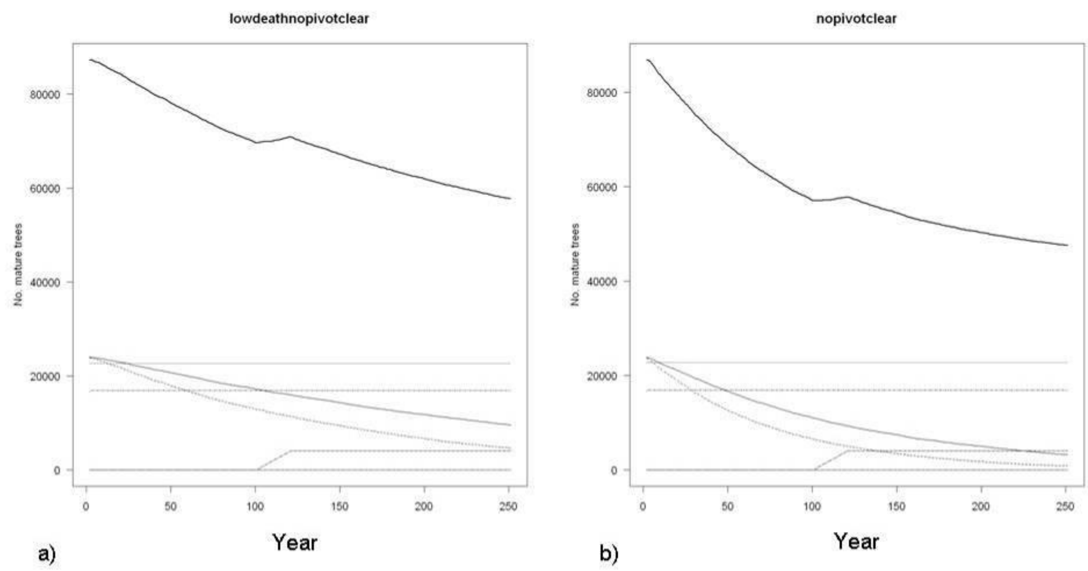



**Figure 6.11.** Scenarios including reduced tree death rates due to cessation of a) deliberate and accidental clearing and b) centre pivot clearing.

**Low death rate and No clearing for centre pivots**

By avoiding deliberate clearing for centre pivots and simulating no other action on tree loss due to other factors, the bottleneck in resource availability declined to 66% of current levels in 100 years. Apart from clearing due to centre pivot installation, the amount of paddock tree loss due to avoidable factors was not able to be quantified. However, a reduction in death rates of 50% as might be expected with increased focus on tree protection was modelled. The results showed resource availability decline to 81% of current levels in 100 years before increasing again.

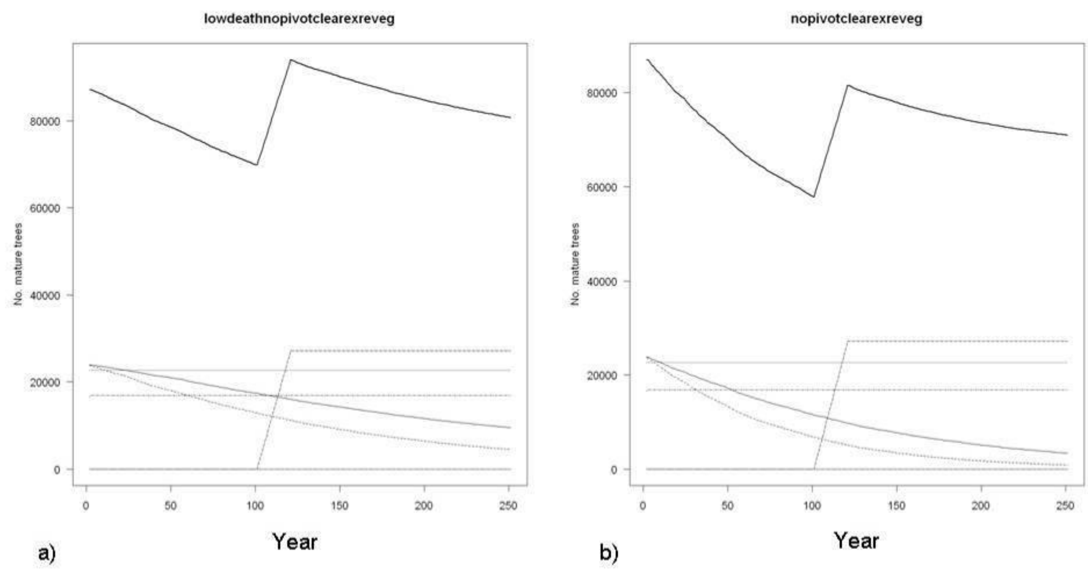



**Figure 6.12.** Scenarios including reduced tree death rates due to cessation of a) deliberate and accidental clearing and b) centre pivot clearing. Includes increased revegetation efforts.

**Scenarios in dry conditions.**

The status quo scenarios were only slightly modified by increasing the percentage of dry years to 70% with differences of only 1% predicted at 100 years (Figure 6.13).

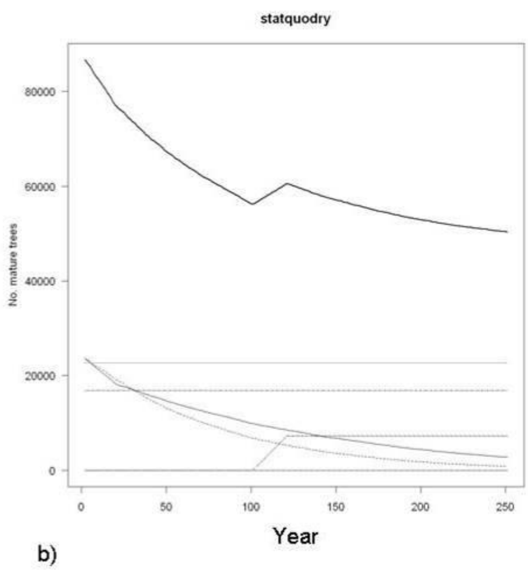



**Figure 6.13** Status quo scenario of Buloke availability under more frequent dry conditions.

**Offsets as patches**

In this scenario, as patches recorded no change in extent between 1997-2004, the scenario predicted the same changes in resource availability as for status quo.

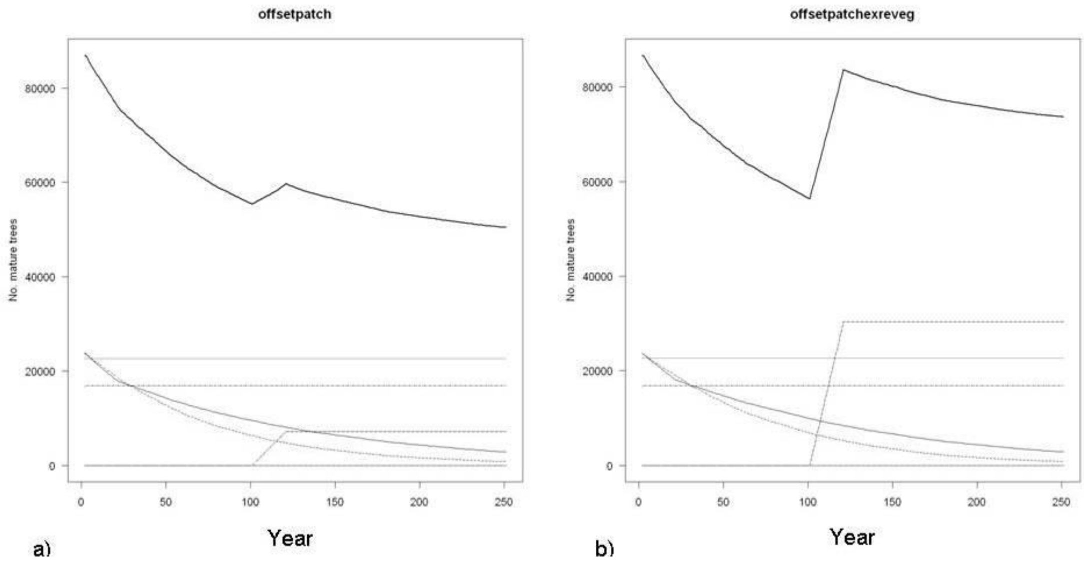



**Figure 6.14.** Scenarios depicting continued clearing for centre pivots with patches of Buloke used as offsets for trees cleared, with a) current and b) increased revegetation effort.

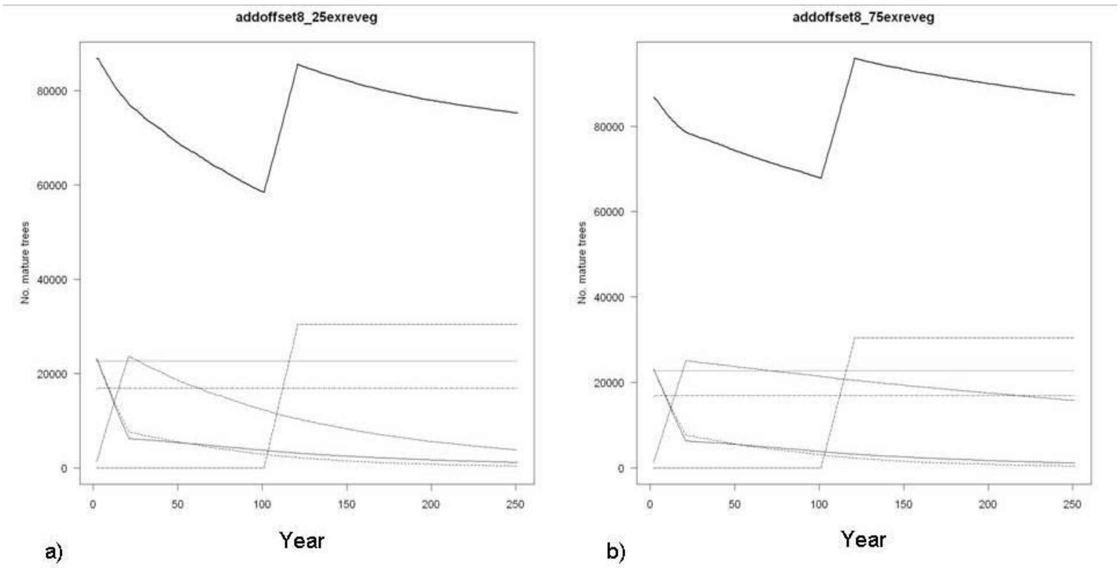



**Figure 6.15.** Future scenarios of Buloke availability assuming additional offset requirements of 4 mature scattered trees protected per tree cleared for centre pivots with increased revegetation effort. Scenarios show reduction in death rate of protected trees of a) 25% and b) 75%.

