The birds of retained vegetation corridors: A pre- and post-logging comparison in dry sclerophyll forest in Tasmania

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Abstract

Birds were surveyed before logging and approximately 10 years after logging in dry sclerophyll forest in Tasmania at a site where retained vegetation corridors in the form of wildlife habitat strips (WHS) of 100 m width and streamside reserves (SR) of 40 m width had been established. Similar surveys were done in nearby extensive forest. After logging, both categories of retained vegetation corridors had lower bird species richness (per count) and abundance that was not observed in the extensive forest. Compositional change was less obvious between the survey periods. Inferential statistics were not performed on individual species' abundances, but absolute counts of some species are discussed. Logging-sensitive species, such as the golden whistler (*Pachycephala pectoralis*), satin flycatcher (*Myiagra cyanoleuca*) and the hollow-nesting green rosella (*Platycercus caledonicus*) persisted in corridors after logging. However, a small number of species, including the dusky robin (*Melanodryas vittata*), yellow wattlebird (*Anthochaera paradoxa*) and strong-billed honeyeater (*Melithreptus flavicollis*) declined in or disappeared from both logged areas and wildlife habitat strips. Retained vegetation corridors maintain a slightly depauperate version of the pre-logged avifauna, and differences in abundances might be expected to lessen as the surrounding matrix of native regeneration matures. Both wildlife habitat strips and streamside reserves are considered to have conservation value for avifauna in the study area, although we recommend more attention be paid to the provision and value assessment of wildlife habitat strips in upper slope habitat.
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1. Introduction

The effects of forestry practices on forest birds have been well documented both in Tasmania (Dickinson et al., 1986; Taylor et al., 1997) and elsewhere (Thompson et al., 1992; Baker and Lacki, 1997; Kavanagh and Stanton, 2003). In an effort to mitigate negative effects, conservation-oriented forest management practices have been implemented. One of these is the retention of corridors of mature forest within the production forest landscape. In Tasmania, such corridors are known as wildlife habitat strips (WHS). These are 100 m wide, include both riparian and non-riparian habitat, and link large areas of unlogged forest

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according to Forest Practices Code (Forest Practices Board, 2000) prescriptions. This code also prescribes streamside reserves (SR) of at least 40 m width (depending on stream catchment area) to maintain water quality, and these reserves also form linear features of mature forest. Taylor (1992) considered that within the production forest landscape, the major value of wildlife habitat strips would be as retained mature forest habitat rather than as movement corridors.

The advantages and disadvantages of corridors have been discussed in several reviews, including Wilson and Lindenmayer (1995) and Bennett (1999). Within production forest landscapes, corridors and/or retained natural habitat are beneficial to birds (Triquet et al., 1990; Whitaker and Montevecchi, 1999; Lindenmayer et al., 2002), with wider corridors providing better conservation results (Spackman and Hughes, 1995; Hagar, 1999). A study of Tasmanian plantation landscapes found that the effectiveness of wildlife habitat strips in maintaining a healthy avifauna declined in strips outside of riparian zones (MacDonald et al., 2002).

When the prescription for wildlife habitat strips was introduced in 1987 it was not known whether a width of 100 m would be adequate for conservation purposes. As a result of the Regional Forest Agreement (Commonwealth of Australia and State of Tasmania, 1997) and the Intensification of Forest Management program arising from it, several studies have been conducted to assess the value of wildlife habitat strips for fauna and flora conservation. A long-term ecological monitoring site was established in dry sclerophyll forest, and pre-logging data were collected for a range of taxa (e.g. Cale, 1994; McQuillan et al., 1998). The intention is to evaluate the value of retained vegetation corridors (wildlife habitat strips and streamside reserves) over the approximately 90-year duration of the native forest regeneration and harvest cycle (Whiteley, 1999).

Dry sclerophyll forests are typically less than 40 m tall, and are dominated by a range of eucalypt species, notably peppermints (e.g. *Eucalyptus amygdalina*, *E. pulchella*) or ashes (e.g. *E. obliqua*, *E. sieberi*) (Forest Practices Board, 2004). Fire interval is typically 5–25 years, leading to uneven-aged stands, and the understorey is dominated by shrubs, sedges, grasses, herbs or bracken. Small patches of more mesic vegetation

occur within dry sclerophyll forest, particularly along drainage lines. Dry sclerophyll forest in Tasmania covers over 1.5 million ha, or 46% of forested area in the state (Forest Practices Board, 2004). It is thus is a major resource for a forest industry that is economically important, yet also controversial at times, due to the widely claimed conservation and aesthetic values of these forests.

This study presents the results of the first post-logging bird surveys and, through comparisons among treatments and with pre-logging survey data, seeks to determine: (a) how representative the avifauna of the retained vegetation corridors is of the avifauna of extensive native forest; (b) the influence of logging in adjacent areas on the avifauna of the reserved areas.

2. Methods

2.1. Study area

The study was carried out on State Forest near Pioneer in the north-east of Tasmania, Australia (Fig. 1), at 41°06′S 148°03′E (UTM: 55GEQ 88178 49531). The treatment area has a gentle southerly aspect, with several gullies (carrying surface water for only part of the year) running into Old Chum Dam. The control area is located 2 km to the south-west, and has a very slightly northerly aspect. Altitude ranges from 100 to 250 m a.s.l. The bedrock is Devonian/Carboniferous granite, with soils comprised of gravel and coarse-grained sand on ridges and slopes and finer grains in gullies (Duncan and Brown, 1995). Climate is temperate and strongly seasonal: average summer maximum temperatures are 23.1 °C in February, and average winter maximum temperatures are 12.0 °C in July, while the average annual rainfall of 1027 mm falls predominantly in the winter months (Bureau of Meteorology records from Scottsdale, 45 km west of the study site).

