Plant species richness

The plant species richness (all trees, shrubs and groundcover species) at the sites ranged between 3 and 18 species, with a mean of 8 spp ± 3.6. The most common eucalypt species were *Eucalyptus rossii* and *Eucalyptus macrorhyncha*, and the most common shrub genera *Acacia*, *Daviesia*, and *Pultenea*. The common groundcover species were grasses and mat rushes such as *Joycea pallida, Austrodanthonia spp., Poa spp., Lomandra spp.*

Table 9.20 provides a list of the case study sites, in ascending order of plant species richness per plot. Table 9.21 presents the mean species richness per plot for the case study sites, grouped by Treatment.

Table 9.20 The 19 case study sites, presented in ascending order of plant species richness.

Location	Site Name	Years since clearing	Treatment	plant species richness/plot
Bungendore	MCthinned1	70-100	thinned	3
Murrumbateman	LB forestry 60yr	60	ringbarked	4
Bungendore	MCno_thin1	70-100	control	4
Murrumbateman	LB forestry 100b yr	100	control	5
Bredbo	pkd15yr2	15	pulled/chained	5
Murrumbateman	LB forestry 100yr	100	control	5
Bredbo	pkd oldest	200	control	6
Bungendore	MCthinned2	70-100	thinned	6
Frogmore	kk old site	100	control	7
Bredbo	pkd15yr2 control	100	control	7
Murrumbateman	LB forestry pit prop	60	ringbarked	7
Bungendore	MCno_thin2	70-100	control	7
Murrumbateman	LB forestry 30yr	30	bulldozed	9
Frogmore	kk medium site	70	ringbarked	9
Bredbo	pkd15yr1	15	pulled/chained	10
Frogmore	kk young site	33	bulldozed	10
Bredbo	pkd50yr	50	ringbarked	10
Bredbo	pkd100yr	100	ringbarked	14
Bredbo	pkd15yr1 control	100	control	18

Table 9.21 Mean plant species richness/plot for the case study sites, grouped by Treatment.

	Treatment 1 (pulled/chained)	Treatment 2 (bulldozed)	Treatment 5 (thinned)	Treatment 4 (control)	Treatment 3 (ringbarked)
Mean species richness per plot	7.5	8.5	4.5	7.6	8.8
Number of sites	2	2	2	8	5

Table 9.22 presents the only significant results of the GLM analysis performed on the species richness for the plants. Only two predictor variables (Table 9.12), i.e. Treatment and Regeneration, were able to account for a change in species richness for the plants, and they were only significant at the $p < 0.05$ level.

Site average species number varied between 4 and 9 (mean = 8.8) with thinned sites containing the lowest diversity and ringbarked sites the highest. Differences between the four treatments, other than thinned treatment, differ only by one species, and it is likely the significance of this reflects the overall low species diversity across these sites.

Table 9.22 Summary of model for plant species richness.

Response variable	predictor variable	P value	χ^2 prob
Plant species richness	Treatment	0.047	0.05
	Regeneration	0.031	0.05

Shrub and ground cover

Analysis of shrub cover indicated that there was no significant relationship between amount of cover and any of the predictor variables (Table 9.12) although there was a trend suggesting that, with more data, Age might become a significant predictor (Table 9.23).

Analysis of groundcover indicated that there was significant relationship between amount of cover and Regeneration and a trend suggesting Treatment as a predictor (Table 9.23).

Table 9.23 Summary of the models for shrub cover and groundcover.

Response variable	Predictor variable	Estimate	P value	χ^2 prob
Shrub cover (percent)	Age	+-	0.059	trend
Groundcover	Regeneration	-	0.007	<0.01
(percent)	Treatment	+	0.066	trend

9.8.4 Coarse woody debris

Coarse woody debris loads for the 19 sites varied between 0.3 t/ha and 47 t/ha (Table 9.24) and are presented in order of increasing coarse woody debris load. It can be seen from the table that bulldozed and thinned sites have the lowest coarse woody debris loads while ringbarked, pulled/chained and control sites had the higher coarse woody debris loads.

Table 9.24 Site locations, age since clearing. treatment and CWD loads (fallen dead wood only), been sorted from lowest to highest tonnages per hectare.