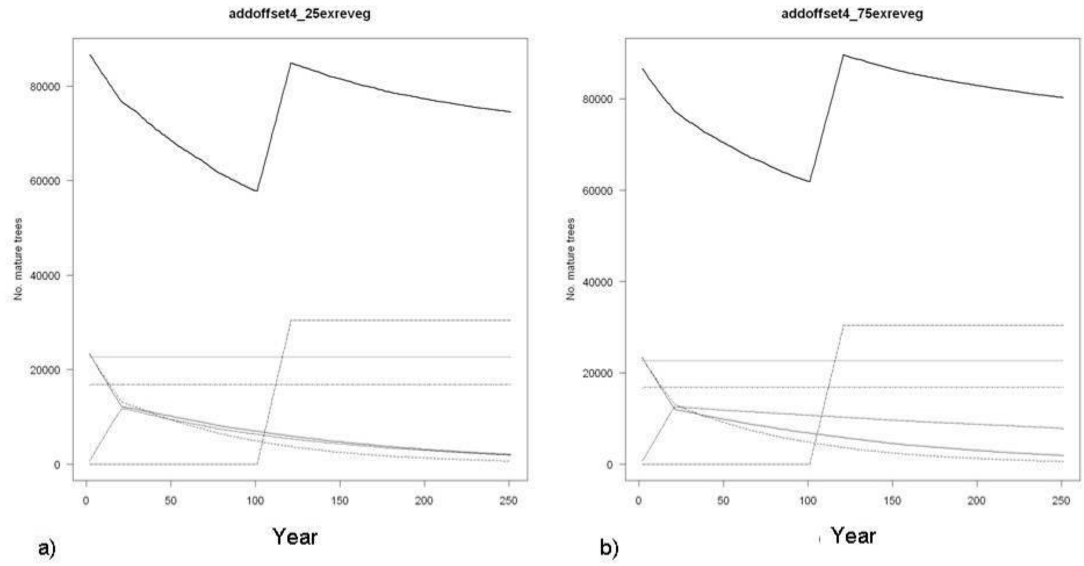



**Figure 6.16.** Future scenarios of Buloke availability assuming additional offset requirements of 8 mature scattered trees protected per tree cleared for centre pivots with increased revegetation effort. Scenarios show reduction in death rate of protected trees of a) 25% and b) 75%.

Most scenarios had their narrowest resource bottleneck at approximately 100 years, although those which include little revegetation suffered a decline below this bottleneck level approximately 150 years into the scenario. The severity of this 100 year bottleneck varied substantially among scenarios. Table 6.2 demonstrates the estimated number of mature Buloke trees at 100, 150 and 200 years as a percentage of 2004 tree numbers for each of the scenarios.

**Table 6.2.** Estimated percentage of mature Buloke trees remaining after 100 years (the narrowest bottleneck given adequate revegetation), 150 years and 200 years under selected scenarios.

| Scenario no. | Scenario name | % mature trees remaining | | |
|---|---|---|---|---|
| | | 100 years | 150 years | 200 years |
| 1 | Status quo | 65 | 65 | 61 |
| 2 | Offset 4 high death rate | 66 | 67 | 62 |
| 3 | Offset 4 moderate death rate | 68 | 70 | 75 |
| 4 | Offset 4 low death rate | 71 | 73 | 69 |
| 5 | Offset 8 high death rate | 68 | 68 | 63 |
| 6 | Offset 8 moderate death rate | 74 | 76 | 71 |
| 7 | Offset 8 low death rate | 78 | 81 | 77 |
| 8 | Low death rate no pivot clearing | 80 | 77 | 71 |
| 9 | No pivot clearing | 66 | 63 | 58 |
| 10 | Patches as offsets | 65 | 65 | 61 |
| 11 | Status quo extra reveg | 65 | 92 | 87 |
| 12 | Offset 4 high death rate extra reveg | 66 | 93 | 88 |
| 13 | Offset 4 moderate death rate extra reveg | 69 | 97 | 92 |
| 14 | Offset 4 low death rate extra reveg | 71 | 100 | 92 |
| 15 | Offset 8 high death rate extra reveg | 67 | 94 | 89 |
| 16 | Offset 8 moderate death rate extra reveg | 74 | 102 | 97 |
| 17 | Offset 8 low death rate extra reveg | 78 | 108 | 104 |
| 18 | Low death rate no pivot clearing extra reveg | 81 | 104 | 98 |
| 19 | No pivot clearing extra reveg | 66 | 89 | 84 |
| 20 | Patches as offsets extra reveg | 64 | 92 | 88 |
| 21 | Status quo increased dry years | 65 | 65 | 61 |

**Revegetation requirements**

In all the scenarios considered for required revegetation rates, more than 10,000 Buloke trees per year for 20 years needed to be planted to restore Buloke availability to 2004 levels. The worst case scenario considered was the status quo, which required 18,000 trees to be planted per year for 20 years (assuming 10% survival to maturity), and the best was the scenario representing no clearing of trees for pivots and reduced death rates for other paddock trees, which only required 11,000 to be planted per year for 20 years. At 84 stems/hectare at maturity, recovery from the status quo scenario would require at least 21 hectares to be planted out per year. In all cases, approximately 1,000 new Buloke trees would need to be planted per year after 20 years to maintain numbers at the new peak. To recover trees to 1963 levels after 120 years, an additional 462,460 Bulokes would need to be planted, or about 23,100 per year. Assuming recovery to 1963 levels and a status quo scenario for rates of loss, this would equate to 41,100 trees planted over at least 49 ha per year for 20 years. Improving long-term survival rates would reduce the number of trees required to be revegetated, but not the area over which they would need to be planted.

**Table 6.3.** Estimated levels of revegetation required to restore 2004 levels of Buloke availability at 12 years and maintain that level indefinitely. Assumes 10% survival of seedlings to maturity.

| Scenario | No. new Bulokes required | |
|---|---|---|
| | Per year for 20 years | Per year after 20 years |
| Status quo | 18 000 | 1 000 |
| Offset 8 moderate death rate | 13 000 | 900 |
| Low death rate | 11 000 | 1 000 |

### 6.2.4 Discussion

It is important to note that, in these scenarios and those developed for the other resource types, the model presents a simplified version of reality and contains many assumptions. Therefore, the projections of the model are not predictions, but projections of likely futures under the assumptions of the models. However, they provide a useful tool for comparing potential outcomes of changes to the status quo.

Under the assumption that dry years increased tree death rates by 20%, the scenarios with changed climate had little influence on resource availability. However, there is considerable uncertainty around this estimate, and further investigation of the potential effects of changed climate on tree survival is required before discounting a potentially significant future impact.

Protection of paddock trees potentially had a substantial effect on resource availability, with a reduction in death rates of 50% coupled with a cessation of clearing among the best scenarios for resource availability. Importantly, the reduction in death rates achieved by protecting paddock trees as offsets to clearing was a particularly influential parameter, with differences in resource availability between the high death rate and low death rate scenarios as great as 11% at 100 years. Therefore, should protection of mature scattered trees as offsets for clearing be considered, the benefits of such an approach will depend largely on whether substantial reductions in death rates can be achieved.

The status quo scenario was the worst for Buloke resource availability, with only 65% of current resource levels remaining at 100 years and subsequent increases due to maturing revegetation being moderate. This outcome was also predicted for a situation where continued clearing of paddock trees was permitted with increased offsets in the form of Buloke trees in patches. As patches were not projected to be lost through time, this form of offset had no effect on reducing the severity of the resource bottleneck. Other approaches to offsets, such as those not requiring like-for-like protection, also would not improve on the status quo scenario.

Revegetation was critical for recovery in resource levels after 100 years, as most existing young trees occurred in areas with an already high density of mature trees, and so did not mature to add to resource availability during the scenarios. Estimated current levels and success rates of revegetation as modelled in the status quo scenarios are inadequate to recover Buloke resources to current levels. However, the increased revegetation scenarios improved the outlook considerably. Although in the scenarios considered, revegetation was undertaken for a 20 year period, ongoing revegetation at some level would be required to stabilise resource levels in the long-term. The required levels of revegetation assume a 10% survival rate to 100 years, which can be considered at the highest end of best case scenarios. A large proportion of planted Bulokes dies within the first few years (A. Bradey, pers. comm.) and uncertainty in wildfire risk and land ownership mean that the real rate of survival to 100 years might be closer to 2-5% (A. Bradey, pers. comm.). Should better estimates of long-term survival of Buloke seedlings be developed, the minimum revegetation efforts suggested by these scenarios must be updated appropriately. The greatest challenge to achieving substantial gains through Buloke revegetation is likely to be obtaining adequate suitable land for planting. Land most suitable for Bulokes is among the highest value agricultural land.

## *6.3 STRINGYBARK SEED AVAILABILITY*

### 6.3.1 Introduction

The range of factors influencing stringybark seed availability are profoundly different to those affecting Buloke seed availability. While the vast majority of Buloke woodlands occur on private land and have been cleared extensively for agriculture, stringybark woodlands are relatively abundant, with around 50% of their pre-European extent remaining and large areas being protected on public land. Stringybarks also regenerate and mature more rapidly than Bulokes(trees less than 15 years of age may be used for foraging; Koch, personal observation), so the age structure of stringybarks is less important than for Bulokes.

