Future Ecosystems

Habitat Extent Mapping and Rates of Habitat Change for the 2003 to 2017 period Across Feeding and Nesting Resources for the South-eastern Red-tailed Black Cockatoo

Report prepared for the South-eastern Red-tailed Black Cockatoo Recovery Team

19/08/19

1. Executive Summary

An ongoing decline in the quality and quantity of food resources has been identified as a major threat to the south-eastern Red-tailed Black-Cockatoo *Calyptorhynchus banksii graptogyne*. Recent research has documented one of the highest rates of loss of scattered paddock trees in Australia within the RtBC's range for the Buloke food resource (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). A recent study investigating historical and recent rates (1997-2004) of habitat loss by Maron et al. (2011) confirmed ongoing and severe habitat loss in the case of the Buloke food resource and relatively minor but significant rates of habitat loss in the case of the stringybark food resource and gum woodland nesting resource.

The aims of the study were to refine existing habitat extent layers for stringybark feeding habitat and gum nesting habitat, and to determine rates of habitat loss occurring over the past 10-13 years and compare them to rates of loss in the preceding 7 years reported by Maron et al. (2011), for stringybark feeding habitat, gum nesting habitat and buloke feeding habitat

For the stringybark food resource, an overall net gain was achieved when averaged across both stringybark species (average gain of 0.03% per annum). Within stringybark woodland types, however, *E. arenacea* showed a slight negative overall trend while *E. baxteri* showed a relatively strong positive trend. This is a substantial improvement on the habitat change figures reported by Maron et al. (2008) for the 1997 to 2004 study period, which averaged 0.04% loss per annum across both species of stringybark. However, the paddock tree component of stringybark cover is strongly declining at a much faster rate than trees in remnants for both *E. arenacea* (0.7% per annum) and *E. baxteri* (0.24% per annum), meaning that it would take 143 years and 417 years to lose all paddock tree cover for these two species, respectively.

Trends for gum woodlands (nesting habitat) were similar to those of stringybark woodlands. An overall net gain was achieved in terms of total cover of gums (0.01% per annum), which was attributed to natural regeneration (from conservation practices such as stock exclusion) and revegetation efforts. Percentage change of total remnant cover was positive for the Glenelg Hopkins CMA and Natural Resources South East regions but was negative for the Wimmera CMA region. This is a substantial improvement on figures reported by Maron et al. (2004) which showed a net per annum loss of 0.32% for the 1997 to 2004 period. The improvement seems to be mainly due to reduced losses from cropping and centre pivots, and no doubt conservation efforts in the region have also played an important role.

However, net paddock tree cover loss for gums was substantial and occurred across all three NRM regions, averaging around 9% across all regions for the time period and 2.4% per annum. This suggests that, although there has been a slight net gain in tree cover overall, gains have occurred mainly in remnant patches (where trees tend to be smaller due to high densities) and paddock trees are still declining. Rates of paddock tree loss for gum woodlands appear to be accelerating since the 1997 to 2004 period. If this rate of paddock tree loss continues, all paddock trees would be dead within 42 years. Investigation into the causes of paddock tree loss suggested that tree mortality due to drought stress and dieback was at least as important as clearing for agricultural activities such as centre pivots. Tree dieback was noted in 32% of samples where gum paddock tree loss occurred and tree mortality (together with trees showing symptoms of dieback) was noted at a further 13% of samples. This trend is likely to accelerate as climate change progresses.

Total Buloke cover showed a slight negative trend of 0.003 per annum (meaning it would take 3000 years at this rate to lose all remaining trees), which is a substantial improvement on the rate of 1.4% loss per annum reported for the 1997 to 2003 period. However, Buloke paddock trees continue to decline at a rate of 0.2% per annum and these trees generally provide much better foraging habitat than trees in remnants and on roadsides.

Much of the loss in Buloke paddock tree cover was attributed to tree dieback. Dieback was noted in 14 out of 70 samples containing Buloke, with Buloke mortality apparent at seven of these samples. By contrast, deliberate clearing of Bulokes for pivots was only noted in two of 70 samples. These results suggest that tree dieback is at least as important as clearance for the Buloke food resource. Bulokes are known for their longevity and resilience as paddock trees and this is the first time to the author’s knowledge that extensive dieback has been reported for this species. It seems likely therefore that Buloke, like many other tree species, is susceptible to climate change related heat and drought stress, which means that we may see accelerating rates of Buloke loss over the coming decades.

In summary, the results of this study indicate substantial improvement in rates of habitat loss and indeed some habitat types (Brown Stringybark and Gum eucalypts) are currently increasing in terms of total tree cover. On the other hand, the paddock tree component of this cover is strongly declining across all major habitat types, due in large part to tree dieback and tree mortality. It is recommended that further analysis and planning is undertaken to confirm trends related to tree dieback and identify climate adaptation strategies to ensure that replanted trees providing critical feeding and nesting habitat can survive into the future.

2. Background

An ongoing decline in the quality and quantity of food resources has been identified as a major threat to the south-eastern Red-tailed Black-Cockatoo *Calyptorhynchus banksii graptogyne*. Recent research has documented one of the highest rates of loss of scattered paddock trees in Australia within the RtBC's range for the Buloke food resource (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). A recent study investigating historical and recent rates (1997-2004) of habitat loss by Maron et al. (2011) confirmed ongoing and severe habitat loss in the case of the Buloke food resource and relatively minor but significant rates of habitat loss in the case of the stringybark food resource and gum woodland nesting resource.

Supervised classification of high resolution imagery can be used to automate the process of discriminating paddock trees, but the method is problematic for dead trees and producing a highly accurate result is time consuming to achieve across the entire range of the RtBC. Nevertheless, having a reasonably accurate baseline of paddock tree density across the range of the RTBC is highly desirable for the spatially targeting of management activities such as nest box installation, protection of stringybark paddock trees, replanting of paddock trees etc.

This study therefore proposes to use and compare both methods for determining rates of habitat loss. Repeating the methodology by Maron et al. (2011) will ensure that results are directly comparable with the previous study so that we can determine if recent rates of habitat loss are becoming more or less severe. Automated classification of high resolution aerial imagery will be used to achieve a reasonably accurate baseline of paddock tree density across the range of the RTBC. The aims of the study were:

- To refine existing habitat extent layers for stringybark feeding habitat and gum nesting habitat, using recent treecover datasets derived from high resolution satellite imagery and adding in paddock trees mapped based on image classification
- To determine rates of habitat loss occurring over the past 12 years (2004-2016) and compare them to rates of loss in the preceding 7 years reported by Maron et al. (2011), for stringybark feeding habitat, gum nesting habitat and buloke feeding habitat

3. Methodology

3.1 Establishing a baseline paddock tree layer for the whole RTBC range

Paddock trees were mapped for this project using supervised classification of high resolution aerial imagery. This included a 2013 image for the Natural Resources South East region, a 2004 image for the Wimmera CMA and a 2003 image for the Glenelg Hopkins CMA. The more recent images for the Wimmera (2017 image) and Glenelg Hopkins (2013 image) CMA's were too large in terms of file size to process at the required scale. Data accuracy was improved using manual GIS techniques to clean up misclassified areas, including exclusion of particular land use types such as irrigated cropping areas, wetlands and plantations. Plantations were particularly problematic in that it was not possible to accurately distinguish between paddock trees from surrounding plantations using classification techniques.

The paddock tree density layer was produced by calculating the amount of paddock tree cover occurring within a 4km radius, which is the scale that seems to be appropriate for the SE RTBC (existing nest trees are usually within a 4km radius of stringybark remnants; Duong and Maron, in press).