Location	Age (years)	Treatment	CWD (t ha^{-1})
Murrumbateman	30	bulldozed	0.3
Bungendore	100	thinned	1.3
Frogmore	33	bulldozed	2.5
Bungendore	100	thinned	2.1
Murrumbateman	100	control	4.6
Bredbo	200	control	8.1
Bungendore	100	control	8.9
Murrumbateman	60	ringbarked	11.2
Murrumbateman	100	control	14.5
Bredbo	100	control	17.0
Murrumbateman	60	ringbarked	21.8
Bredbo	100	ringbarked	22.4
Bungendore	100	control	25.1
Bredbo	50	ringbarked	26.8
Bredbo	100	control	31.0
Frogmore	100	control	35.1
Bredbo	15	pulled/chained	40.1
Bredbo	15	pulled/chained	47.0
Frogmore	70	ringbarked	47.5

Five of the seven sites with less than 10 t ha^{-1} have had coarse woody debris either physically removed from the site during harvesting (treatment = thinning) or reduced by frequent low intensity fires. Sites with more than 30 tonnes per hectare have been either undisturbed for 70-100 years or have accumulated large tonnages through the harvesting method (treatment = chaining). It is therefore not surprising that Treatment was a significant predictor of coarse woody debris load at the $p \leq 0.05$ level (Table 9.25) however no other forestry habitat variables were found to be significant.

Table 9.25 Summary of the model for CWD load.

Response variable	Predictor variable	P value	χ^2 prob
CWD	Treatment	0.023	0.05

The mean coarse woody debris load over all 19 sites is presented in Table 9.26. The table demonstrates that Treatment 2 (bulldozed) and Treatment 5 (thinned) had the lowest coarse woody debris loads, Treatment 3 (ringbarked) and Treatment 4 (control) had medium loads, while Treatment 1 (pulled/chained) had 2-25 times the mean levels of coarse woody debris found at all other Treatments.

Table 9.26 Mean coarse woody debris load at case study sites grouped by Treatment.

	Treatment 1 (pulled/chained)	Treatment 2 (bulldozed)	Treatment 4 (control)	Treatment 3 (ringbarked)	Treatment 5 (thinned)
Mean CWD load (t ha^{-1})	43.5	1.4	18.25	25.9	1.7
Number of sites	2	2	8	5	2

9.8.5 Regeneration

The number of regenerating trees per site ranged from none at all at the oldest site to 144 in one of the 70-100 year Bungendore plots (Table 9.27). The majority of regeneration at two Bungendore sites (MCthinned1, MCthinned2) was coppice from cut stumps, with up to 5 regenerating stems per stump.

Table 9.27 Number of regenerating stems of dominant canopy trees (DBH = 5cm) for the 19 study sites. The sites are presented in ascending order of numbers of regenerating stems.

Location	Site Name	Age (years)	Treatment	no. regenerating stems/plot	no. regenerating stems ha^{-1}
Bredbo	pkd oldest	200	control	0	0
Bredbo	pkd15yr1	15	pulled/chained	3	24
Murrumbateman	LB forestry 100b yr	100	control	6	48
Frogmore	kk old site	100	control	7	56
Murrumbateman	LB forestry 60yr	60	ringbarked	10	80
Bungendore	MCno_thin1	70-100	control	10	80
Bredbo	pkd100yr	100	ringbarked	14	112
Bredbo	pkd15yr1 control	100	control	15	120
Frogmore	kk young site	33	bulldozed	16	128
Murrumbateman	LB forestry 30yr	30	bulldozed	17	136
Bredbo	pkd50yr	50	ringbarked	17	136
Bredbo	pkd15yr2	15	pulled/chained	19	152
Bredbo	pkd15yr2 control	100	control	19	152
Murrumbateman	LB forestry pit prop	60	ringbarked	23	184
Frogmore	kk medium site	70	ringbarked	29	232

Location	Site Name	Age (years)	Treatment	no. regenerating stems/plot	no. regenerating stems ha^{-1}
Murrumbateman	LB forestry 100yr	100	control	45	360
Bungendore	MCthinned2	70-100	thinned	50	400
Bungendore	MCthinned1	70-100	thinned	119	952
Bungendore	MCno_thin2	70-100	control	144	1152

The GLM models found significant relationships between regenerating trees and groundcover, treatment and age (Table 9.28).