Importantly, the factor which most influences foraging site selection for stringybarks is the density of food items per tree (measured in terms of crop size, the number of seed capsules per tree and capsule density, the mean number of seed capsules per branch, Koch 2003). By contrast, the most important factor influencing foraging site selection for Bulokes is fruit profitability (seed weight per cone, cone size etc., Maron and Lill 2004). The importance of food density at the level of the tree (and probably the level of the "patch" of trees) has profound implications because it suggests that factors that are likely to influence the quantity of seed capsules per tree for stringybarks (eg. fire, grazing, no. of paddock trees etc.) are likely to be just as important for overall food availability as the total amount of habitat *per se*. This has been demonstrated for the effect of prescribed burns on capsule availability and foraging site selection, with dramatically lower rates of habitat use in more recently burnt areas (Koch 2003). Any model of stringybark food availability thus needs to take into account the quantity of seed capsules per tree (crop size) as well as the total number of stringybarks in the landscape.

Finally, the RtBC tends to feed on one species of stringybark or the other depending in which has fruited more recently (both species produce a large seed crop approximately once every 3 years but they often fruit in different years, Koch 2003). This means that only half of the total stringybark habitat is effectively available during most years, except that the ratio of *E. arenacea* to *E. baxteri* woodland area in the range of the RtBC is 28:72, not 50:50. The cockatoos thus have much less feeding habitat available to them in years when they are feeding exclusively in *E. arenacea*, and it is highly likely that food supplies limit breeding success in such years (Koch 2003). The ratio of *E. arenacea*: *E. baxteri* is thus an additional important factor that needs to be incorporated into consideration of stringybark seed availability.

### 6.3.2 Methods

Due to the A conceptual model relating stringybark seed availability to a series of factors, including tree death and recruitment, fire and surrounding land use was developed (Figure 6.17). Using this as a guide, a dynamic mathematical model was constructed. Stringybark seed availability was modelled as the number of stringybark “tree equivalents”; that is, the average food value of a stringybark tree of either species in a continuous block of ungrazed forest. Stringybarks in blocks of grazed woodland and scattered paddock trees contributed more than one tree equivalent per tree due to their greater food value (see Section 4). The multipliers applied to trees in these landscapes are shown in table 6.4 below. Annual fluctuations in fruit availability in different parts of the RtBC range (except for those due to fire) or between the two stringybark species are not dealt with explicitly in this model; rather, the modelled resource availability should be interpreted as average availability under the scenario conditions.

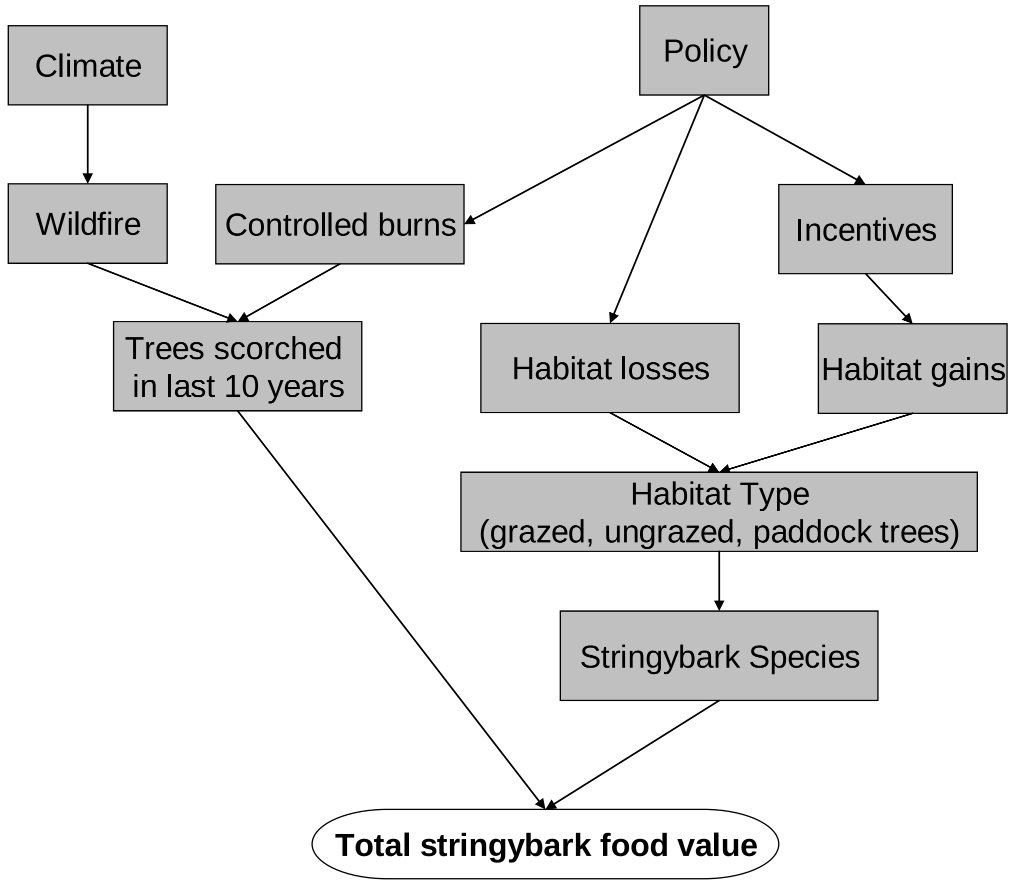



**Figure 6.17.** Conceptual map of factors influencing stringybark food availability.

**Table 6.4.** Food value multipliers applied to each stringybark species in different land uses.

| Species | Landscape | Multiplier |
|---|---|---|
| *E. arenacea* | Grazed forest | x1 |
| | Paddock tree | x27 |
| *E. baxteri* | Grazed forest | x3 |
| | Paddock tree | x12 |

A further factor which affected the contribution of a single tree to the number of tree equivalents was whether the tree had been subject to a fire resulting in canopy scorch within the previous ten years. For these trees, a multiplier of 0.49 was applied, which is the average reduction in food value of trees during the ten years following canopy scorch (Koch 2003).

Trees could be scorched by either wildfire or prescribed burns. For wildfires, 100% of the trees affected were penalised to simulate the typical canopy scorch due to wildfire. For trees affected by prescribed burning, the proportion of trees affected could be varied to simulate different approaches.

Finally, the number of trees in each land use changed as a function of landscape- and species-specific death rates and replanting rates. Numbers of trees used were from the ~2% of the study area sampled in Section 4. Absolute numbers should be multiplied by a factor of 45.45 to extrapolate to the entire study area.

The status quo scenario projected current rates of change and values for area burnt and canopy scorched over 200 years. Subsequent scenarios included the following:

**Fire 9 and Fire 63:**

These scenarios simulated an increase in wildfire extent by either 9% or 63% of the current extent (area burnt by wildfire in the last 10 years). These scenarios are based on the modelling work of Hennessy et al. (2005), which used the latest climate change models to comprehensively examine the influence of climate change on fire weather patterns. They found that the number of days when the Forest Fire Danger Index (FFDI) rating is very high or extreme could increase from its current average of 9.0 days per year to 9.8–11.1 days (9–23%) by 2020 and 10.8–14.7 days (20–63%) by 2050. The FFDI is a useful indicator of fire risk based on a combination of weather variables including: daily temperature, precipitation, relative humidity and wind-speed. The modelling work presented in the current study assumes that a percentage increase in FFDI will produce an equivalent increase in extent of area burnt by wildfire. This is likely to be a conservative estimate of increase in wildfire extent because it is likely that climate change will produce record high temperatures and associated fire conditions that may produce catastrophic wildfires which are very difficult to suppress. A single catastrophic fire event has the potential to burn the vast majority of RtBC feeding habitat because much of the stringybark habitat exists as large contiguous forest blocks.

**Scorch 30:**

This scenario simulated the impact on stringybark seed resources if all prescribed burning was conducted with 30% canopy scorch, rather than the 80% in the status quo model. The figure of 80% is an estimate based on observations by R. Hill and P. Koch that needs to be verified in the field. A reduction in canopy scorch to 30% has been demonstrated to be realistic through field trials of cool burns in the Horsham fire district (where canopy scorch due to prescribed burns tends to be complete due to the low canopy heights in the region and the high levels of suspended fine fuels, Koch pers. obsv.) involving pre and post-burn measurements of capsule density (Rudolph 2004).

**No paddock trees:**

This scenario simulated the loss of all scattered paddock trees to identify the sensitivity of the model to this component of habitat.

**Reveg 10 000 paddock (mixed) and Reveg 10 000 as ungrazed forest:**

These scenarios demonstrated the effect on the food resource of replanting or recruiting 10,000 extra stringybark trees (50:50 mix of species) as either scattered (low-density) trees or in the form of a block of ungrazed forest to compare the two strategies. The figure of 10,000 is added to the figures from the ~2% of the study area which was sampled. To obtain an equivalent figure for the entire study area, the 10,000 should be multiplied by a factor of 45.45.

**Reveg 10 000 baxteri and Reveg 10 000 arenacea:**

In this scenario, the effect of planting only *E. arenacea* or only *E. baxteri*, as low-density paddock trees, were compared. Again, the absolute numbers should be considered relative to the 2% of the study area sampled, from which figures for the modelling were taken.

**Reveg 100 ha of arenacea forest:**

This scenario simulated the revegetation of 100 hectares of stringybark forest (ungrazed) with *E. arenacea* only. This additional area is added to the total area of stringybark within the 2% of the study area sampled.

**Catastrophic wildfire:**

Estimating wildfire risk is a difficult task and is complicated by the unpredictable nature of wildfires. Although estimates of mean changes in extent of wildfires can be made, a single catastrophic fire could have substantial impacts on stringybark food availability for a considerable period. This scenario simulated a wildfire extending over 85% of stringybark forests in the RtBC range.

**Multiplier uncertainty**

As the large multipliers used for paddock trees and trees in grazed areas are likely to have substantial influence in the model, the above scenarios were all run with 30% lower multipliers to test this sensitivity.

### 6.3.3 Results

The rates of change used in the scenarios were very minor, and the scenarios therefore changed little through time. In all scenarios, gradual losses of paddock trees resulted in a slow decline in resource availability over time. Therefore, they are not presented graphically but are summarised below in tabular form as the percentage of currently available food resources at two time points: scenario start time (changes applied immediately) and after 150 years (Table 6.5).

The scenarios Fire 9 and Fire 63 reduced stringybark resource availability by 1% and 5%, respectively. On the other hand, reducing canopy scorch in prescribed burns led to a projected 3% increase in resources. The most severe negative effect, however, was elicited by simulating catastrophic wildfire with 85% of stringybark blocks burnt, which produced an overall reduction in stringybark seed resources of 49%. Simulating the loss of all paddock trees also produced a severe reduction in stringybark food availability of 11%.