Prior to logging, Duncan and Brown (1995) classified the vegetation of the study site into twelve floristic groups, not all of which were present in bird survey plots. While vegetation is dynamic, the same floristic groups were considered to remain present at unlogged sites at the time of the post-logging survey. For the purposes of the present study, unlogged vegetation can be divided into three broad types: (1)
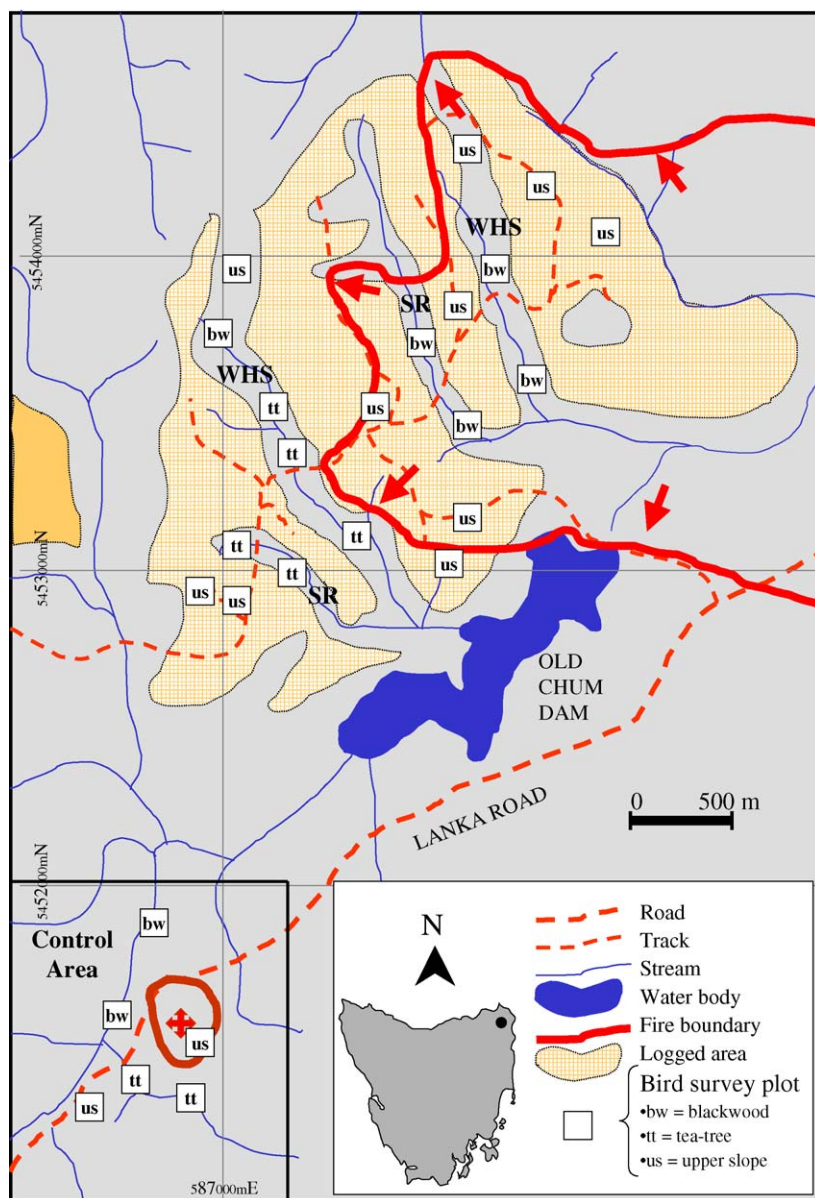
Fig. 1. Location of study area in Tasmania and location of survey plots within the study area. Study area centroid is 41°06′S 148°03′E. (Note that survey plot symbols represent quadrat locations and are not to scale.)

blackwood (*Acacia melanoxylon*) gullies, where the low (10–20 m) subcanopy of blackwood and *Atherosperma moschatum*, is overshadowed by a higher (20–35 m) canopy of *E. obliqua* on the surrounding slopes and where the dense understorey consists mostly of *Pomaderris apetala*, *Olearia argophylla*,

Dicksonia antarctica and *Zieria arborescens*; (2) tea-tree (*Melaleuca squarrosa*) gullies, with an overstorey (20–30 m tall) of predominantly *E. amygdalina* and a dense low (<10 m) understorey of tea-tree and *Leptospermum scoparium*; (3) upper slope (dry sclerophyll forest) vegetation, dominated by (20–30 m

tall) *E. amygdalina* and/or *E. obliqua* and with a variable understorey that was more open than that of the gullies.

Current silvicultural practices in dry sclerophyll forests operate under the banner of 'uneven aged treatment' (McCormick and Cunningham, 1989), i.e. polycyclic silviculture. Areas in which there is a cohort of saplings in addition to the mature trees that contribute to the canopy are generally managed as "advanced growth retention" (Forestry Commission, 1994), in which the mature trees are removed while leaving the saplings to mature (although a scattering of non-commercial trees may be left). In time, a further generation of saplings is likely to arise as the 'released' saplings mature, and the cycle can be repeated. In other areas, where stocking of saplings is poor (e.g. due to a more closed canopy, or recent wildfire), a scattering of mature trees is left at harvest to act as seed trees. The newly opened up forest (which may additionally be scarified or burnt; Neyland, 2000) is a receptive seedbed so a new generation of seedlings/saplings soon emerges. At this stage, the remaining seed trees will be felled to release the saplings and allow them to mature.