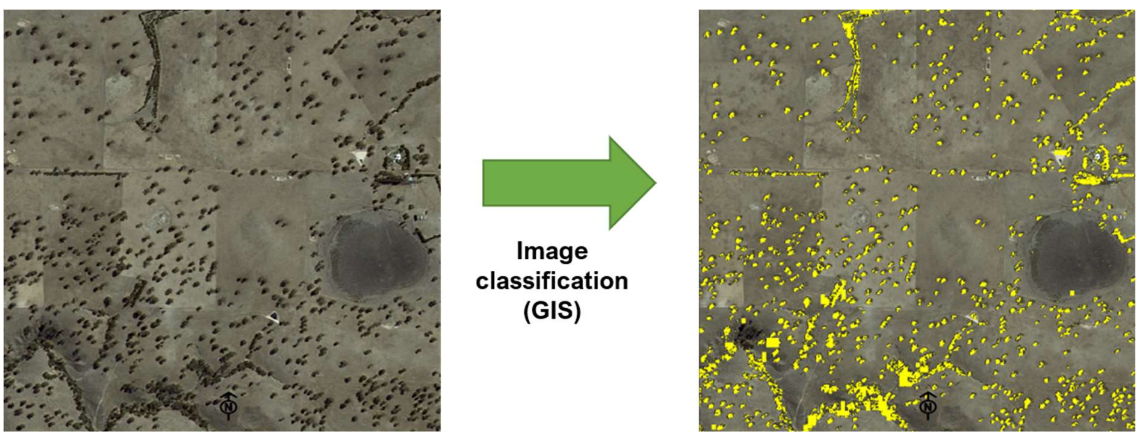

Figure 3.1. Sample area showing results of image classification used to map paddock tree cover.

3.2 Habitat extent mapping

Habitat extent mapping was refined through this project using various GIS techniques to update existing habitat extent datasets based on native vegetation mapping with new landcover/ treecover datasets that provide more accurate mapping for small remnants on private land.

Paddock trees mapped using supervised classification of aerial imagery (refer to section 3.1) were incorporated into the habitat extent layers and classified as Buloke, Desert Stringybark, Brown Stringybark or gums based on pre-European vegetation mapping and Buloke mapping work by Maron et al. (2005). Additional native vegetation layers such as those produced by plantation forestry companies were used to refine the dataset where possible (these layers were pulled together by Darren Herpich from DEW).

3.3 Determining rates of habitat loss

The principle method for quantifying habitat loss follows that developed by Maron et al. (2008) and uses the same sample areas, including sixteen 30 km long x 1 km wide transects were located within critical habitat areas across the RtBC range, with 5 transects located in each NRM region. Transects were further divided into 1 x 1 km sample plots. Each sample plot has previously been classified into different land uses, different land tenures, different habitat types and different habitat configurations (remnant, paddock trees or mixed). Total percentage native vegetation cover was recorded for each sample for each time period.

Aerial imagery used for the study comprised 2003 and 2013 capture years for the Glenelg Hopkins CMA and Natural Resources South East regions and 2004 and 2017 capture years for the Wimmera CMA region. The resolution of imagery varied between 125cm for the GHCMA 2003 imagery to 20cm for the Wimmera CMA 2017 imagery.

One issue with the previous methodology was that many samples were a mixture of paddock trees and remnant cover, therefore it was difficult to clearly distinguish between losses associated with paddock trees and losses associated with remnant cover. To address this issue, paddock tree losses/gains were additionally quantified by counting the total number of paddock trees for every sample area and counting the number of paddock trees lost or gained between the two time periods. By doing this, paddock tree losses could be quantified more accurately, while still enabling total cover comparisons with the previous time period. Buloke distribution mapping from the Maron et al. (2008) study was used to help identify Bulokes from gum trees and separate paddock tree counts and loss/gain counts were undertaken for Bulokes.

In addition, notes were made for every sample about suspected causes of paddock tree losses and gains. Tree dieback and tree mortality (presence of a standing dead tree that had previously been alive in the earlier capture period) were also noted for each cell where it occurred. In the case of tree dieback, it was only noted if it clearly and obviously affected multiple trees within the sample. This was only possible due to the improved resolution of and quality of aerial imagery in recent years (20cm to 25cm).

This method produces a highly accurate estimate for particular areas, but only represents 2% of the total RtBC critical habitat area (which includes all area within 5km of stringybark woodlands). However, other methods that were investigated through this study proved to be unfeasible. Existing global, national and state-based tree cover datasets were too coarse to pick up individual paddock trees. Aerial or satellite imagery can be used for the classification of paddock trees provided that the resolution is less than 5m. Unfortunately, differences in image quality across capture years and across NRM regions meant that different amounts of tree cover are detected by image classification in a given year. More recent images tended to be higher quality (in terms of image contrast and resolution) and thus produced the best outputs of tree cover following image classification. This variability in image quality precluded accurate comparison of tree cover across time periods using automated classification methods. The only other option would be to purchase high resolution satellite imagery such as Quickbird, but this would be expensive given the scale of imagery required.

4. Results

4.1 Habitat Extent Mapping

Figures 4.1 and 4.2 show the results of paddock tree mapping in relation to different habitat types (Buloke feeding habitat, stringybark feeding habitat and gum nesting habitat). *E. arenacea* made up about 22% of the total stringybark paddock tree cover, which is slightly less than proportional to the pre-European distribution of the two species of stringybark (28% *E. arenacea* to 72% *E. baxteri*; Koch 2003). Stringybark paddock tree cover was around four times higher by area in the South Australian part of the range than the Victorian part of the range (WCMA and GHCMA figures combined). The vast majority of stringybark cover in SA was comprised of *E. baxteri,* but *E. arenacea* paddock tree cover in SA was still about twice that of Victoria.

The total area of Buloke paddock tree cover was 1294 ha and the number of vegetation units (which are generally equivalent to individual paddock trees) was 132,493. This is considerably higher than the 51,477 Buloke paddock trees reported by Maron et al. (2004), however it includes substantial areas of roadside vegetation which would account for the discrepancy.

Gum nesting habitat made up the vast majority of paddock tree cover, with a relatively even distribution between South Australian (NRSE) and Victorian (WCMA and GHCMA combined) parts of the range. Cover of gum nesting habitat was also comparable between the WCMA and the GHCMA parts of the RTBC range.

Table 4.1. Paddock tree and small remnant cover (less than one hectare) mapped through this project (excludes all vegetation in existing native vegetation datasets). Area figures are given for each NRM region and the number of vegetation units (N) is given in brackets.

Habitat Type	**NRSE** Ha (N)	**WCMA** Ha (N)	**GHCMA** Ha (N)	**Total** Ha (N)
Stringybark feeding habitat	6848 (2,713,335)	1,004 (81,970)	709 (56,680)	8,561 (2,851,985)
E. arenacea	1236 (513,575)	542 (47,315)	98 (8,386)	1,876 (569,276)
E. baxteri	5612 (2,199,460)	462 (34,655)	611 (48,294)	6,685 (2,282,409)
Buloke Feeding habitat	39 (13,782)	1,255 (118,711)	0	1,294 (132,493)
Gum Nesting habitat	22,404 (7,851,672)	11,969 (829,695)	12,821 (906,390)	47,194 (9,588,062)

Figure 4.3 shows the results of the refined stringybark mapping, classified into stringybark-dominant areas and stringybark subdominant areas, which generally include a combination of overstorey species or a mosaic of different habitats including areas where other tree species dominate and treeless heathlands.