The importance of paddock trees was further illustrated by comparing scenarios that simulated the replacement through revegetation or restoration of 10 000 trees, either as paddock trees or as forest blocks (within the 2% of the study area being considered). A 50:50

mix of *E. arenacea* and *E. baxteri* replanted as scattered trees would result in a 10% increase in available stringybark resources, but the same number of trees regenerated as ungrazed forest would achieve less than 1% increase. A greater gain in food availability is achieved by planting *E. arenacea* scattered trees compared with *E. baxteri*.

For all scenarios, reductions in paddock trees drove declines of about 2% after 150 years. Despite the relatively low numbers of paddock trees, the effect on overall resource availability was disproportionately large due to the high multipliers used to represent the greater food value of such paddock trees. Under the status quo, food availability is expected to be 98% of current availability in 150 years, and 96% in 250 years (Table 6.5).

**Table 6.5.** Results of stringybark scenario modelling.

| no. | Scenario | % of status quo<br>At scenario start | <br>After 150 years |
|---|---|---|---|
| 1 | status quo | 100 | 98 |
| 2 | fire 9 | 99 | 97 |
| 3 | fire 63 | 95 | 93 |
| 4 | scorch 30 | 103 | 101 |
| 5 | no paddock trees | 89 | 89 |
| 6 | reveg 10 000 paddock (mixed) | 110 | 106 |
| 7 | restore 10 000 as ungrazed forest | 100 | 98 |
| 8 | reveg 10 000 arenacea paddock | 114 | 109 |
| 9 | reveg 10 000 baxteri as paddock | 106 | 103 |
| 10 | restore 100 ha of arenacea forest | 101 | 98 |
| 11 | No prescribed burning | 105 | 102 |
| 12 | 50% paddock trees lost | 94 | 93 |
| 13 | Catastrophic wildfire | 51 | 49 |

Table 6.6 shows the results of the same scenarios run using 30% lower multipliers, to demonstrate the sensitivity of the model to different input data. Changes in the outputs were comparatively minor, with no change occurring for some scenarios.

**Table 6.6.** Results of stringybark scenario modelling using 30% lower multipliers.

| no. | Scenario | % of status quo (assuming lower multipliers)<br>At scenario start | <br>After 150 years |
|---|---|---|---|
| 14 | fire 9 | 99 | 97 |
| 15 | fire 63 | 95 | 93 |
| 16 | scorch 30 | 103 | 101 |
| 17 | no paddock trees | 91 | 91 |
| 18 | reveg 10 000 paddock (mixed) | 108 | 104 |
| 19 | restore 10 000 as ungrazed forest | 101 | 99 |
| 20 | reveg 10 000 arenacea paddock | 111 | 107 |
| 21 | reveg 10 000 baxteri as paddock | 105 | 102 |
| 22 | restore 100 ha of arenacea forest | 101 | 99 |
| 23 | No prescribed burning | 105 | 103 |
| 24 | 50% paddock trees lost | 96 | 95 |
| 25 | Catastrophic wildfire | 48 | 46 |

The model also provides some useful insights into the relative availability of *E. arenacea* and *E. baxteri*. Within the sample area, the estimated number of *E. arenacea* trees was 294,805, and of *E. baxteri* 1,273,170 (18% *E. arenacea*). However, when food value multipliers are applied, 30% of food availability is estimated to be from *E. arenacea*.

Assuming that 51% of original *E. arenacea* and 49% of *E. baxteri* remains, the multipliers suggest that although half of all stringybark trees have been lost, stringybark food availability has only declined by 31%. To recover this pre-settlement amount of food availability by planting *E. arenacea* as 'paddock trees', we would need to replace 1,640,568 over the entire range. Interestingly, were this to be achieved, food availability would then be very close to a 50:50 *E. baxteri/E. arenacea* split.

### 6.3.4 Discussion

Surprisingly, stringybark seed availability was not very sensitive to climate-change induced increases in wildfire extent. The scenarios fire 9 and fire 63 (which simulated climate change-associated increases in wildfire extent by 9% and 63% of the recent extent, respectively) reduced overall stringybark resource availability by 1% and 5%, respectively. These scenarios thus produced relatively minor changes in resource availability. However, it is important to remember that these scenarios are based on percentage increases above recent wildfire events. A scenario modelling a single catastrophic fire that burnt 85% of all stringybark blocks reduced total food availability by 49%. Although this is an unlikely scenario, it is possible, as much of the stringybark remnant vegetation occurs as large, contiguous areas that are highly prone to wildfire. A single catastrophic wildfire of the type witnessed in 1939 has the potential to produce a dramatic and sustained reduction in stringybark seed availability across the range of the RtBC.

Prescribed burns performed in strategic areas and wildfire suppression efforts are therefore an important habitat protection mechanism for the RtBC, but they need to be performed in such a way as to minimise direct impacts on food availability. A simulated reduction of canopy scorch levels to 30% canopy scorch in prescribed burn applications produced relatively minor increases in resource availability (3%). However, changes in levels of canopy scorch (or the total area burnt) are the only way to increase food supply to the cockatoos in the short term, and low scorch burns (with target reductions to 30% scorch) have been demonstrated to be a feasible and cost-effective alternative means of increasing capsule availability whilst meeting fuel reduction objectives (Rudolph 2004). Lower intensity fires also tend to produce small unburnt patches within the larger burn area (Angus pers. comm.) that are important refuges for reptiles, mammals and other ground-foraging animals (Tolhurst 1996, Wilson 1996).

The most effective management action in terms of increasing overall stringybark seed availability was found to be the replanting of 'paddock' trees, or trees at low density. An addition of 10000 trees in paddocks (or at low density) would produce an increase of 10% within the 2% of the study area sampled. The number of additional 'paddock' trees (both species mixed) required to achieve this 10% gain across the entire study area would be over 450,000. This large effect of paddock trees was because of the high food value multipliers which apply. Although scattered paddock trees make up a very small proportion of the total number of stringybark trees in the RtBC range, they appear to be particularly influential on total food availability due to their large seed crops. As the multiplier on *E. arenacea* is greater than that for *E. baxteri*, scenarios involving *E. arenacea* restoration showed a greater gain in food resources than those involving *E. baxteri*. *E. arenacea* is also the less well represented of the two stringybark species (ratio of *E. arenacea* to *E. baxteri* is 28:72; Koch 2003). Restoring a more even ratio has the potential to greatly increase food availability in years when the population depends solely on *E. arenacea* (discussed further in Section 6.3.1). It is important to realise that this is a long term solution, however, as trees may take 50 years or longer to reach peak levels of seed availability. On the other hand, growth rates of paddock trees may be substantially greater than those of trees in remnants due to lower levels of competition and greater light levels (leading to greater photosynthetic capacity).

## *6.4 NEST SITE AVAILABILITY*

### 6.4.1 Introduction

Although the availability of nest hollows is currently not thought to be a factor limiting RtBC population size, trends in nest hollow availability suggest that this resource is decreasing and may become a limiting factor in the future. Large old paddock trees are being lost from paddocks, from where most nests are known. Nevertheless, there are substantial areas of relatively young and mid-aged gums occurring on roadsides throughout the study area. This component aimed to develop a dynamic model to project current rates of loss of mature trees, maturation of young trees and revegetation efforts over the next 200 years to identify at what point the availability of large old gums is lowest. It also simulated several alternative scenarios in which changes in revegetation effort or rates of tree loss result in alternative outcomes.

### 6.4.2 Methods

The model used to investigate future scenarios of the availability of large old gums was similar to that used for Buloke availability. Both resource types need to be of considerable age before they are of use to the RtBC, and both differ substantially in their age class distributions and stem densities among different parts of the landscape. The influence of fire on gums in less important than for stringybark; indeed, fire may even assist in hollow development. Therefore, the dynamic mathematical model used to investigate future scenarios of large old gum was similar to that used for Bulokes.

As the availability of hollows suitable for nesting by the RtBC in the future is likely to be roughly proportional to the number of large old gum eucalypts >100 cm DBH in the landscape, this was the response variable modelled. Critically, the currently preferred nesting resource (hollows in dead eucalypts), which is likely to decline dramatically in availability over the next few decades, could not be considered in the modelling as estimates of current availability were not available. Therefore, changes in availability of large old gums were assumed to reflect changes in hollow availability. Factors which affected the availability of large old gums included landscape-specific death rates (determined in Section 5.3) and maturation of young gums. Gums >100 cm DBH were estimated to be at least 200 years of age (see Section 5.1). Once the density of large mature gums in woodland patches reached 10 per hectare, the landscape was considered to have reached capacity for such large trees. For trees in linear roadside strips, given the reduced competition expected, large old trees were allowed to increase to a maximum density of 15 per hectare.

Revegetation of gums across the sampled area (~2% of the study area) was estimated at 10,000 per year (and assumed to continue for 20 years). However, these revegetation sites are typically planted or seeded at very high stem density, and given the low density required for the development of large old trees, 1% of these trees (100) were assumed to survive past 200 years.

In addition to the status quo, the following scenarios were considered:

**Extra pivot clearing**:

In this scenario the number of paddock trees cleared within the sample areas for centre pivot irrigation per year was doubled from 25 to 50.

**Extra revegetation:**
The revegetation effort/success rate was doubled.

**Slow roadside growth:**
This simulated a decrease in growth rates of roadside trees compared to trees elsewhere of 50%.

**Offset 4:**
For each tree cleared for centre pivots, four other scattered paddock trees (half from cropping landscapes, half from grazing landscapes) were protected and their death rates halved. 50% of trees protected were in cropping landscapes.

### 6.4.3 Results

The typical trajectory of large gum availability in the scenarios was a gradual decline for 100 years as paddock trees were lost, followed by a plateau until about 150 years as young roadside gums matured, offsetting further paddock tree losses. After 150 years, a relatively large cohort of roadside gums matured, resulting in an increase in the number of mature trees, although in most scenarios, not to 2004 levels. At this point, however, the maximum density of mature trees on roadsides was attained, and so losses then continued to be driven by paddock tree loss until the 200 year mark, when revegetation began to mature (Figure 6.18).