Coupes are generally 50 ha in extent or less, but can be up to 100 ha. A single coupe may have areas of each of the above treatments. In either silvicultural system, the Forest Practices Code (Forest Practices Board, 2000) prescribes the retention within the coupe of Wildlife Habitat Clumps comprising patches of mature forest containing "habitat" trees. Additionally, many areas remain off-limits for logging due to steep or rocky terrain, sensitive positions in the landscape, requirements for protection of riparian areas, wildlife habitat strips, etc.

Partial logging, using advanced growth retention (Forestry Commission, 1994) occurred at the treatment site in three stages between August 1990 and January 1993. Vegetation was retained in two WHS (100 m wide) and two SR (40 m wide), one of each in blackwood and tea-tree gully habitat (Fig. 1). The WHS contained more non-riparian vegetation than the SR, and both WHS also continued (longitudinally) up slope to incorporate dry sclerophyll forest habitat. In September 1991 the logged area was subjected to a top disposal burn (burning of logging residue to promote regeneration). A wildfire in October 1991 burned those parts that had been logged at that time, as well as some

of the eastern (blackwood) wildlife habitat strip and streamside reserve (Fig. 1). Impact on the gully vegetation itself was minimal, but the fire did affect the edges of the retained vegetation corridors. A spot fire in the period 1991–1993 burned a small part of the control area (Fig. 1).

Following logging, eucalypt regeneration formed a fourth vegetation type on logged upper slopes. In 2002, *E. obliqua* and/or *E. amygdalina* formed a low canopy (approximately 5 m), although a few mature trees remained to form an intermittent higher canopy of up to 30 m. Undergrowth was variable, but generally relatively dense, with *Zieria arborescens* and *Acacia terminalis* common in some places, *Leptospermum scoparium* and *Pteridium esculentum* in others.

The study site sits within an extensively forested region (the lowlands of NE Tasmania) in which vegetation clearance for non-forest use has been relatively minor, although much of this forest has been selectively logged over the past 100 years while more intensive forestry (including conversion to plantation) has affected a small proportion of the area in recent decades.

2.2. Bird surveys

Twenty-six 0.25 ha (50 m × 50 m) quadrats were located within the treatment and control areas (Fig. 1). Seven quadrats were located in blackwood gullies (three in WHS, two in SR, and two in the control area); seven were located in tea-tree gullies (three in WHS, two in SR, and two in the control area); 12 quadrats were located in upper slope habitat (two in WHS, two in the control area, and eight in areas to be logged). Quadrats were not considered to be independent replicates, and all surveys within a treatment (or control area) were pooled for analysis in the GLM described below.

Birds were surveyed using the area search method, a valid method for surveying logged and unlogged areas (Craig and Roberts, 2001). Each survey lasted a minimum of 4 min. A maximum time period was not specified, as this can lead to surveys of differing completeness, depending on vegetation type and density of birds (Watson, 2003). This was an issue in the current study, which surveyed several vegetation types. In practice no survey took longer than 10 min.

Quadrats were surveyed twice per month. Pre-logging surveys were done from June 1989 to May 1990 by a single observer (P. Cale, not the author) and post-logging surveys were done from January to December 2002 by the senior author. Surveys were conducted within 3 h of sunrise, recording species and number of individuals. Birds that were not identified (seven in total) were used for analysis of bird abundance, but not for species richness or avifaunal composition.

2.3. Analysis

To examine the effects of logging and the establishment of WHS and SR on bird species richness and abundance we fit generalized linear models (GLM), considering a Poisson distribution, log link function and a first-order autoregressive correlation process as follows:

response = constant + time + treatment

$\qquad$ + vegetation + time × treatment + time

$\qquad$ × vegetation + error.

"time" refers to the pre- and post-logging survey periods. Additional and higher order interactions were avoided due to imbalance in the data.

The estimates were obtained using a generalized estimating equations (GEE) approach, implemented by the function gee in Splus. The expected value and variance for the model are

$$\log(E[\boldsymbol{Y}_i]) = \boldsymbol{X}_i'\boldsymbol{\beta}, \qquad \mathrm{var}(\boldsymbol{Y}_i) = \phi E[\boldsymbol{Y}_i]$$

where $\boldsymbol{Y}_i$ refers to the vector of observations for population i, $\mathbf{X}_i$ to the design matrix and $\boldsymbol{\beta}$ to the vector of linear model coefficients. The dispersion parameter ϕ was estimated from the data rather than using a value of 1, accounting for overdispersion.

2.4. Species composition

Abundances of individual species could not be modelled using the above procedure. Despite issues of lack of independence of quadrats within the same landscape features, ordination of pre-logged quadrats was used as an exploratory method to check expectations of differences in species composition according to vegetation type. This was done using non-metric multidimensional scaling (NMS) with the autopilot option in PC-ORD (McCune and Mefford, 1999) using frequency data. This procedure automatically determines dimensionality by comparing final stress values. In addition, the multi-responses permutation procedure (MRPP) available in PC-ORD (McCune and Mefford, 1999) was used to test whether there were significant differences in species composition among vegetation types. This method tests for between-group separation, and also measures a within-group homogeneity statistic (A), where a value of 0 indicates homogeneity expected by chance alone (McCune and Mefford, 1999).

Differences in species composition between pairs of treatments and between the two survey periods were examined by calculation of the Morisita–Horn similarity index using the EstimateS program (Wolda, 1981; Colwell, 2005). The Morisita–Horn similarity index is not strongly influenced by overall species richness or sample size, and also takes into account the relative proportion of species within samples (Magurran, 1988). Values range from 0 (no species in common) to 1 (complete agreement in species composition and abundance). All samples within a treatment and survey period were pooled to provide the raw data for calculation.