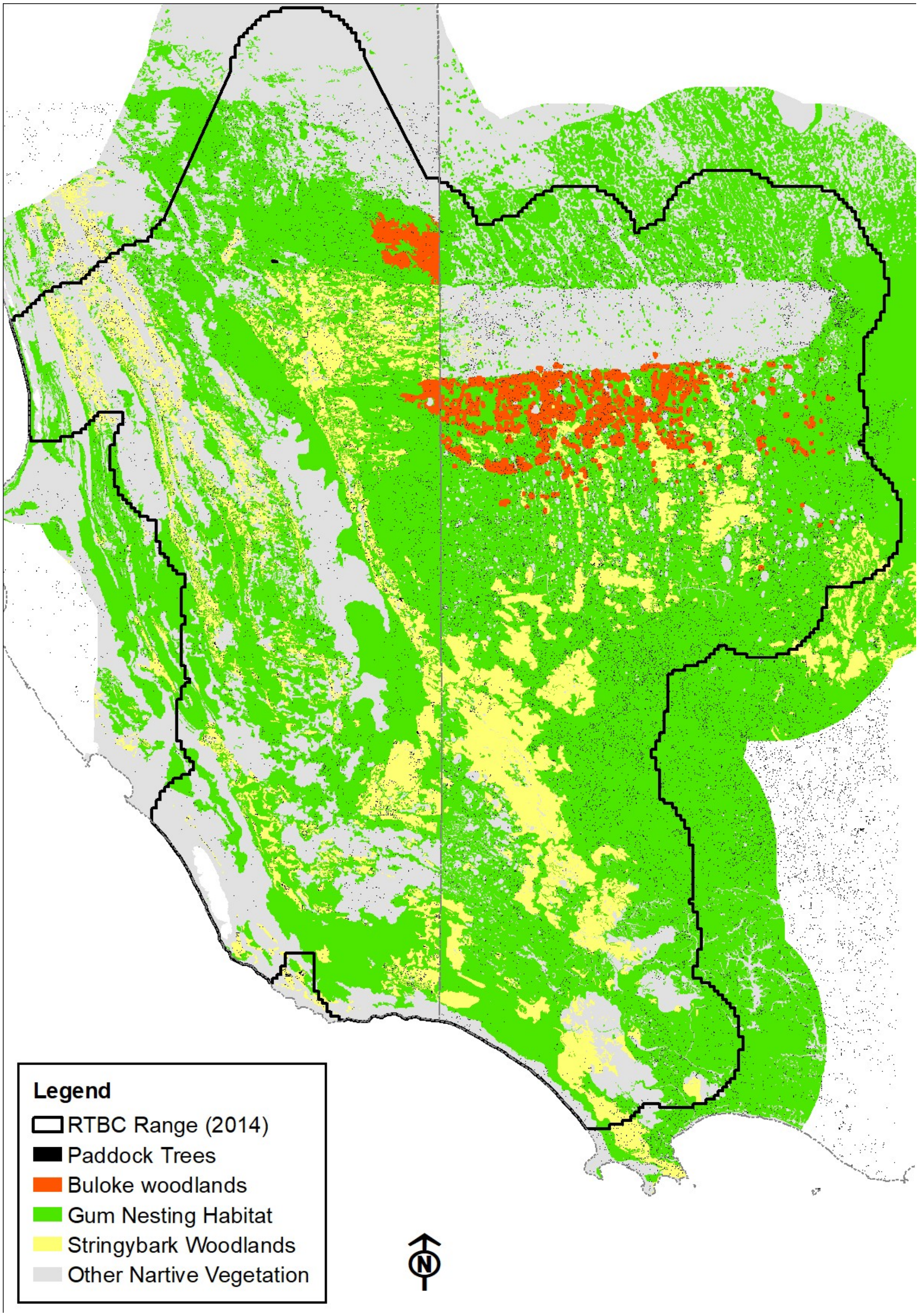

Figure 4.1. Overview of paddock tree mapping against a background of pre1750 habitat types.

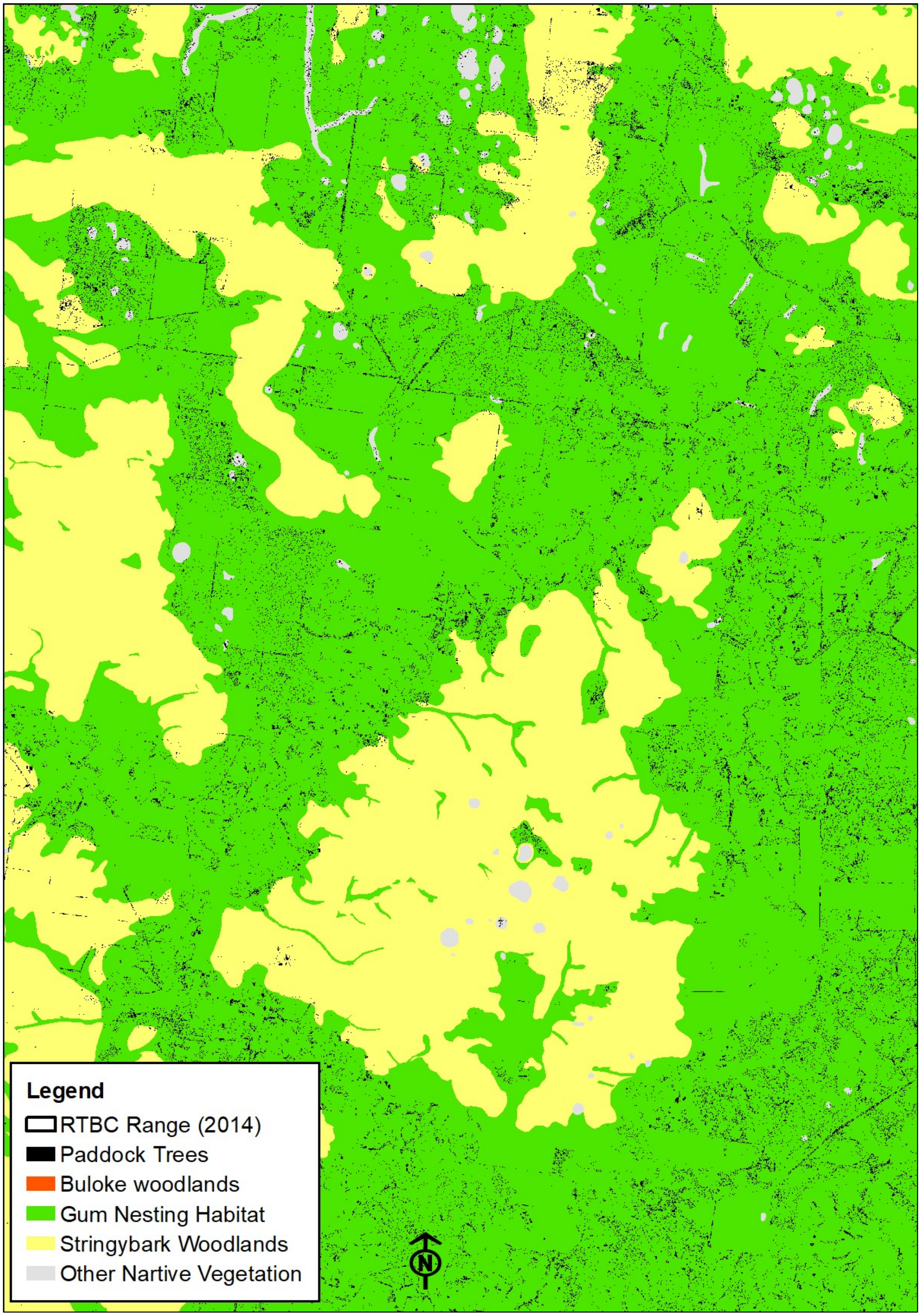

Figure 4.2. Paddock tree mapping against a background of pre-1750 habitat types (zoomed in to Dergholm region of Victoria).