Changes through time were not as pronounced as in the Buloke model, as the relatively large numbers of trees in patches and reserves buffered the losses of trees from cropping landscapes. Tree loses from grazing landscapes were partially compensated for by maturation of small numbers of younger paddock trees. All scenarios showed similar trajectories, although the Offset 4 and Extra revegetation scenarios slightly increased large old gum availability beyond 200 years (Table 6.7).

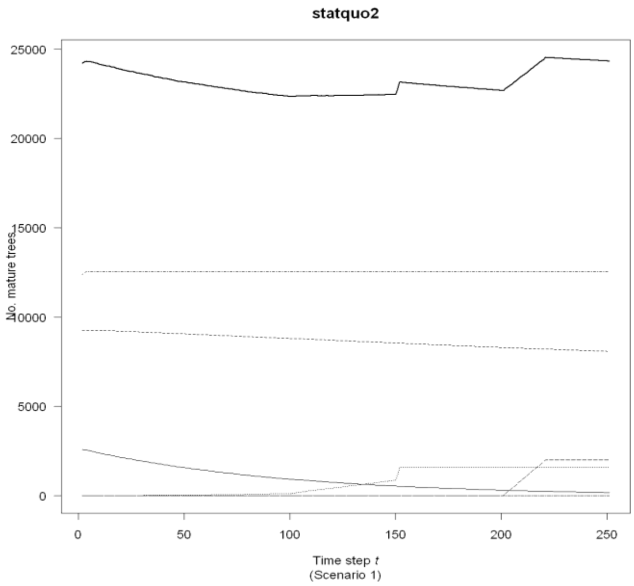



**Figure 6.18.** The status quo scenario of large mature gum availability over 250 years (dark line represents total no. mature trees).

**Table 6.7**. Changes in the number of mature gums as a percentage of current numbers under selected scenarios.

| Scenario no. | Scenario name | % mature trees remaining | | | |
|---|---|---|---|---|---|
| | | 50 years | 100 years | 200 years | 250 years |
| 1 | Status quo | 96 | 92 | 94 | 100 |
| 2 | Extra pivot clearing | 95 | 92 | 94 | 100 |
| 3 | Extra revegetation | 96 | 92 | 94 | 109 |
| 4 | Slow roadside growth | 96 | 93 | 94 | 101 |
| 5 | Offset 4 | 98 | 98 | 102 | 109 |

### 6.4.4 Discussion

The scenarios of gum availability did not show very substantial declines, except in cropping landscapes. At 100 years, approximately 8% of 2004 trees had been lost. This somewhat unexpected outcome was due to the very low rates of tree loss in the grazing landscapes which dominated the sample areas, the maturation of small numbers of ‘younger’ trees in grazing landscapes, the larger number of trees in patches and reserves which had effectively zero rates of loss, and the growth of roadside trees. However, the rates of paddock tree loss, while currently low, may accelerate in the future, as trees age and land use intensifies. Furthermore, there is considerable uncertainty over the maturation rates of roadside trees, which occur at high density and may be prevented from maturing to a stage where they are likely to form hollows by competition.

A major limitation to the model was our inability to include dead hollow trees. The prevalence of such trees in the landscape could not be determined using aerial photography, nor were such trees common enough to be detected during the field component of the study. Thus, the contribution of such trees to current availability of nesting resources and the relevant rates of loss could not be calculated. It seems likely that dead hollow trees are being lost at a much greater rate than live paddock trees (Hill, unpublished data) and so rates of change in nesting resources over the next 50–100 years might be much higher than projected under the scenarios considered. Moreover, it seems likely that dead hollow trees and large paddock trees with hollows are in fact preferred for nesting by RtBCs (Hill, unpublished data), a behaviour which would introduce further uncertainty into interpretation of the scenarios.

However, despite the limitations of the model, the scenarios seem to suggest that the rates of loss of large old gums are likely to be not as great as those of Buloke, or losses of stringybark food availability under several scenarios. Under the conditions we used for the scenarios, revegetation effort of 100 trees per year for 20 years surviving to >200 years within the sampled area, or approximately 4,550 across the RtBC range, was more than adequate to recover resource availability after 200 years. Whether this success rate can be achieved is highly uncertain; however, Red Gums are already frequently included in revegetation mixes and grow easily when direct seeded, so it could be considered that effort is adequate. Gum revegetation and regeneration might be encouraged to grow more quickly through thinning or low-density planting.

# 7. GENERAL DISCUSSION & CONCLUSION

The current availability of both feeding and nesting resources for Red-tailed Black-Cockatoos has declined substantially over the past 40–60 years. From this, it appears likely that the population of cockatoos has declined as a consequence. Assuming a unitary relationship between stringybark availability and the RtBC population size, it may be estimated that the population has declined by approximately 44% since 1947. This does not take into account the greater losses of Buloke, a preferred resource, consideration of which may increase the estimate of population decline. However, although food appears to be limiting the population currently, this does not mean that it always limited population size. In the intact landscapes in which the taxon evolved, other factors may have regulated population size and food availability only became important once feeding habitat was reduced.

The availability of resources of all types for the RtBC has reduced significantly, by at least 30% since 1947. However, although clearing of stringybark, Buloke and gum eucalypt woodland has been extensive in the past, the current trends differ substantially among these resource types. Current trends show little change in stringybark cover from year to year when considered as a proportion of the overall stringybark distribution. On the other hand, Buloke and gum eucalypt resources continue to be lost. This is related to the spatial pattern of their occurrence. Both of these resource types now predominantly occur as paddock trees on private land, while stringybark is mostly located in forest blocks, many of which are publicly owned. There are therefore two factors driving the ongoing losses of Bulokes and gum eucalypts. Firstly, these species continue to be subject to deliberate removal to allow more intensive agriculture. Secondly, as they are often surrounded by agricultural practices which place additional stress on the trees, such as cultivation and pressure from livestock grazing, ‘natural’ attrition is likely to be increased.

Agricultural land uses also prevent regeneration of the trees in affected areas. Smaller, younger Buloke and gum eucalypt trees are restricted to roadsides, state forests and reserves, as well as sites on private land set aside for conservation. However, the models suggested that very few of these young trees are likely to reach maturity, and if they do, most will simply replace other large old trees as those are lost, as they occur at a very high density. When considered across the landscape, there is limited potential for increases in Buloke availability in patches and on roadsides due to already high tree densities. For gum eucalypts, trees on roadsides conceivably could add to future resource availability, but the rate of hollow formation in trees occurring at such high density is unknown.

The representation of gum eucalypts and especially Bulokes in forests and reserves is very low, and the trees typically are smaller than those in paddocks. This smaller mean size is due to the inclusion of dense young regeneration in the figure, but even the largest trees in woodland patches have a relatively smaller trunk diameter. This may be due to the greater levels of competition for light, water and nutrients, and to differences in soil fertility between reserves and paddocks. Such differences are likely to make Bulokes in paddocks more productive than those in woodlands, and increase growth rates of paddock gums, potentially resulting in a greater probability of hollow formation.

The food value of stringybark trees varies substantially depending on the landscape context of the trees. This has substantial implications for the focus of current efforts to increase the availability of stringybark food resources for the RtBC. The focus currently is on managing prescribed burns so that less of the canopy is scorched. However, this research suggests that loss of paddock trees would be of greater consequence to stringybark food resource availability than is prescribed burning. As a consequence of the unequal resource value of trees of each species of different size, the scenario models revealed high sensitivity to losses of paddock trees, rather than the smaller or less valuable trees in forests and woodlands.

Even within a species, all trees are not equal in their value for the RtBC. Across stringybarks, gums and Bulokes, we can conclude that the most threatened resources in the landscape—paddock trees—are also the most valuable. They are also the most difficult and expensive to protect. Our response to managing the losses of these paddock trees over the next 100 years will be critical to the recovery of the RtBC in the medium- to long-term.

## *7.1 RECOMMENDATIONS*

It is important to remember that the scenarios presented in the report are not predictions of future resource availability; rather, they are projections using the best data we have on current availability and rates of change. Further research, particularly to establish more firmly the values of parameters to which the scenarios were sensitive such as the effect of grazing on stringybark food value, will be important in refining the scenarios. Nevertheless, the variation among scenarios led us to a series of recommendations which would appear to offer the best chance of maintaining and increasing the future population of the RtBC, through maximising long-term resource availability for the RtBC. We suggest several general principles to help guide future planning and investment within the RtBC range, and a suite of more specific recommendations for each of the key resource types.

**General principles**

**1. Plant and protect scarce resources to maximise complementarity.**
The RtBC population in any given year will be limited by the point of lowest food availability. Therefore, to reduce the severity of food availability bottlenecks, resources which are most limiting in the landscape should be the focus of revegetation and protection efforts. This means encouraging a focus on Buloke and *E. arenacea* wherever possible.

**2. Protect paddock trees**
Paddock trees of all resource types are disproportionately valuable for the RtBC. Deliberate clearing of paddock trees should be minimised and the lifespan of such trees enhanced through reducing damage from stubble burns, livestock and cropping practices wherever possible.

**3. Ensure offsets for clearing paddock trees protect other paddock trees**
Where clearing of paddock trees is unavoidable, offsets must include the protection of other paddock trees under similar threat. Resource bottlenecks can be minimised more effectively in this way as trees in patches are typically of lower food value and less threatened than paddock trees.

**4. Maintain appropriate stem density**
For all resource types, larger trees at lower density are more valuable for the RtBC. Current revegetation and regeneration on roadsides is at very high density, so efforts to reduce densities or plant at lower initial densities may increase growth rates of individual trees.

**Specific recommendations**

*Buloke*

- ❑ **Increase the level of protection of Buloke paddock trees across the range of the RtBC.** Reduced clearing, the use of paddock trees as offsets and provision of financial incentives to landholders who protect paddock buloke trees could assist in improving the long-term outlook for this resource type, thereby reducing the severity of the likely resource bottleneck. Reducing the death rates of other scattered paddock trees when clearing of paddock Bulokes will achieve the greatest reduction in the severity of the resource bottleneck among scenarios which include ongoing clearing. Protection of paddock Bulokes could include a cessation of stubble burning, stock impacts and tillage near the root zone and planting of young trees nearby to reduce wind impact. Mistletoe removal does not appear to be warranted.