3. Results

Thirty-nine species were recorded during the surveys, 30 pre-logging, 38 post-logging, and 29 common to both (Appendix 1). Total species richness varied from 14 in pre- and post-logging tea-tree streamside reserves to 24 in pre-logging upper slope habitats (intended for logging). Total species richness did not increase or decrease consistently by treatment or in control areas, ranging from an increase of seven species in tea-tree gully control areas, to a decrease of two species in blackwood WHS, upper slope WHS and upper slope control areas (Appendix 1).

Small insectivorous birds such as the brown thornbill (*Acanthiza pusilla*), Tasmanian thornbill (*A. ewingii*), Tasmanian scrubwren (*Sericornis humilis*), grey fantail (*Rhipidura fuliginosa*), superb fairy-wren (*Malurus cyaneus*) and several honeyeater

(Meliphagidae) species, which combine insectivory and nectarivory, formed the bulk of the avifauna during both survey times. There were seasonal differences, with some species arriving as summer migrants, and others showing less predictable movements into and out of the study area. The pallid cuckoo (*Cuculus pallidus*) was the only species recorded solely in the pre-logging survey period, two individuals being recorded in a control quadrat. Nine species were only recorded from the post-logging survey period, most of them in very small numbers and generally in control as well as in treatment quadrats (Appendix 1). Of note were: the presence in moderate numbers (in both logged and control quadrats) of the dusky woodswallow (*Artamus cyanopterus*), a hawking bird typical of open woodland; the presence in moderate numbers (in WHS and logged quadrats) of the common bronzewing (*Phaps chalcoptera*), a ground-feeding seed-eater; the presence, albeit in very small numbers, of two large omnivores typical of open woodland, the grey butcherbird (*Cracticus torquatus*) and grey currawong (*Strepera versicolor*), in WHS and logged quadrats.

3.1. Bird species richness and abundance

The two response variables showed similar patterns in the GLM (Table 1). There was no significant first-order effect of time (post-logging against pre-logging survey periods), indicating that there was not an across-the-board difference in bird abundance and species richness between survey periods. Nor was there a significant first-order effect of treatment. Thus, the observed differences in bird species richness and abundance were not a result of fundamental differences at the survey sites regardless of the treatment applied. There was a significant first-order vegetation class effect, with lower species richness and abundance in tea-tree gullies and upper slope habitat relative to blackwood gullies. The most interesting results are the significant interaction effects for time and treatment in SR and WHS. Both SR and WHS showed declines in bird abundance and species richness of similar magnitude (approximately one-third) relative to the control areas in the second survey period. By contrast, there was no significant interaction effect for time and treatment for logged areas;

Table 1
Model term effects, their standard errors, and effects expressed as percentages

Terms (reference level)	Effect	Abundance[a]			Species richness[a]		
		Value	S.E.	%[b]	Value	S.E.	%[b]
Constant (n/a)		1.377[**]	0.138		0.783[**]	0.142	
Time[c] (pre-logged)	Post-logged	0.001	0.092	100.1	0.143	115.37	0.082
Treatment (control)	Logged	−0.057	0.134	94.46	−0.105	0.157	90.03
	SR	−0.116	0.214	89.05	−0.059	0.171	94.27
	WHS	0.028	0.153	102.84	−0.034	0.154	96.66
Vegetation (blackwood)	Tea-tree	−0.682[**]	0.172	50.56	−0.552[**]	0.145	57.58
	Upper slope	−0.763[**]	0.110	46.63	−0.854[**]	0.148	42.57
Time × treatment (time × control)	Time × logged	0.000	0.315	100.00	0.123	0.302	113.09
	Time × SR	−0.418[*]	0.183	65.84	−0.513[**]	0.164	59.87
	Time × WHS	−0.466[**]	0.163	62.75	−0.380[**]	0.144	68.39
Time × vegetation (time × blackwood)	Time × tea-tree	0.250	0.198	128.40	0.134	0.159	114.34
	Time × upper slope	−0.117	0.270	88.96	−0.231	0.259	79.37
Autocorrelation		0.020	0.032		0.049	0.034	
Dispersion parameter		3.707			1.548		

[a] Coefficients are in a logarithmic scale. Effects can be translated to the measurement scale using antilogarithm. For example, the effect of the "logged" treatment $e^{-0.057} = 0.945$ or a 5.5% reduction from the reference level.

[b] % is the effect expressed as a percentage compared to the reference, where the reference = 100%, so that >100% indicates a positive effect and <100% indicates a negative effect.

[c] Time refers to pre- and post-logging survey periods.

[*] Statistical significance is specified at the <0.05 level.

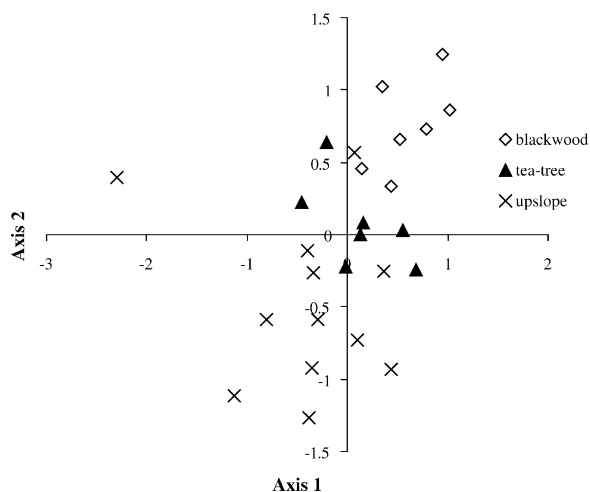[**] Statistical significance is specified at the <0.01 level.