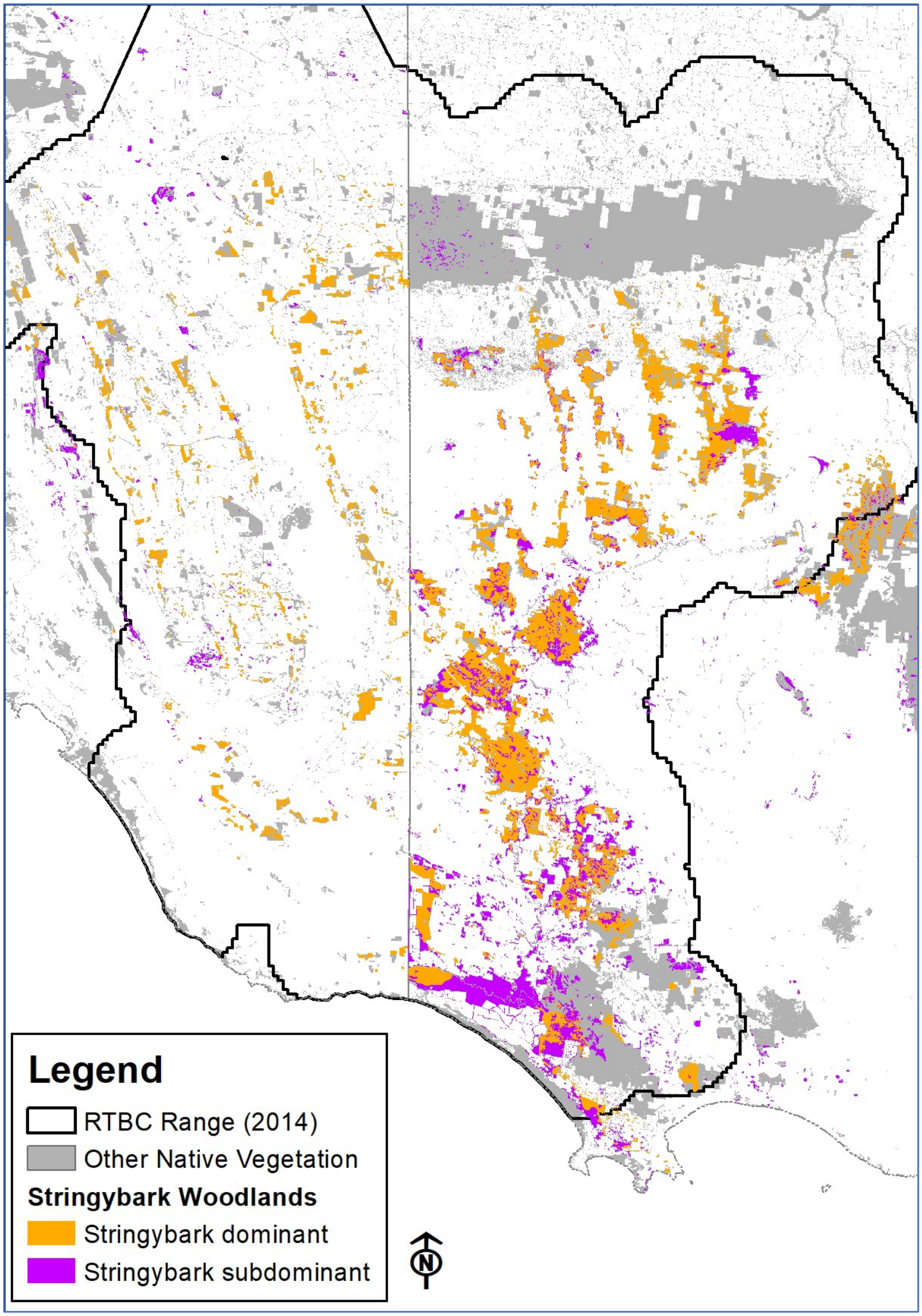

Figure 4.3. Mapping of stringybark habitats classified into stringybark dominant and subdominant areas. Areas mapped as stringybark subdominant generally include a combination of overstorey species or a mosaic of different habitats including areas where other tree species dominate and treeless heathlands.

4.2 Rates of Habitat Change: Stringybark Feeding Habitat

4.2.1 Total stringybark vegetation cover

Changes in the percentage cover of stringybark vegetation are summarised for each NRM region and across all samples in Table 4.1. *E. arenacea* showed a slight negative overall trend while *E. baxteri* showed a relatively strong positive trend. A positive trend was recorded for the stringybark vegetation type as a whole (both species combined), which was associated with natural regeneration (presumably due to exclusion of stock) and revegetation programs.

Negative trends were associated with the WCMA for both species of stringybark. The NRSE region showed a negative trend for *E. baxteri.* For GHCMA, percentage change was neutral for *E. arenacea* and positive for *E. baxteri*.

Table 4.1. Changes in percent cover native vegetation cover for the two species of stringybark averaged across samples between 2003/04 and 2013/17, stratified by natural resource management region.

	No. of samples	Average % change (2003/04 to 2010/13)	Mean % change per annum
E. arenacea	**106**	**-0.03**	**-0.003**
GHCMA	17	0.00	0.00
NRSE	40	-0.06	-0.01
WCMA	49	-0.01	-0.001
E. baxteri	**179**	**0.45**	**0.05**
GHCMA	69	0.07	0.01
NRSE	73	1.34	0.13
WCMA	37	-0.59	-0.05
All stringybark	**285**	**0.27**	**0.03**
GHCMA	86	0.05	0.01
NRSE	113	0.84	0.08
WCMA	86	-0.26	-0.02

Table 4.3 shows changes in the percentage cover of native vegetation stratified by land use type. Slight negative trends were associated with conservation, cropping and grazing land uses while positive trends were associated with centre pivot and plantation land uses. The negative trend within conservation areas was recorded for seven out of 90 samples and was explained by tree dieback and a recent fire at one site. The centre pivot land use recorded a higher number of samples with negative change than positive change, however the level of change for positive samples was proportionally higher than samples with negative change due to replanting of trees adjacent pivots. Within the plantation land use, all samples showed either no change or positive change and the overall trend was strongly positive.

Table 4.2. Changes in percent cover of native vegetation across samples between 2003/04 and 2013/17, stratified by land use type. Number of samples with negative change, no change and positive change and average percent change within these categories are also given.

	No. of samples	Average % change (2003/04 to 2010/13)	Mean % change per annum
Conservation	77	-0.09	-0.007
Negative	7	-1.31	-0.11
No change	66	0.00	0.00
Positive	4	0.57	0.06
Cropping	29	-0.02	-0.001
Negative	5	-0.26	-0.02
No change	18	0.00	0.00
Positive	6	0.15	0.02
Grazing	116	-0.14	-0.01
Negative	31	-0.81	-0.07
No change	71	0.00	0.00
Positive	14	0.63	0.06
Centre Pivot	4	1.90	0.19
Negative	1	-0.10	-0.01
No change	1	0.00	0.000
Positive	2	3.85	0.39
Plantation	59	1.59	0.16
No change	46	0.00	0.00
Positive	13	7.21	0.72

4.2.1 Stringybark paddock tree cover

Table 4.3 shows changes in the cover of stringybark paddock trees, stratified by stringybark species and NRM region. A net loss of stringybark paddock trees occurred across both species of stringybark. The Glenelg Hopkins CMA region showed positive change for *E. baxteri,* although this was based on a small sample size of three samples.