- **Increase revegetation efforts and improve revegetation survival rates.** Current estimated revegetation effort will not achieve sufficient recovery in resource availability for the RtBC. To achieve a return to 2004 numbers of large old Bulokes in 120 years, revegetation at the level of 11,000 to 18,000 trees planted per year (depending on clearing and offset scenarios) for the next 20 years, with a survival rate of 10% at 100 years, would be required. To maintain this level, another 1,000 Bulokes would need to be planted per year thereafter. Natural Buloke revegetation should also be encouraged in areas where stem density is currently < 84 stems per hectare.
- **Focus Buloke protection and revegetation in higher rainfall areas**. There are two reasons for this recommendation. Firstly, the climate envelope in which Buloke currently occurs is expected to shift substantially under scenarios of 20% reductions in rainfall. By focussing on areas where rainfall is currently highest, the risk of negative climate change impacts is reduced. Secondly, higher rainfall areas appear to produce larger and more productive Buloke trees, and most sightings of RtBCs are from these areas. Therefore, although all Buloke within the RtBC range should be considered critical habitat, greater gains may be achieved through focussing investment in higher rainfall areas (see Appendix 4 for a map of priority planting areas).
- **Focus Buloke revegetation on soils with a shallow A-horizon**. Current efforts by La Trobe University researchers are suggesting that greater revegetation success is likely at sites where the sandy A-horizon is less than 30 cm deep (J. Morgan, pers. comm.). This is likely to be due to the moister clay of the B-horizon being more available to the root systems of young plants. Although this research is ongoing, it suggests that where a choice is possible, such soils should be preferred for revegetation.

*Stringybark*

- **Increase the level of protection for stringybark paddock trees across the range of the RtBC.** This could potentially be achieved through a community and council/shire awareness-raising campaign emphasising the value of stringybarks in paddocks combined with a change in vegetation clearance regulations at the policy and planning level. On the other hand, an incentive program focussing on the protection and replanting of paddock trees may produce better results. Further consultation with landholders is required to determine community perceptions about stringybarks on farms and identify the best management approach.

- **Plant *E. arenacea* paddock trees in appropriate areas** (ideally, within 1–5 km of existing stringybark remnants as this is where most nesting occurs; see Appendix 5). Replanting and protection of 'paddock' or low-density trees is likely to have a much greater positive impact on stringybark seed availability than the enhancement of remnant patches on private land. However, a continued emphasis on remnant protection through conservation covenants, bush heritage agreements etc. is essential to prevent further loss.
- **Broad-scale revegetation initiatives involving stringybark-dominated EVCs should focus on *E. arenacea* where possible.** This will achieve a more balanced ratio of *E. baxteri: E. arenacea* that will ensure a good supply of seed in years when only one species of stringybark has a fresh seed crop.
- **Continue emphasis on prescribed burns that minimise canopy scorch.** Ongoing monitoring of this approach is also important, as the long-term impacts on tree and woodland health and condition are not known.

*Gums*

- **Continue existing revegetation efforts.** Such efforts might be enhanced by consideration of the growth rates of trees planted at different densities. Management of regeneration and revegetation sites to achieve a more open structure might be beneficial to growth of trees and increase the probability of hollow formation. Revegetation effort should be focussed within 2–5 km of stringybark blocks to increase suitability of potential future nests for RtBCs (see Appendix 6 for priority planting areas).
- **Avoid losses of paddock trees wherever possible.** Significant numbers of trees are still being lost from cropping landscapes, and clearing for centre pivots is a threat to nesting resources in such landscapes. Although it is difficult to determine whether RtBCs show a preference for nesting in farmland trees, we recommend a precautionary approach wherein such resources are protected wherever possible. If clearing cannot be avoided, the protection of other scattered paddock trees is recommended to help reduce overall resource decline.

## *7.2 KNOWLEDGE GAPS*

*Buloke*

- Perhaps the most critical knowledge gap on which the effectiveness of recommendations for protection of paddock Bulokes depends is the unknown degree to which 'protection' can reduce paddock Buloke death rates. The health and productivity of trees in paddocks from which negative impacts have been removed should be investigated and rates of loss followed over time to determine whether offsets for clearing which consist of paddock tree protection can to any degree offset tree losses.

- For the purposes of the model, we have assumed that trees occurring in dense regrowth areas naturally thin and develop into large mature trees at the same rate as trees spaced more widely. This assumption is unlikely to be realistic, as densely-growing roadside trees appear 'locked up' with little growth occurring. Investigation is required into the potential for manually thinning areas of dense buloke regrowth where large mature trees do not already occur at high density to increase the growth rates of individual trees.
- Should methods for increasing Buloke seedling survival and growth rates be identified, the severity of resource bottlenecks could be reduced by increased revegetation. Investigating the effects of alternative methods of revegetation and the presence of microbial symbionts on Buloke seedling growth should be a priority.
- Buloke appears to be more productive and suitable as an RtBC food source where it occurs as paddock trees than when in denser woodlands and as subcanopy in forests. This observation should be further investigated, as it would have significant implications for identifying the relative value of trees in paddocks and woodlands from the perspective of the RtBC.
- More detailed research is required to quantify the proportional contribution of land management practices other than centre pivot irrigation (such as stubble burning, cattle grazing) as specific causes of Buloke deaths. This would allow the targeting of education programs aimed at encouraging farmers to reduce the impacts of farming practices on paddock trees.
- Most sightings of RtBCs feeding in Buloke are from the western half of the cockatoo's range. This may reflect different observer effort or a genuine preference for trees in higher rainfall areas or on particular soils. It is important to determine whether such a preference exists as the rates of tree loss in the western part of the range are higher and development pressures are greater in that area.

*Stringybark*

- A study is required to determine the factors influencing paddock tree productivity (eg. soil nutrients, stem density, fertilizer application and grazing practices) in terms of tree health and seed production to assist in the development of planting recommendations. Ideally, additional paddock tree data should be collected to determine the amount of variability in paddock tree multipliers for *E. baxteri* and *E. arenacea* across the RtBC range.

- A significant question remains about the importance of stringybark paddock trees as a food resource. In particular, it is uncertain whether the cockatoos use paddock trees as a primary food source or as a supplementary food source when food availability in surrounding remnants is low. This study could involve foraging observations to compare the foraging efficiency of birds feeding in paddock trees (where flight distances are greater) to those feeding in remnant vegetation and monitoring of bird foraging locations at different times since new seed crop production.
- The high rates of stringybark habitat loss within the GHCMA region are of concern. A more detailed study investigating the causes (illegal and/or legal) of recent habitat loss is required to determine whether further action is required.
- An in-depth study is required to determine the importance of food availability as a factor limiting the breeding success of the RtBC population. The Recovery Team is currently monitoring stringybark seed availability across the range and, to a lesser degree, nest success. Another possibility would be to compare nestling weights and nest success between "good" (years of new seed production) and "poor" (one or more years since new seed crop production). Ideally, this study would be conducted in the same nesting area (eg. Corndale).

*Gums*

- A major limitation of this study is that it could not quantify the availability and rates of loss of dead hollow trees. Before the projections of gum availability produced under the scenarios considered can be translated to potential availability of nesting resources, data need to be gathered on the current availability and recent rates of change of dead gums and their probability of containing a suitable hollow compared with large live gums.
- Growth and hollow formation rates of gums occurring at different stem densities in the study regions are unknown. Although we have modelled the availability of gums 200 years and older assuming these represent a reasonable approximation of nest resources available to RtBCs, further work in this area could refine the model to include additional information on growth and hollow formation rates.

# REFERENCES

Burnard, T. and Hill, R. (2002). *Draft South-eastern Red-tailed Black-Cockatoo Recovery Plan.* Birds Australia report to Department of Environment and Water Resources, Canberra.

Cameron, M. and Collins, D. (1992). *Point Henry Bird Surveys 1991-1992*. Report prepared for ALCOA of Australia Limited, Geelong, Victoria.

Carr, G.W. and Buxton, M. (1995). *ALCOA Point Henry Drainage Consolidation Project: W1 and W2/3 Drain Outlets – Wetland and Vegetation Design and Site Amelioration at Pondages*. Report prepared for ALCOA of Australia Limited, Geelong, Victoria. Ecology Australia Pty Ltd, Victoria.

Commonwealth of Australia. (2007). *National Recovery Plan for the South-eastern Red-tailed Black-Cockatoo Calyptorhynchus banksii graptogyne*. Department of the Environment and Water Resources, Canberra.

DSE (2004) *Vegetation Quality Assessment Manual–Guidelines for applying the habitat hectares scoring method. Version 1.3.* Victorian Government Department of Sustainability and Environment, Melbourne.

EPBC (1999). *Environment Protection Biodiversity Conservation Act (1999).* Commonwealth of Australia, Canberra.

Garnett, S. T. and Crowley, G. M. (2000). *The Action Plan for Australian Birds.* Environment Australia, Canberra.

Gibbons, P. and Lindenmayer, D. B. (2002). *Tree Hollows and Wildlife Conservation in Australia*. CSIRO publishing, Collingwood.

Gibbons, P., Lindenmayer, D. B., Barry, S. C. and Tanton, M. T. (2000) Hollow formation in eucalypts from temperate forests in southeastern Australia. *Pacific Conservation Biology* 6:218-228.

Hedditch, K. (2007). *Chain of Waterholes. A History of the Glenelg River*. Program Print, Victoria, Australia.

Hill, R and Burnard, T. (2001). *A Draft Habitat Management Plan for the South-eastern Red-tailed Black-Cockatoo.* Unpublished report to the Red-tailed Black-Cockatoo Recovery Team.

Hill, R. (in prep). *The conservation biology of the south-eastern Red-tailed Black-Cockatoo*. Unpublished report to the Red-tailed Black-Cockatoo Recovery Team.

Joseph, L., Emison, W. B. and Bren, W. M. (1991). Critical assessment of the conservation status of the Red-tailed Black-Cockatoo in south-eastern Australia with special reference to nesting requirements. *Emu* 91:46–50

Koch, P. (2003). *Factors influencing food availability for the endangered south-eastern Red-tailed Black-Cockatoo* Calyptorhynchus banksii graptogyne *in remnant stringybark woodland, and implications for management*. PhD thesis, University of Adelaide, Adelaide.