Table 9.28 Summary of the model for regeneration.

Response variable	predictor variable	estimate	P value	c2 prob
Regeneration	groundcover	+	0.003	**
	Treatment	+	0.033	*
	age	+	0.017	*

P = significance of Two-Sample Assuming Unequal Variances t-test
** $p<0.01$
* $p<0.05$

Table 9. describes regeneration at each different treatments site and it can be seen that pulled/chained sites had the lowest level of regeneration, bulldozed and ringbarked sites have similar levels, while control and ringbarked sites the highest.

Table 9.29 Mean regeneration at case study sites, grouped by Treatment.

	Treatment 1 (pulled/chained)	Treatment 2 (bulldozed)	Treatment 3 (ringbarked)	Treatment 4 (control)	Treatment 5 (thinned)
Mean regeneration (stems ha^{-1})	88	132	149	246	676
Number of sites	2	2	5	8	2

9.8.6 Landscape Function Analysis

LFA scores do not automatically classify a site into poor, moderate or good landscape condition. The significance of the LFA value comes from the comparison of study sites with a reference or analogue site. For the case study sites, an LFA value between 50-60 is regarded an indicator of good functional condition (David Tongway personal communication).

Table 9.30 presents the results from soil surface condition indicators for each site. The means are calculated from 5 replicates of 10 soil condition indicators for each landscape unit within each site (average of 3). The results (individual observations of the soil surface), are grouped into three indices:

1. *Stability*: assesses the ability of the soil to withstand erosive forces and to reform after disturbance
2. *Infiltration/runoff*: an assessment of the infiltration rate of rainfall into the soil.
3. *Nutrient cycling*: the efficiency of soil organic matter cycling back into the soil

Table 9.30 Summary of the model for LFA soil surface condition indices.

Soil Surface Condition		**Individual zone contribution to the whole Landscape** Values are scored with maximum of 100*								
Property/site	**Treatment**	**Stability**	**Std err**	***P***	**Infiltration**	**Std err**	***P***	**Nutrient cycling**	**Std err**	***P***
Bredbo 15yr1	pulled/chained	70.9	2.5		51.6	3.4		47.5	3.9	
	control	68.4	3.4	*ns*	53.7	5.0	*ns*	49.1	6.9	*ns*
Bredbo 15yr 2	pulled/chained	65.6	4.4		44.1	4.1		43.0	4.9	
	control	68.2	2.0	*ns*	52.4	2.4	**	49.4	3.2	*
Bungendore (MC 1)	Thinned	65.4	2.9		50.4	3.9		49.7	2.6	
	No thinning	64.1	2.9	*ns*	59.7	2.8	*	59.3	4.0	**
Bungendore (MC 2)	Thinned	72.2	2.9		55.9	4.3		54.3	6.2	
	No thinning	71.6	1.6	*ns*	50.8	5.9	*ns*	50.9	6.7	*ns*

P = significance of Two-Sample Assuming Unequal Variances t-test
** p<0.01
* p<0.05
ns P>0.05

The model detected no significant differences in soil surface stability, infiltration or nutrient cycling found between Bredbo Site 15yr 1 (pulled/chained 15 years ago) and the Bredbo 15yr 1 control (Age =100 years), located 500 metres away.

However we found significant differences in infiltration and nutrient cycling between Bredbo 15yr 2 site (pulled/chained 15 years ago) and the Bredbo 15yr 2 control (Age =100 years), located 200 metres away. The pulled/chained site had lower values for the infiltration index than the control, implying that infiltration of rainfall had not returned to its pre-treatment level.

There were also significant differences in soil surface infiltration and nutrient cycling between Bungendore MCthinned_1 (thinned 10 years ago) and the Bungendore MCno_Thin1 site (70-100 years), located 200m away. The thinned site had lower values than its non-thinned control for infiltration.

There were no significant differences detected in soil surface stability, infiltration or nutrient cycling between Bungendore MCthinned_2 (thinned 10 years ago) and the Bungendore MCno_Thin2 site (70-100 years) located 200m away.