Table 4.3. Paddock tree density and number of paddock trees lost per sample, for those samples where loss occurred between the years 2003 to 2013 (GHCMA and NRSE) and 2004 to 2017 (WCMA).

	No. of Samples	Mean No. of Paddock Trees	Mean No. of Paddock Trees Lost/Gained	Mean % Paddock Tree Change	Mean % Paddock Tree Loss Per Annum
E. arenacea	**23**	**23.0**	**-9.2**	**-8.3**	**-0.70**
SENRM	13	13.0	-11.2	-5.2	-0.52
WCMA	10	10.0	-6.7	-12.3	-0.94
E. baxteri	**31**	**31.0**	**-6.3**	**-3.2**	**-0.23**
GHCMA	3	3.0	1.3	55.5	5.55
NRSE	21	21.0	-7.1	-6.9	-0.69
WCMA	7	7.0	-7.1	-17.2	-1.32
All stringybark	**54**	**102.0**	**10.4**	**0.9**	**-0.43**
GHCMA	3	10.7	1.3	55.5	5.55
NRSE	34	133.4	-8.6	-6.2	-0.62
WCMA	17	55.5	-6.9	-14.3	-1.10

Tree dieback and mortality were important contributing factors to paddock tree loss. Tree dieback was noted in 8% of samples where stringybark paddock tree loss occurred and dead trees (together with trees showing symptoms of dieback) were noted at a further 24% of samples. By contrast, loss due to deliberate clearance accounted for only 3% of samples. The cause of loss could not be determined for the remaining 60% of samples, which were probably associated with a combination of causes. For example, farmers may remove trees that are dead or dying in order to “tidy up” paddocks.

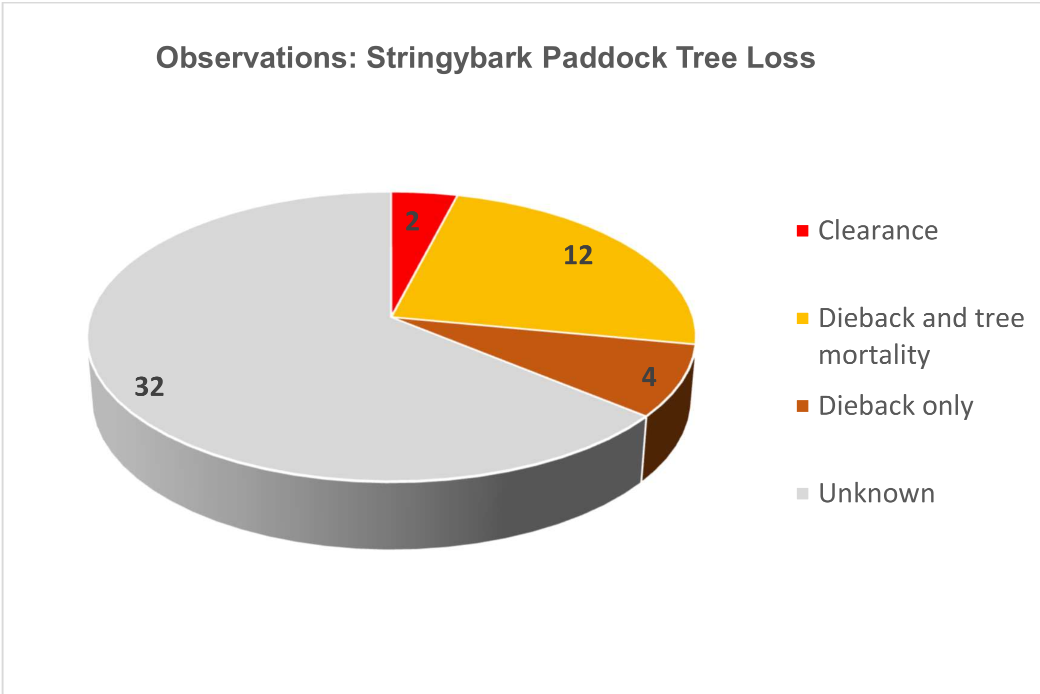

Figure 4.1. Observations associated with stringybark paddock tree loss, including clearance (where centre pivots were put in or significant areas of vegetation disappeared), dieback (where tree dieback and mortality was evident, with at least some dead trees still standing) and unknown (where no clear cause of loss could be determined). Number of samples are given as labels.

4.2 Rates of Habitat Change: Gum Woodland Nesting Habitat

4.2.1 Total gum woodland vegetation cover

Changes in percentage cover of gum woodlands are presented in Table 4.4 (stratified by NRM region). More samples showed negative change than positive change but overall there was a net gain for gum woodland cover, which was attributed to natural regeneration and revegetation efforts. Percentage change was positive for the Glenelg Hopkins CMA and Natural Resources South East regions but was negative for the Wimmera CMA region.

Table 4.4. Changes in percent cover of gum woodlands, averaged across samples between 2003/04 and 2013/17 and stratified by natural resource management region.

	No. of samples	Average % change (2003/04 to 2010/13)	Mean % change per annum
Gum Woodlands	**189**	**0.08**	**0.01**
Negative	26	-0.51	-0.05
No change	146	0.00	0.00
Positive	17	1.61	0.16
GHCMA	52	0.34	0.03
NRSE	73	0.01	0.001
WCMA	64	-0.06	-0.01

Table 4.5 shows percentage changes in total cover of gum woodlands stratified by land use. Trends were slightly negative for cropping and centre pivot land uses and were slightly positive for conservation, grazing and plantation land uses.

Table 4.5. Changes in percent cover of gum woodlands, averaged across samples between 2003/04 and 2013/17 and stratified by land use type. Number of samples with negative change, no change and positive change and average percent change within these categories are also given.

	No. of samples	Average % change (2003/04 to 2010/13)	Mean % change per annum
Conservation	189	0.08	0.01
Negative	13	0.12	0.01
No change	10	0.00	0.00
Positive	3	0.53	0.05
Cropping	48	-0.01	-0.001
Negative	8	-0.14	-0.01
No change	37	0.00	0.00
Positive	3	0.27	0.02
Grazing	101	0.12	0.01
Negative	13	-0.89	-0.08
No change	81	0.00	0.00
Positive	7	3.44	0.34
Centre Pivot	15	-0.01	-0.001
Negative	5	-0.11	-0.01
No change	8	0.01	0.001
Positive	2	0.15	0.01
Plantation	12	0.06	0.01
No change	10	0.00	0.00
Positive	2	0.35	0.04

4.2.1 Gum paddock tree cover

Table 4.6 shows the mean number of gum paddock trees per sample and the percentage change in paddock trees for samples containing gum paddock trees. Trends in gum paddock tree cover were negative for all three NRM regions, with rates of habitat loss being greatest in GHCMA (although note the small sample size for this region).

Overall, this loss was substantial, averaging around 9% across samples for the time period and 2.4% per annum. This suggests that, although there has been a slight net gain in tree cover overall, this has occurred mainly in remnant patches and paddock trees are still declining.

Table 4.6. Paddock tree density and number of paddock trees lost per sample, for those samples where loss occurred between the years 2003 to 2013 (GHCMA and NRSE) and 2004 to 2017 (WCMA).