Fig. 2. NMS Ordination of pre-logged quadrats by bird frequency data (final stress = 17.86).

relative to the control areas abundance barely differed, while species richness increased slightly.

3.2. Species composition

Examination of the ordination of pre-logged quadrats indicated that the three vegetation types supported different avifaunas (Fig. 2). This was supported by MRPP, which found that within-group homogeneity was quite high ($A = 0.2184$) and that between-group distances were significantly different ($p < 0.001$) from those which might have been expected by chance. In addition, the GLM indicated a main effect of vegetation on bird abundance and on bird species richness. Therefore, examination of bird community differences pre- and post-logging was separated by vegetation type. Tables of Morisita–Horn similarity index values (Tables 2–4) present comparisons of a treatment between the two survey periods and

comparisons of all pairs of treatments within a survey period (but not, for example, pre-logging WHS with post-logging SR).

3.3. Blackwood gullies

The least similar avifaunas were those in control areas and the streamside reserve prior to logging, and in the streamside reserves between sample periods (Table 2). However, the streamside reserve was much more similar to control areas post-logging, suggesting that the changes in the avifauna in the streamside reserve had been in a direction towards the avifauna of the control areas. The avifauna of the wildlife habitat strip was relatively similar between survey periods, as was that of the control area: these two areas were also similar to each during both survey periods (Table 1). While counts of individual species were not sufficient to perform inferential statistics, some changes are worthy of note. These include reductions of over 50% of counts of the Tasmanian scrubwren and Tasmanian thornbill in retained vegetation corridors, while those in control areas remained steady (Appendix 1). Both of these species are typical of moist habitats in the area, the scrubwren feeding on invertebrates in leaf litter and soil, the thornbill leaf gleaning largely in the mid-canopy layer. Counts of the black-headed honeyeater (*Melithreptus affinis*), increased across all treatments and in control areas, while those of its congeneric, the strong-billed honeyeater (*M. flavicollis*), declined in retained vegetation, although from very small numbers (Appendix 1).

3.4. Tea-tree gullies

Similarity index values for all pairings were generally lower in tea-tree gullies than in blackwood

Table 2
Morisita–Horn similarity index between pairs of treatments in the same sample period and for the same treatments between the two sample periods for blackwood gully vegetation

	Pre-logged SR	Pre-logged Con	Post-logged WHS	Post-logged SR	Post-logged Con
Pre-logged WHS	0.931	0.822	0.943		
Pre-logged SR		0.675		0.587	
Pre-logged Con					0.902
Post-logged WHS				0.801	0.834
Post-logged SR					0.930

WHS: wildlife habitat strip, SR: streamside reserve, Con: control.

Table 3
Morisita–Horn similarity index between pairs of treatments in the same sample period and for the same treatments between the two sample periods for tea-tree gully vegetation

	Pre-logged SR	Pre-logged Con	Post-logged WHS	Post-logged SR	Post-logged Con
Pre-logged WHS	0.844	0.681	0.904		
Pre-logged SR		0.732		0.469	
Pre-logged Con					0.756
Post-logged WHS				0.529	0.773
Post-logged SR					0.711

WHS: wildlife habitat strip, SR: streamside reserve, Con: control.

Table 4
Morisita–Horn similarity index between pairs of treatments in the same sample period and for the same treatments between the two sample periods for upper slope vegetation

	Pre-logged LS	Pre-logged Con	Post-logged WHS	Post-logged LS	Post-logged Con
Pre-logged WHS	0.967	0.888	0.669		
Pre-logged LS		0.911		0.847	
Pre-logged Con					0.576
Post-logged WHS				0.642	0.381
Post-logged LS					0.631

WHS: wildlife habitat strip, LS: logged sites, Con: control.

(Table 3). Once again the least similar pair of samples was the streamside reserve pre- and post-logging. However, differences between the control area and both retained vegetation corridors altered very little between sample periods, indicating that species turnover and changes in species' abundances in the streamside reserve were from within the suite of species found within that habitat in the absence of logging treatments. Among all treatments and in controls, counts of the Tasmanian scrubwren declined, while those of the black-headed honeyeater increased (Appendix 1). In the streamside reserve, counts of the crescent honeyeater (*Phylidonyris pyrrhoptera*), a species that combines nectarivory and insectivory and that feeds on *Banksia marginata* nectar in the winter, reduced from 23 to 1 (Appendix 1).

3.5. Upper slopes

The avifaunas of the treatment areas and the controls were all very similar before logging (Table 4). This correspondence did not persist after logging, and the control area and wildlife habitat strips were particularly dissimilar in this sample period. There was lower similarity between survey periods in the wildlife habitat strips and the control area than was measured in the other vegetation types. The logged areas were much more similar between sample periods than were either the WHS or the controls (Table 4).

Many of the counts of individual species were very small in both sample periods. Worth noting are the reduced counts post-logging of the brown thornbill, strong-billed honeyeater, yellow wattlebird (*Anthochaera paradoxa*) and dusky robin (*Melanodryas vittata*) in wildlife habitat strips and logged areas, and reduced counts of the black-faced cuckoo-shrike (*Coracina novaehollandiae*), golden whistler (*Pachycephala pectoralis*) and scarlet robin (*Petroica multicolor*) in logged areas (Appendix 1). Once again, counts of the black-headed honeyeater increased across all treatments and in controls in the post-logging survey period.