9.8.7 Forestry/habitat variables

Modelling of forestry/habitat variables has already been done at a larger scale across the MDB, in the development of a model to predict stem wood volumes (see Section 6). Results of this analysis indicated that, at a stand level, the model has a large standard error i.e. ±153% of predicted level at the 95% confidence limits. However, when the model was used to predict across stands in a wider area in the same NPP class, the standard error was significantly reduced i.e. to ±17% of predicted level at the 95% confidence limits.

In the light of this, it is interesting that the modelling for assessment of ecological impacts of firewood harvesting showed that both treatment and live tree stocking density were significant predictors at the $p < 0.05$ level of live tree basal area (Table 9.31). Sites which have been bulldozed or pulled/chained, where there are the younger, regenerating stands, have smaller live tree basal areas, frequently between 10-20m², in comparison to older ringbarked or control sites, where live tree basal areas average around 30m², ranging between 20-40m².

Table 9.31 Summary of the model for live tree basal area.

Response variable	predictor variable	P value	χ^2 prob
Live tree basal area	Treatment	0.030	0.05
	Live tree stocking density	0.040	0.05

Table 9. Summary of GLM of individual predictor (forestry/habitat) variables found to be a significant predictor of live tree basal area. ns = P>0.05, * = p<0.05, ** = p<0.01, ***=p<0.001

9.9 Discussion

The case studies have created a framework for the investigation of the potential impacts of firewood harvesting through the selection of forest and woodland sites which have forest stands of different ages since they were harvested or cleared over the last 150 years, and which have been harvested using different harvesting methods. The framework also includes two sites where disturbance has been minimal.

The key question is whether this framework is able to provide evidence that firewood harvesting, through clearing or thinning of live trees, has a some significant effect on biodiversity, either positive or negative. Thee case study approach has provided a number of significant relationships between habitat/forestry variables and differently aged and/or harvested sites. It is important to ask whether these results are comparable with the effects of the silvicultural regimes proposed for the green-wood scenario. This issue can be examined through consideration of silvicultural management scenarios and the type of harvesting the dead-wood and green-wood scenario entails.

The green-wood scenario does not imply a simple harvesting formula. Low forest productivity i.e. low commercial productivity, has meant that appropriate sites for investigating the ecological impact of green-wood scenario, i.e. native woodlands and forests with the appropriate harvesting regimes and methods of harvest, are very limited in number. There is also a broad range of forest and woodland types (see Section 8.6) so at this stage the green-wood scenario does not specify long-term silvicultural management regimes for low rainfall-area forests.

For these reasons there are a number of recommended ecological principles which should underpin silvicultural management under the green-wood scenario:

1. Silvicultural practices should encourage the retention of multi-aged stands to provide a diverse mosaic of ecological niches and resources ie. for food, breeding, shelter;
2. Silvicultural practices should encourage retention of larger trees for hollows as wildlife habitat i.e. only thinning trees =15cm DBH; and
3. Silvicultural practices should encourage the maintenance of heterogenous landscape matrix, by using a combination of “flexible selection” harvesting regimes (thinning approximately 50% of the forest stand basal area) and “group selection” harvesting regimes (clear felling of small gaps to create matrix of open spaces).

We found only two sets of experimental sites in NSW older than ten years. One set comprises the Australian National University School of Resources, Environmental and Society experimental sites, established to examine the effects of silvicultural treatments on the growth rates of trees and diversity of the understorey in a dry sclerophyll forest. The other wet was a eucalypt growth and thinning plantation trial, established by NSW State Forests near Parkes, a low rainfall area in forestry terms.

Additionally, it is extremely difficult to find forest stands with a known age since harvesting. It is the exception rather than the rule that any State agencies have any written records, at the stand-specific level of detail, for sites older than 30 years. The process of locating sites of known age is reliant upon locating and talking to private landowners who have a clear knowledge of regeneration and clearing events, on their own properties, over a significant time period.

Of the 19 case study sites, 2 were pulled/chained, 2 were thinned, 2 were bulldozed, 2 were "undisturbed", and 11 ringbarked between 50 and =100 years ago. As a group these represented sufficient representation of the range of harvesting and age classes necessary to examine the ecological effects of harvesting under the green-wood scenario, however the actual number of available sites was not sufficient to adequately replicate the forest types, treatments and ages. As a consequence, the types of analyses were severely limited by the lack of adequate data.