Luke, R. H., and McArthur, A. G. (1978). *Bushfires in Australia*. Australian Government Publishing Service, Canberra.

Maron, M. (2000). *Characteristics of feeding sites of the endangered south-eastern Red-tailed Black-Cockatoo* Calyptorhynchus banksii graptogyne *in remnant Buloke* Allocasuarina luehmannii *woodland*. Honours Thesis, Monash University.

Maron, M. (2005). *Threatened Species and Farming Report No 2. South-eastern Red-tailed Black Cockatoo.* Arthur Rylah Institute for Environmental Research, Heidelberg.

Maron, M., and Fitzsimons, J. A. (2007). Agricultural intensification and loss of matrix habitat over 23 years in the West Wimmera, south-eastern Australia. *Biological Conservation* 135:603–609.

Maron, M., and Lill, A. (2004). Discrimination among potential Buloke (*Allocasuarina luehmannii*) feeding trees by the endangered south-eastern red-tailed black-cockatoo (*Calyptorhynchus banksii graptogyne*). *Wildlife Research* 31:311–317.

Purdie, R.W. and Slayter, R.O. (1976). Vegetative succession after fire in sclerophyll woodland communities in south-eastern Australia. *Australian Journal of Ecology* 1:223-236.

Reekie, E. G., and Bazzaz, F. A. (1987). Reproductive Effort in Plants. 3. Effect of Reproduction on Vegetative Activity. *American Naturalist* 129:907–919.

Rudolph, P. (2006). *DRAFT Durong and Jilpanger State Forests ecological post-burn survey 2006*. DSE Horsham, Victoria.

Saunders, D. A. 1989. Changes in the avifauna of a region, district and remnant as a result of fragmentation of native vegetation: the wheatbelt of Western Australia. A case study. *Biological Conservation* 50:99–135.

Tolhurst, K.G. (1996). Effects of fuel reduction burning on fauna in a dry schlerophyll forest. In *Fire and biodiversity: the effects and effectiveness of fire management*. Biodiversity Unit, Department of Environment, Sport and Territory.

Underwood, A. J. (1981). Techniques of analysis of variance in experimental marine biology and ecology. *Oceanogrography and Marine Biology Annual Review* 19:513–605.

Watson, D. M. (2001). Mistletoe—a keystone resource in forests and woodlands worldwide. *Annual Review of Ecology and Systematics* 32:219–249

Wilson, B.A. (1996). Fire effects on vertebrate fauna and implications for fuel reduction burning and management. Pp. 131–147. In *Fire and biodiversity: the effects and* *effectiveness of fire management*. Biodiversity Unit, Department of Environment, Sport and Territory.

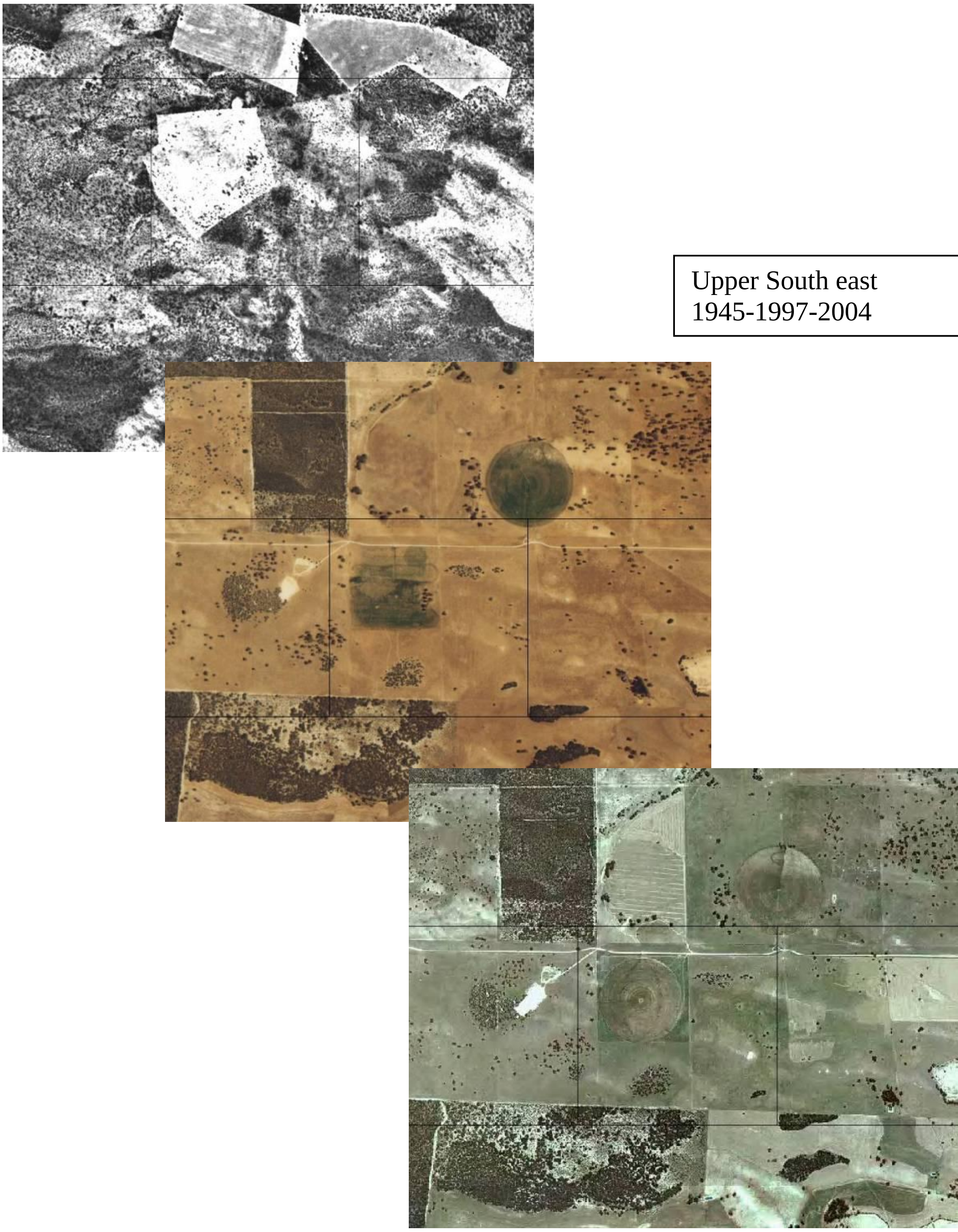
Yates, C. J. and Hobbs, R. J. (1997).Temperate eucalypt woodlands: a review of their status, processes threatening their persistence and techniques for restoration. *Australian Journal of Botany* 45:949–973

Yates, C. J., Hobbs, R.J. and Bell, R.B. (1994). Factors limiting the recruitment of *Eucalyptus salmonophloia* in remnant woodlands. I. Patterns of flowering, seed production and seed fall. *Australian Journal of Botany* 42:531–542.

Zar, J.H. (1984). *Biostatistical Analysis (2$^{nd}$ Edition).* Prentice-Hall, USA.