4. Discussion

There were identifiable changes in the avifauna of the study area between the survey periods, and in particular there are significant reductions in bird abundance and species richness in the retained vegetation corridors that were not observed in the control areas. Some of these may be attributed to

factors other than the treatment effects of logging. However, we do not consider that observer bias, climatic variation, flowering variation and other possible factors unrelated to treatment effects are likely to be the major causes of the changes observed in the retained vegetation corridors. The effects of the wildfire in 1991 that burned parts of one wildlife habitat strip and one streamside reserve are more difficult to assess, although the impact on gully vegetation was minimal. Bird numbers in eucalypt forest are depressed immediately after wildfire (Recher et al., 1985; Wooller and Calver, 1988), but composition and abundance have been shown to be similar to pre-fire levels within 5 years (Loyn, 1997; Abbott et al., 2003), well within the time scale of the present study. Compositional changes in the retained vegetation corridors that were affected by fire were not consistently greater than those in corridors that were not. We consider that more than 10 years after the 1991 fire the residual effects on bird populations are minimal. In the Australian landscape, which is largely shaped by fire, most avian communities recover rapidly following single fires, regardless of intensity, and the greatest effect on bird communities is likely to come from long-term changes in fire frequency, especially increases (Woinarski and Recher, 1997). If this proves to be the case in the study area over the duration of this long-term study, then fire effects may have to be considered more seriously in future analyses and in future management decisions related to the provision of wildlife habitat strips and streamside reserves. However, increased likelihood of fire disturbance might be considered a normal result of the creation of retained vegetation corridors and adjacent logging, and therefore akin to a treatment effect.

The use of species richness and abundance to infer changes in the avifauna is open to criticism, as a community exhibiting large changes in composition and species' abundances may return similar values to a stable community. However, we measured changes in community composition with the Morisita–Horn similarity index, and we discuss below some of the changes in individual species counts from raw data. In the absence of a large shift in species composition we feel that these response variables accurately reflect the differences between the two survey periods. It may have been desirable to collect demographic data from the study area, as population dynamics have been shown to change following disturbance such as fire (Baker et al., 1997). However, the intensity of fieldwork that this would have required was outside the remit of the present study.

The results of the present study suggest that wildlife habitat strips and streamside reserves in the study area do not maintain an avifauna equivalent to that of extensive native forest 10 years after logging. There were significant reductions in bird abundance and species richness in wildlife habitat strips and streamside reserves that did not occur in the control sites. The reduction in species richness (per count) appears to be driven by the reduction in overall abundance, as total species richness did not show a downward trend in any treatment. However, the evidence from measurements of compositional similarity suggests that the reduction in numbers in retained vegetation corridors is not concomitant with large changes in avifaunal composition, at least in riparian vegetation. In blackwood and tea-tree streamside reserves, where composition was least similar pre- and post-logging, it was still relatively similar to that of control areas post-logging, suggesting that the temporal differences were within the natural avifaunal variation. In a radically different forest type (Douglas-fir forest) in North America, Pearson and Manuwal (2001) found considerable species turnover in riparian buffer strips (which were slightly narrower than the streamside reserves in the present study) but little change in bird abundance. Differences in responses between forest types are to be expected, although the results of the current study are likely to be relevant not only to Tasmania but to dry sclerophyll forest elsewhere in Australia. In the south-east of the mainland, the vegetation and avifauna are broadly similar to those of Tasmania; however, bird species comprising the relatively depauperate island avifauna may have evolved more generalised habitat requirements, and may therefore be more resilient to the effects of logging. The present study suggests that both streamside reserves and wildlife habitat strips are able to maintain similar suites of species to those present before logging, but these can be at significantly reduced abundances.

While the main purpose of the present study was not to examine changes in the avifauna in logged areas per se, some of the results from logged areas are

worthy of discussion, particularly in the context of the ability of wildlife habitat strips in upper slope areas to conserve bird populations. There was no significant interaction between logging and survey period in the GLM; indeed, the model suggests that abundance was almost exactly the same, and species richness slightly higher post-logging. Community composition in logged areas (which, of course, were not logged at the time of the first survey period) was more similar between survey periods than it was in either wildlife habitat strips or the control area. And post-logging community composition was more similar between the control area and logged sites than between the control area and wildlife habitat strips. This lack of a logging effect, while both streamside reserves and wildlife habitat strips experienced a decline in bird abundance and species richness, might at first glance suggest that birds would be better off without retained vegetation corridors. However, we suggest that two conclusions can be drawn from these results: firstly that 10-year-old regenerated eucalypt forest (especially when mature trees are present) does indeed provide suitable habitat for many bird species present before logging, notably foliage gleaners; secondly that species of conservation concern are likely to be present only in small numbers at any time, and thus contribute less to abundance-based similarity indices (Magurran, 1988). It is also probable that the presence of retained vegetation corridors ameliorates the effects of logging in the landscape generally, and contributes to the retention of birds in logged areas.

The new growth of regenerating eucalypts provides foraging opportunities for leaf gleaners such as the striated pardalote (*Pardalotus striatus*) and black-headed honeyeater, while the flush of *Acacia terminalis* in logged areas provides nectar resources for species such as the eastern spinebill (*Acanthorhynchus tenuirostris*). However, the counts of several species did decline in logged areas. Of less concern are those species, such as the scarlet robin, which were not recorded from logged areas, but which persisted in wildlife habitat strips. More emphasis should be placed on species whose numbers declined in both logged areas and in wildlife habitat strips, as provision of habitat for logging-sensitive species is one of the major reasons for the prescription of wildlife habitat strips (Taylor, 1992). Several species could be tentatively described as such, with a repeat of the