	No. of Samples	Mean No. of Paddock Trees	Mean No. of Paddock Trees Lost/Gained	Mean % Paddock Tree Change	Mean % Paddock Tree Loss Per Annum
All gum woodlands	**53**	**81.2**	**-4.5**	**-9.2**	**-2.38**
GHCMA	2	23.0	-2.0	-35.5	-3.55
SENRM	21	104.5	-5.3	-8.7	-0.87
WCMA	30	68.7	-4.1	-7.8	-0.60

Tree dieback and mortality were important contributing factors to paddock tree loss. Tree dieback was noted in 32% of samples where gum paddock tree loss occurred and dead trees (together with trees showing symptoms of dieback) were noted at a further 13% of samples. By contrast, loss due to deliberate clearance accounted for only five samples. The cause of loss could not be determined for the remaining 50% of samples, which were probably associated with a combination of causes. For example, farmers may remove trees that are dead or dying in order to “tidy up” paddocks.

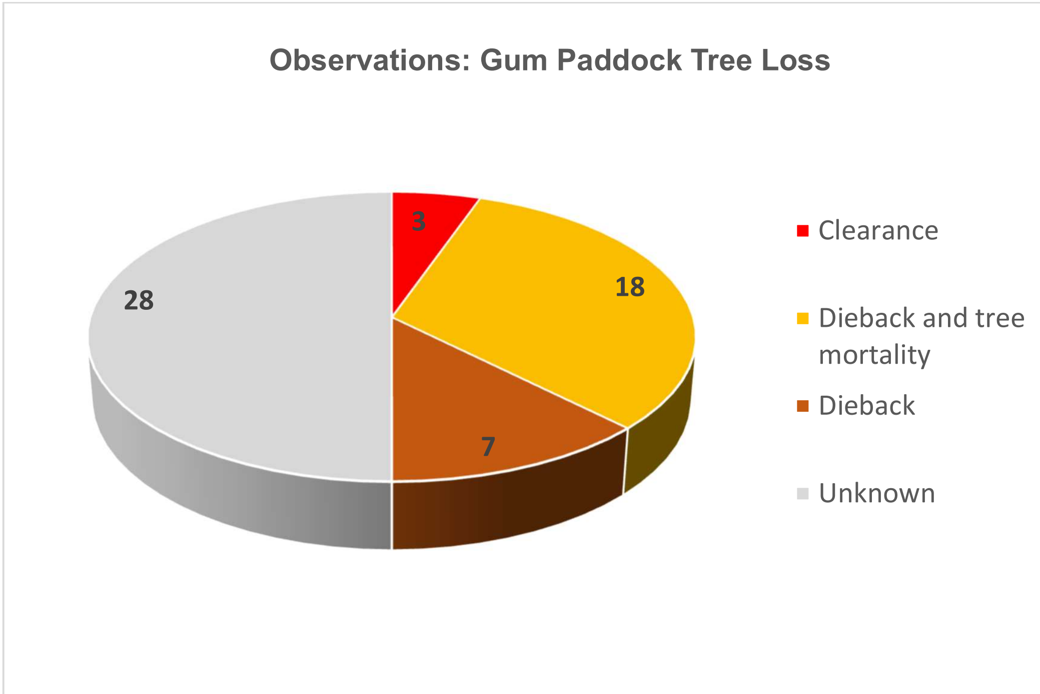

Figure 4.2. Observations associated with gum paddock tree loss, including clearance (where centre pivots were put in or significant areas of vegetation 18isappeared), dieback (where tree dieback and mortality evident, with at least some dead trees still standing) and unknown (where no clear cause of loss could be determined). Number of samples are given as labels.

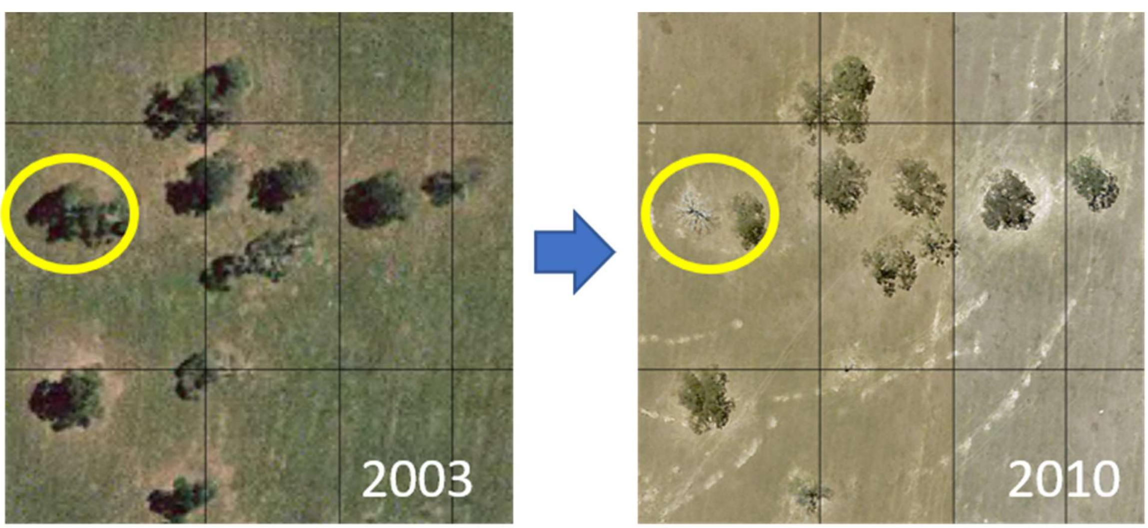

Figure 4.2. Example of tree dieback and mortality apparent from aerial imagery. The yellow circle points out an example of tree mortality, where the dead tree is still standing.

4.3 Rates of Habitat Change: Bulokes

4.3.1 Total Buloke vegetation cover

Table 4.4 shows rates of habitat loss/gain for Buloke remnants, stratified by NRM region and by land use type. Total Buloke cover showed a slight negative trend of 0.003 per annum. Rates of Buloke remnant cover change were slightly negative for the NR South East region, slightly positive for the WCMA region and slightly negative for the region as a whole. Among land use types, a negative trend was recorded for the cropping land use while a slight positive trend was recorded for the grazing land use. Positive gains in Buloke remnant cover were attributed to tree planting activities.

Table 4.4. Average percentage change in remnant Buloke cover, stratified consecutively by NRM region and by land use.

	Number of Samples	Average Buloke Remnant % Change	Average Buloke Remnant % Change Per Annum
NRSE	**7**	-0.26	-0.03
WCMA	**36**	0.01	0.001
Conservation	**2**	0.00	0.00
Cropping	**21**	-0.09	-0.01
Grazing	**17**	0.03	0.002
Centre Pivot	**3**	0.00	0.00
Grand Total	**43**	-0.03	-0.003

4.3.2 Buloke paddock tree cover

Table 4.5 shows rates of paddock tree loss for Buloke paddock trees, stratified by NRM region and land use type. Rate of Buloke paddock tree loss for the Natural Resources South East region was ten times higher than the rate for the Wimmera CMA, although note the small number of samples for the NRSE region. Overall rate of loss for Buloke paddock trees averaged at 0.2% per annum, which is considerably lower than that recorded by Maron et al. (2008) for the 1997 to 2004 study period.

Much of the loss in Buloke paddock tree cover was attributed to tree dieback. Dieback was noted in 14 out of 70 samples containing Buloke, with Buloke mortality apparent at seven of these samples (see Figure 4.3 for an example). By contrast, deliberate clearing of Bulokes for pivots was only noted at two of 70 samples. The reasons for the loss of the remaining 54 samples could not be determined, but was probably due in many cases to deliberate removal of dead or dying trees as a means of “tidying up” paddocks.

Table 4.5. Number of Buloke paddock trees lost per sample and average rates of loss, stratified consecutively by NRM region and by land use.