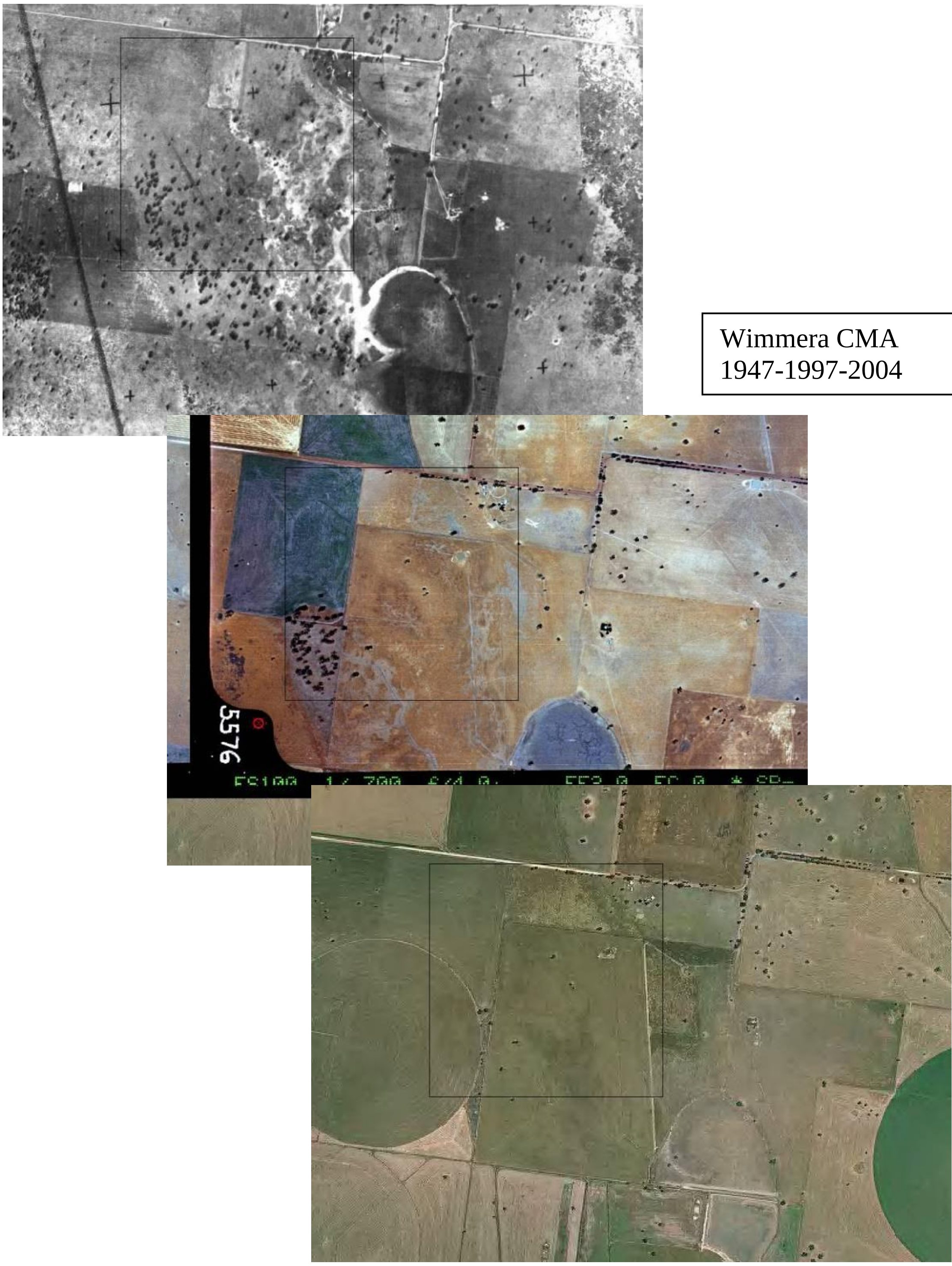
# APPENDICES

***APPENDIX 1.*** Example landscape showing changes in the Upper South east of South Australia 1945-1997-2004.



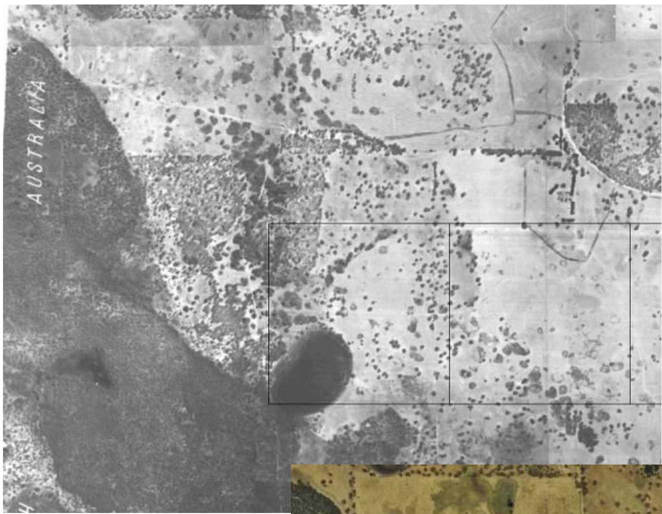
Upper South east
1945-1997-2004

## ***APPENDIX 2.*** Example landscape showing changes in the Wimmera CMA region 1945-1997-2004.



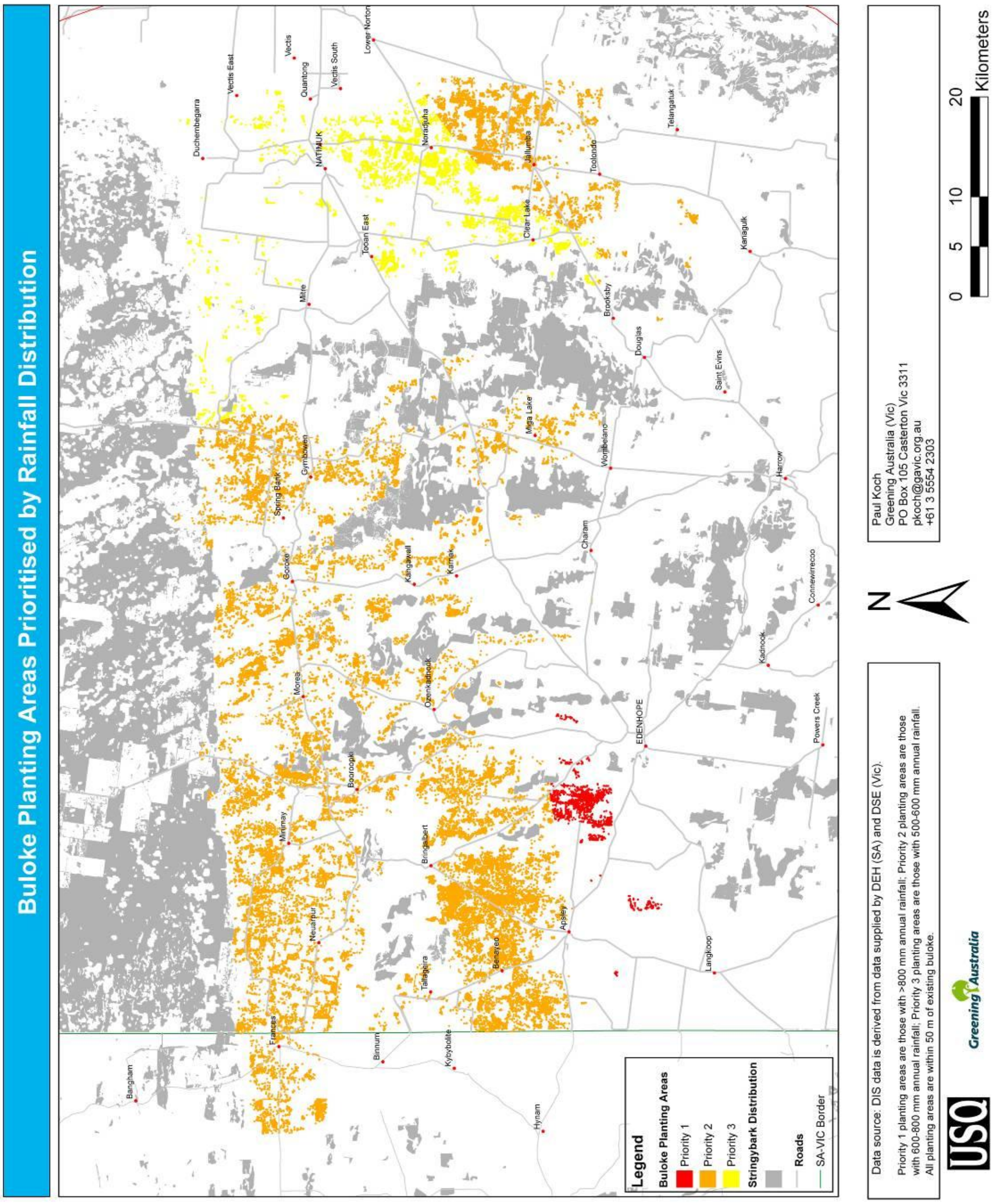

Wimmera CMA
1947-1997-2004

## ***APPENDIX 3.*** Example landscape showing changes in the Glenelg-Hopkins CMA region 1945-1992-2004.

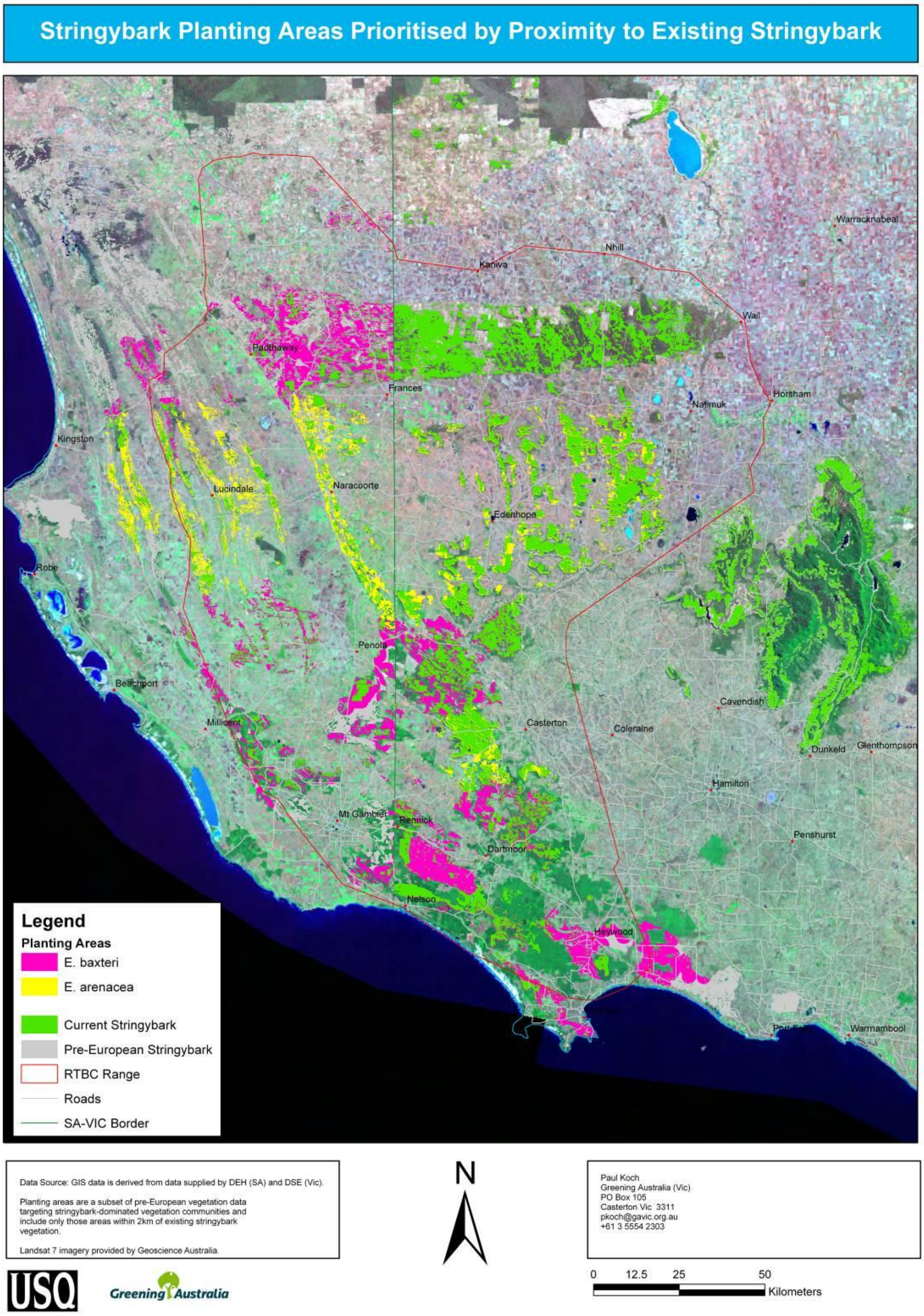



Glenelg Hopkins CMA
1947-1992-2004

# ***APPENDIX 4.*** Recommended Buloke planting areas



# ***APPENDIX 5.*** Recommended stringybark protection and planting areas



## ***APPENDIX 6.*** Recommended stringybark protection and planting areas