caveat that counts for all species were small: the dusky robin is a ground pouncer that prefers open ground for foraging; the yellow wattlebird is a large honeyeater for which eucalypt nectar forms an important part of the diet (Blakers et al., 1984); the crescent honeyeater also forages for nectar during much of the year (Cale, 1994), although elsewhere in Tasmania this species had recovered numbers in regeneration forest of similar age to the present study (Taylor et al., 1997); the strong-billed honeyeater relies on loose bark as a foraging substrate (Cale, 1994). Thus, while foliage gleaners were generally not disadvantaged by the logging and changes in configuration of mature forest, some species with more specialised needs did less well, especially in upper slope habitat. Wildlife habitat strips in upper slope habitat in Tasmania have been shown to be less valuable for bird conservation than those in riparian areas, at least within a plantation matrix (MacDonald et al., 2002). There are also issues with the provision of wildlife habitat strips in non-riparian areas, as these are relatively rare, probably because they are more of an impediment to efficient wood production (Claridge and Lindenmayer, 1994). We suggest that the provision of retained vegetation in upper slope habitat, and the value of such vegetation, needs greater consideration in Tasmanian forest production landscapes.

Possible reasons for the reduction in bird numbers in retained vegetation corridors include habitat changes as a result of increased edge effects. Wildlife habitat strips within a plantation matrix are more likely to suffer disturbance than continuous mature native forest (Loofs et al., 2001). Microclimatic changes at forest edges have been documented in eucalypt forest (Dignan and Bren, 2003) and elsewhere in the world (Brosofske et al., 1997), although the extents of the changes vary with location and variable. In Canadian boreal forest leaf litter arthropods are less abundant at the edges between logged and mature forest than in forest interiors (van Wilgenburg et al., 2001). Although results from this forest type may not be transferable to eucalypt forests, in the present study counts of the Tasmanian scrubwren, which feeds on leaf litter arthropods, decreased considerably in retained vegetation corridors of all types. Riparian habitats may be more resilient to edge effects, as they are naturally linear features of the environment, and this may explain why

the wildlife habitat strips in upper slopes appear to perform worse at conserving populations of certain bird species.

Landscape configuration may also contribute to the reduction in bird abundance in retained vegetation corridors. The wildlife habitat strip width (100 m) is greater than that recommended as necessary to maintain bird communities by most studies (Hagar, 1999; Pearson and Manuwal, 2001). However, MacDonald et al. (2002) considered that wildlife habitat strips in a plantation matrix might be functioning as linear fragments. Reduction in bird abundance has been observed in many fragmented habitats (Hinsley et al., 1996; Hobson and Bayne, 2000; Robinson, 2001), at least of forest-interior species, and the present study area has not experienced an influx of edge and open country birds. If retained vegetation corridors act as fragments, increasing width would increase area, but the similarity of the response of streamside reserves (40 m wide) and wildlife habitat strips (100 m wide) in the present study suggests that this might not prevent the reduction in abundance.

Despite the limitations mentioned above, the results of the present study indicate that both wildlife habitat strips and streamside reserves have considerable value for the conservation of avifauna. Many species maintained populations in retained vegetation corridors similar to those before logging. These included two species, the striated pardalote and green rosella (Platycercus caledonicus), that nest in tree hollows, for which the retention of habitat is considered to be particularly important (Taylor, 1992). Two other species considered to be especially susceptible to logging (Taylor et al., 1997), the satin flycatcher (Myiagra cyanoleuca) and golden whistler, persisted in retained vegetation corridors after logging. Species typical of disturbed environments were either absent after logging, e.g. the common starling (Sturnus vulgaris) and noisy miner (Manorina melanocephala), or present in very small numbers, e.g. the grey butcherbird (Cracticus torquatus). Furthermore, to expect wildlife habitat strips to maintain populations of all species present prior to logging may be unrealistic. In this context, the results from retained vegetation corridors in the present study (the maintenance of a slightly depauperate version of the pre-logging avifauna) are somewhat reassuring for

land managers trying to maintain avian diversity at the landscape scale. Unless habitat deterioration occurs in corridors over time, the current study period is expected to provide the most radical differences between treatment and control areas. It will be interesting to see whether shortfalls between retained corridors and control areas lessen as the surrounding regeneration forest matures.

The current study forms part of a significant long-term study into the responses of flora and fauna to logging in the presence of retained vegetation corridors, and should ultimately be the first post-logging analysis of bird populations in the study area over a 90-year regeneration process. As such it provides valuable information on the impacts of logging and forest management on birds, both in Tasmanian dry sclerophyll forest and in similar habitat on the Australian mainland. Such long-term studies involve a large investment of resources, and as such frequently have to be restricted in spatial scale and involve fewer replicates than shorter term studies. While a spatially based study with a large number of replicates may be more statistically satisfying, such studies make inferences on temporal changes by observing spatial patterns. The current study reports actual changes over time, which we feel complements spatial studies to build a picture of the effects of logging practices on birds.

5. Conclusion

There is a general reduction in bird abundance in retained vegetation corridors after logging. The reduction in bird species richness per survey is a function of this, rather than of loss of species from the study area. Some sensitive species are identified, which may merit more intensive investigation in logged landscapes. We also draw attention to the fact that wildlife habitat strips in upper slope habitat appear to be less effective at conserving some logging-sensitive bird species. However, retained vegetation corridors within a matrix of native forest regeneration in dry sclerophyll forest can be considered to be an important component of the informal reserve system, as most species present prior to logging persist in wildlife habitat strips and streamside reserves.

Acknowledgements

Appendix A

Bird species recorded pre- and post-logging by treatment. Nomenclature follows Christidis and Boles (1994).