	No. of samples	Average No. of Bulokes	Average No. of Bulokes Lost	Average % of Bulokes Lost	Average % of Bulokes Lost Per Annum
NRSE	7	86.1	8.4	9.8	1.0
WCMA	63	38.7	2.0	5.2	0.1
Conservation	4	19.8	0.0	0.0	0.0
Cropping	31	44.2	4.4	9.9	0.7
Grazing	30	48.6	1.1	2.3	0.2
Centre Pivot	5	26.2	3.4	13.0	1.0
Grand Total	70	43.4	2.6	6.1	0.2

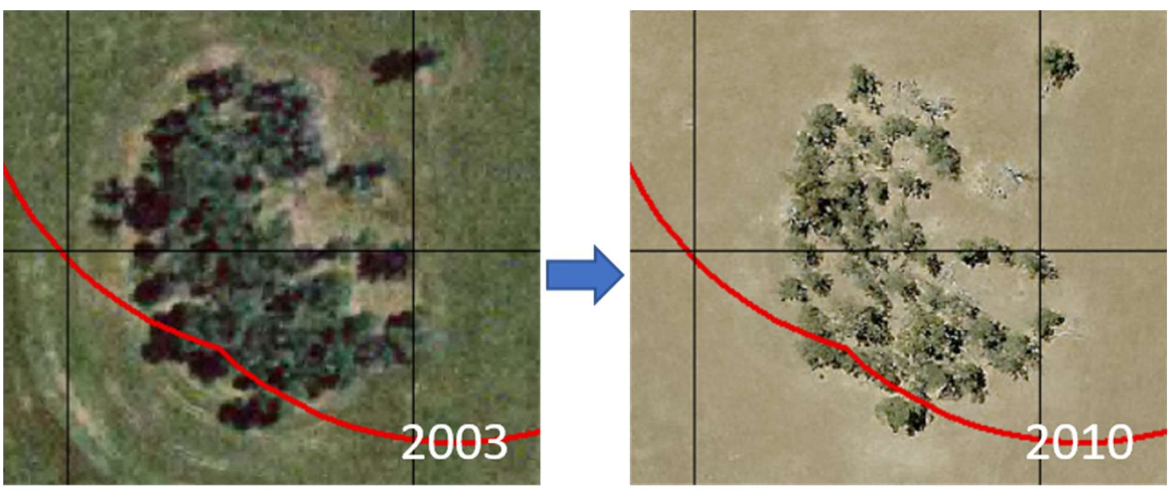

Figure 4.3. Example of Buloke dieback and mortality apparent from aerial imagery.

5. Discussion

5.1 Trends in Stringybark Feeding Habitat

An overall net gain was achieved across both stringybark (Average gain of 0.03% per annum). Within stringybark woodland types, however, *E. arenacea* showed a slight negative overall trend while *E. baxteri* showed a relatively strong positive trend. This is a substantial improvement on the habitat change figures reported by Maron et al. (2008) for the 1997 to 2004 study period, which averaged 0.04% loss per annum across both species of stringybark.

However, the paddock tree component of this cover is strongly declining at a much faster rate than trees in remnants for both *E. arenacea* (0.7% per annum) and *E. baxteri* (0.24% per annum) species of stringybark, meaning that it would take 143 years and 417 years to lose all tree cover, respectively. This is of concern because *E. arenacea* is the most limiting of the two species of stringybark in terms of habitat area and *E. arenacea* paddock trees produce the largest seed crops, producing around 27 times as much seed as trees in remnants (Maron et al. 2008). Therefore, the disproportionate loss of stringybark paddock trees is likely to represent a net loss in terms of food availability for the RtBC.

The strongest negative trends in stringybark cover were associated with the grazing land use, which is not surprising because stringybarks are highly vulnerable to cattle rubbing and browsing of bark by sheep and cattle, which eventually ringbarks the tree leading to dieback and tree death. Based on the author's observations, stringybark paddock trees exhibit less climate change-related dieback than some of the gum eucalypts such as Red Gums, although they show signs of climate-related dieback in stringybark remnants in some areas (Koch 2019a). The dieback that was reported in this study was likely more due to impacts by stock.

5.2 Trends in Gum Nesting Habitat

Trends for gum woodlands were similar to those of stringybark woodlands. An overall net gain was achieved in terms of total cover of gums (0.01% per annum), which was attributed to natural regeneration (from conservation practices such as stock exclusion) and revegetation efforts. Percentage change of total remnant cover was positive for the Glenelg Hopkins CMA and Natural Resources South East regions but was negative for the Wimmera CMA region.

This is a substantial improvement on figures reported by Maron et al. (2004) which showed a net per annum loss of 0.32% for the 1997 to 2004 period. The improvement seems to be mainly due to reduced losses from cropping and centre pivots. Only a few samples were associated with new centre pivot developments, suggesting that rates of land use change to centre pivots have slowed (centre pivot installations occurred mainly between 1993 and 2005, increasing steadily during this period; Maron and Fitzsimmons 2007). In addition, policies for reducing and offsetting paddock tree losses cleared for centre pivots have been strengthened since this time, meaning that clearance has been offset by tree planting. Conservation efforts in the region have no doubt played an important role as well.

However, net paddock tree cover loss for gums was substantial and occurred across all three NRM regions, averaging around 9% across all regions for the time period and 2.4% per annum. This suggests that, although there has been a slight net gain in tree cover overall, gains have occurred mainly in remnant patches and paddock trees are still declining. Furthermore, rates of paddock tree loss appear to be accelerating since the 1997 to 2004 period, in which Maron et al. (2004) reported negative change of 0.43% per annum cover for gum paddock trees.

If this rate of paddock tree loss continues, all paddock trees would be dead within 42 years. Paddock trees are likely to grow much faster and larger than trees in remnants due to reduced competition, therefore they are more likely to grow to a size sufficient to produce large hollows that provide suitable nesting habitat. Of course, dead paddock trees can still provide nesting habitat for the RtBC, but only if landholders retain them and only if they grow large enough and old enough to produce large hollows. Large standing dead trees that are likely to provide nesting habitat are protected in the range of the RtBC, but enforcement of this policy is difficult and deliberate removal of dead trees may still occur.

Perhaps the greatest concern is that rates of tree loss may accelerate due to climate change. Tree dieback and mortality were important contributing factors to paddock tree loss in this study period. Tree dieback was noted in 32% of samples where gum paddock tree loss occurred and dead trees (together with trees showing symptoms of dieback) were noted at a further 13% of samples. There are no estimates of tree dieback rates from previous habitat studies for the RtBC region, but from the author's observations over the past 20 years, it is obvious that dieback in paddock trees (particularly in Red Gums *E. camaldulensis*) has gone from being a rare event to a very widespread problem (Koch 2019a), occurring throughout the range of the RtBC. This dieback appears to affect both young and old trees and seems to affect trees on roadsides and paddocks more than trees in remnants, however dieback in remnants has been observed in some cased (Koch, pers. obsv.).

These observations are consistent with studies from other parts of Australia (eg. Matusick et al. 2018) and worldwide (Allen et al. 2010 and Anderegg et al. 2015), where climate change is being increasingly implicated as the underlying cause of tree dieback and tree mortality. Butt et al. (2011) implicated the steady increase in evapotranspiration deficit over recent decades as the cause of significant dieback of red gums (*Eucalyptus camaldulensis*) across the Murray–Darling Basin. Matusick et al. (2018) and others have confirmed that drought and heat stress are the drivers of mass tree mortality in Marri-Jarrah forests of southwestern WA. Impacts may result from acute, short term events such as heatwaves and/or longer-term events such as droughts which may produce legacy effects.

As previously mentioned, dead paddock trees can still provide suitable nesting habitat, but only if they are large enough and mature enough before they die. Based on the authors' observations, tree dieback due to drought stress and repeated insect infestations is affecting young trees as well as old trees, therefore many trees will die before they reach maturity and are large enough to produce suitable RtBC nesting hollows. These smaller dead trees will not be covered by current Environmental Significance Overlays and are unlikely to be retained by farmers in most cases.

Furthermore, Red Gums seem to be most severely affected by tree dieback (Koch, pers obsv.), with Blue/Yellow Gums also showing dieback in some cases. This is perhaps not surprising given that Red Gum woodlands are generally considered to be a groundwater-dependent ecosystem and groundwater levels are generally dropping in the region due to decreasing rainfall and a tendency to less spring rainfall (Hekmeijer et al 2008; Raleigh and Dixon 2005). Red Gums are generally larger than Blue/Yellow Gums and so tend to produce more large hollows suitable for nesting cockatoos (Koch pers. obsv.).

5.3 Trends in Buloke Feeding Habitat

Total Buloke cover showed a slight negative trend of 0.003 per annum (meaning it would take 3000 years at this rate to lose all remaining trees). This is a substantial improvement on the rate of 1.4% per annum reported by Maron et al. (2008). Rates of Buloke remnant cover change were slightly negative for the Natural Resources South East region and were slightly positive for the WCMA region.

Total cover of Buloke vegetation showed no change overall for the centre pivot land use, but Buloke paddock trees associated with this land use continue to decline at a rate of 1% per annum. These results suggest that native vegetation offset policies such as the Net Gain policy in Victoria are working as intended, at least from the point of view of replacing overstorey trees. No doubt the efforts of conservation groups such as Kowree Farm Tree Group have also contributed to these results. However, as has been noted by Maron et al. (2008), Bulokes are slow-growing and do not produce significant amounts of seed capsules for many years. Young Bulokes that have been recently planted may contribute to increased tree cover but are unlikely to provide useful feeding habitat (with a suitable density of cones) until they approach maturity. Furthermore, Bulokes regenerating on roadsides and in remnants often occur at high densities and tend to sucker from existing trees. The suckering stems don't generally produce seed crops and the high densities often mean that individual stems are small and spindly. Therefore, the disproportionate loss of mature Buloke paddock trees is likely to represent a net loss in terms of food availability for the RtBC.

Much of the loss in Buloke paddock tree cover was attributed to tree dieback. Dieback was noted in 14 out of 70 samples containing Buloke, with Buloke mortality apparent at seven of these samples. By contrast, deliberate clearing of Bulokes for pivots was only noted at two of 70 samples. These results suggest that tree dieback is at least as important as clearance for the Buloke food resource. Bulokes are known for their longevity and resilience as paddock trees and this is the first time to the author's knowledge that extensive dieback has been reported for this species. It seems likely therefore that Bulokes, like many other tree species, are susceptible to climate change related heat and drought stress (Allen et al. 2010, Anderegg et al. 2015, Matusick et al. 2018). Unfortunately, this means that we may see accelerating rates of Buloke loss over the coming decades, because drought conditions are predicted to worsen during this period even if strong action on climate change is taken.

Maron et al. (2008) undertook preliminary climatic envelope modelling of Buloke to understand potential changes to Buloke distribution and related impacts on the RtBC. The results showed the expected general southwards shift of the current climatic envelope. However, because Buloke occurs across a wide geographic distribution extending to north Queensland, an increase in temperature alone was not predicted to have a negative effect on Buloke distribution within the RtBC range. However, a decrease in rainfall was predicted to be an important limiting factor. With a minor decrease of rainfall (-10%) suitable climate conditions were predicted over the entire RtBC range but, with a decrease of 20% rainfall, would result in unsuitable climatic conditions for the entire present-day population of Buloke within the RtBC range. The authors acknowledge the limitations of these models (modelling techniques and spatial data have also advanced rapidly since this time).

One of the key limitations is that tree mortality and dieback are often driven by acute, short-term events such as heatwaves or droughts and average changes in climate conditions often mask these extremes. Another limitation is that the models do not account for biotic interactions which can play an important role in accelerating tree mortality, as acknowledged by Maron et al. (2008). For example, trees are weakened by repeated episodes of drought and thus have fewer resources available to defend against insect attacks. Conversely, insect populations may increase exponentially in response to more favourable conditions brought about by a warming climate and reduced predation due to predator satiation (typically insect predators are woodland birds which are generally declining). This leads to a situation where insect pests are much more numerous and tree defences against insect attacks are weakened, with the result of widespread and extensive tree defoliation. Trees can recover from occasional episodes of such events, but not repeated episodes.

5.5 Conclusion and Recommendations for Future Research

The results of this study indicate substantial improvement in rates of habitat loss and indeed some habitat types (Brown Stringybark and Gum eucalypts) are currently increasing in terms of total tree cover. On the other hand, the paddock tree component of this cover is strongly declining across all major habitat types, due in large part to tree dieback and tree mortality. As climatic drying progresses, this trend is likely to worsen substantially (Butt and McAlpine, 2013), impacting on both feeding (especially Buloke) and nesting (especially Red Gum) resources. It seems likely that Red Gums will be completely lost from the region within a decade or two, but it is recommended that the habitat loss assessment (with particular attention to tree dieback and mortality) is repeated when new aerial imagery becomes available, in order to confirm that tree dieback and mortality is indeed accelerating. An additional important knowledge gap relates to the distribution of the different eucalypt species that make up the "gum" nesting habitat, which includes Red Gums *E. camaldulensis,* Yellow/Blue Gums *E. leucoxylon,* Pink/Hill Gums *E. fasciculosa,* Manna Gums *E. viminalis* and Grey Box *E. macrocarpa* and other eucalypts. The distribution of these species could be mapped by obtaining records from existing databases (ideally supplemented with records obtained through field work) and then running species distribution modelling software such as Maxent to develop predictive distribution models. This would allow us to determine the rates at which key eucalypt species are declining and more accurately predict where hollow availability is likely to become most limiting in the future.

However, we now need to give much greater attention to planning for a rapidly warming climate. Given the scale of dieback and mortality already occurring and the amount of climate change still to come, we need to consider planting different species of eucalypts in the case of the nesting resource in order to ensure a future supply of hollows and the other important resources and values that paddock trees provide for both conservation, agriculture and landscape amenity. Addressing this problem is a significant challenge, but replanting and revegetation programs are already being undertaken by conservation groups throughout the RtBC range and the results of this study confirm that conservation efforts have made a substantial difference.

In the case of the stringybark food resource, climate change-related dieback seems to be occurring mainly within remnant woodlands (Koch 2019a) and does not seem to be affecting stringybark paddock trees (stringybark paddock tree dieback perceived in this study appeared to be mainly associated with grazing impacts) to the same extent. This confirms the need to continue stringybark replanting efforts on private land, albeit with greater attention to plant provenance (collecting seed from stringybark woodlands occurring in the hottest and driest parts of the range).

In the case of Buloke, it may be feasible to replant with seed provenances from a hotter and drier climate, because the species is widely distributed. Further analysis is warranted to determine suitable bioclimatic regions for seed collection. Many species exhibit considerable inter-specific variation among provenances including morphological adaptations to local climate conditions and the capacity to regulate osmotic potential in response to water deficits (Merchant et al. 2007). Common morphological adaptations are associated with leaf shape (typically narrower in drier climates) and leaf stomatal density (typically denser in drier climates; Guerin et al. 2018). Both species of stringybark span a wide climatic gradient and are thus likely to exhibit these genetic variations. One recommended action therefore is to trial the use of seed stock from drier and hotter provenance areas in stringybark replanting programs, since this would likely confer improved drought resilience.

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