Species	Blackwood						Tea-tree						Upper slope						Total	
	WHS (n=72)		SR (n=48)		WHS (n=72)		WHS (n=72)		SR (n=48)		Control (n=48)		WHS (n=48)		logged (n=192)		Control (n=48)		(n=624)	
	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Brown goshawk (*Accipiter fasciatus*)	3	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	3	1
Common bronzewing (*Phaps chalcoptera*)	0	3	0	0	0	0	0	0	0	0	0	0	0	1	0	6	0	3	0	13
Green rosella (*Platycercus caledonicus*)	18	10	3	0	23	8	3	8	1	0	2	2	3	4	7	8	1	0	61	40
Pallid cuckoo (*Cuculus pallidus*)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	2	0
Fan-tailed cuckoo (*Cacomantis flabelliformis*)	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1
Shining bronze-cuckoo (*Chrysococcyx lucidus*)	0	0	0	1	0	0	0	0	0	0	0	5	0	0	0	0	0	0	0	6
Laughing kookaburra (*Dacelo novaeguiniae*)	1	0	0	0	0	0	0	1	0	0	3	0	1	0	1	0	2	0	8	1
Superb fairy-wren (*Malurus cyaneus*)	0	4	3	1	7	3	7	3	4	11	17	6	12	1	46	31	14	30	107	89
Spotted pardalote (*Pardalotus punctatus*)	0	0	0	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	0	3
Striated pardalote (*Pardalotus striatus*)	7	3	0	2	4	8	2	1	2	0	3	1	2	4	5	16	1	8	26	43
Tasmanian scrubwren (*Sericornis humilis*)	88	38	44	9	18	15	18	6	8	2	16	10	5	0	21	19	8	1	226	100
Scrubtit (*Acanthornis magnus*)	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
Brown thornbill (*Acanthiza pusilla*)	55	33	13	29	13	25	30	43	25	6	16	21	38	9	123	71	25	5	338	242
Tasmanian thornbill (*Acanthiza ewingii*)	69	24	38	16	18	17	5	4	3	0	7	5	1	0	4	2	1	0	146	68
Yellow wattlebird (*Anthochaera paradoxa*)	1	0	0	0	0	2	0	0	0	0	0	0	5	0	8	1	0	0	14	3
Yellow-throated honeyeater (*Lichenostomus flavicollis*)	1	2	0	2	0	0	2	3	1	8	2	11	1	1	3	8	1	1	11	36
Black-headed honeyeater (*Melithreptus affinis*)	0	4	0	7	9	18	0	4	0	3	0	11	0	1	9	29	4	7	22	84
Strong-billed honeyeater (*Melithreptus flavicollis*)	9	0	4	2	0	3	3	5	0	5	23	13	5	1	12	0	0	13	72	42
Crescent honeyeater (*Phylidonyris pyrrhoptera*)	8	0	12	1	5	3	7	5	23	1	18	10	2	0	10	4	1	1	86	25
New Holland honeyeater (*Phylidonyris novaehollandiae*)	0	0	0	0	0	0	0	0	0	0	0	2	0	0	2	0	0	0	2	2
Eastern spinebill (*Acanthorhynchus tenuirostris*)	4	6	6	12	2	5	1	0	0	2	0	1	0	0	5	14	0	0	18	40
Scarlet robin (*Petroica multicolor*)	0	1	0	0	3	1	2	5	0	0	3	3	3	7	12	0	10	5	33	22
Flame robin (*Petroica phoenicea*)	1	0	2	0	3	0	0	0	0	1	1	0	2	1	3	1	0	5	11	10
Pink robin (*Petroica rodinogaster*)	9	10	1	5	2	6	0	3	2	1	0	0	0	0	4	2	1	0	19	27
Dusky robin (*Melanodryas vittata*)	2	0	0	0	1	0	0	1	3	0	8	1	8	0	12	1	6	4	41	9
Olive whistler (*Pachycephala olivacea*)	4	1	4	2	2	1	1	0	0	3	1	1	0	0	0	3	0	0	12	11
Golden whistler (*Pachycephala pectoralis*)	1	5	0	2	1	1	0	3	3	0	0	2	1	0	6	0	2	2	14	15
Grey shrike-thrush (*Colluricincla harmonica*)	7	1	4	1	4	3	5	3	1	1	4	3	2	1	8	1	2	1	34	15
Satin flycatcher (*Myiagra cyanoleuca*)	1	2	0	1	2	1	0	0	0	0	0	0	0	0	1	0	0	0	4	4
Grey fantail (*Rhipidura fuliginosa*)	36	20	11	33	23	35	13	18	14	4	8	21	1	1	21	43	1	7	129	182
Black-faced cuckoo-shrike (*Coracina novaehollandiae*)	0	0	0	0	0	0	0	0	0	0	1	0	0	0	10	2	0	0	11	2
Dusky woodswallow (*Artamus cyanopterus*)	0	0	0	0	0	0	0	0	0	0	0	5	0	0	0	18	0	0	0	23
Grey butcherbird (*Cracticus torquatus*)	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	1	0	0	0	3
Black currawong (*Strepera fuliginosa*)	0	1	0	0	0	0	1	2	0	0	0	0	0	2	0	0	2	0	3	5
Grey currawong (*Strepera versicolor*)	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0	0	1
Beautiful firetail (*Stagonopleura bella*)	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	1
Tree martin (*Hirundo nigricans*)	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0	1	1
Silvereye (*Zosterops lateralis*)	11	13	18	4	4	6	4	4	0	2	0	0	0	4	0	11	0	0	37	44
Bassian thrush (*Zoothera lunulata*)	2	5	0	0	0	0	0	1	3	0	0	0	0	0	1	0	0	0	6	6
Total species richness	22	20	15	20	19	20	16	21	14	14	16	23	17	15	24	23	18	16	30	38

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