Finally, it is important to recognise that all study sites were situated within well forested agricultural areas, where forests and woodlands form between 30 and 80% of each property. Ecological impacts may differ where sites fall in more isolated remnants within an agricultural landscape, surrounded by open paddocks and cropping with occasional remnant trees. This consideration contributes to the rationale for exclusion of remnants with an area < 100 hectares from the estimates of area available for harvesting under the green-wood scenario (exploitation criterion 9, Section 3.3).

9.9.1 Birds

Species richness

Bird species richness has been correlated to a number of habitat variables in forest and woodland remnants in the Southern Tablelands. The structural diversity of a patch, i.e. its canopy cover, density of shrubs, and the amounts of litter and ground cover, has been shown to strongly influence the diversity of bird species in an area (Wiens 1989). For example in the Central Lachlan hill communities, comparable to these Southern Tableland hill communities, increased vegetation cover at heights between 0.5 and 2 m in was related to increased bird diversity (Seddon et al. 2002, in press). The size of the forest or woodland remnant, the habitat complexity score (a ranking of tree, shrub and groundcover cover levels) and the distance to the nearest 10ha woodland remnant significantly influence the number and presence of particular bird species (Watson et al.2001). However in large area forests, the main attributes influencing bird abundance and species are the structure and composition of the vegetation (Ford 1985).

In the case study, between 3 and 28 species were recorded at each site, with 41 species recorded in total. Twelve of the 15 sites had observations ranging between only 8 to 15 bird species. The top 4 species-rich sites were either the youngest, oldest or located on deeper soils along a creek or valley floor. In comparison, the 5 most species-poor sites were the 50 to =100 year-old ringbarked stands, characterised by dense, even-aged stands and located along dry slopes and ridges. The mid-range of species richness was occupied by a combination of older and younger sites.

There are a number of spatial and temporal factors influencing this variation. Species richness for each site will vary according to daily, seasonal and longer term patterns over time, therefore it would be expected that these figures significantly underestimate true species richness compared to what might be recorded from longer term observations. Also, Autumn is not the optimum time for bird surveys, when bird vocalisations are reduced as a result of post-nuptial moulting, so survey observations based on calls are less likely than during the spring breeding season.

However, the positive relationship found between bird species richness and shrub cover is also significant. as up to 10 of the 15 sites had less than 5% shrub cover. The lack of shrub cover will influence species richness and the significant results correlating bird species richness with shrub cover in this study was consistent with previous research.

The lack of a detectable significant response of bird species richness to groundcover, canopy cover, plant diversity and coarse woody debris was surprising. As discussed in Section 9.2, the more structurally complex sites have been related to bird species richness. For example MacNally et al.(2002) found that the densities of wood-dependent Brown Treecreeper increased with higher loads of coarse woody debris. A greater number of occurrences would have enabled more individual species like the Brown Treecreeper to be analysed and modelled.

However, this result is also consistent with contradictory results from studies examining relationships between coarse woody debris and bird species number (Driscoll et al. 2000). Results may have been different if these site variables were combined to produce a matrix similar to that for habitat complexity scores (ie. a ranking of the value of the habitat based on combining the individual scores). Certainly the habitat complexity index, which combines the percent cover of canopy, shrub, ground, leaf litter and log cover, is a significant predictor of bird species richness in remnant woodlands (Drew et al. 2002; Freudenberger 2001; Watson et al. 2001) and all these variables were present as separate habitat variables in this study.

Bird abundance

The bird survey data was also analysed for bird abundances over all sites. Although birds species richness is traditionally undertaken as the initial analysis, it is also useful to investigate abundance, because both the distribution and abundance of birds are strongly related to landscape variables such as size of a habitat patch and its composition (Watson et al.2001). Glanznig (1995) provided an overview of native vegetation clearance in Australia and its implications for biodiversity and found that numerous studies in many states concluded bird abundances are directly related to the degree of habitat loss and fragmentation. In the Western Australian wheatbelt, 95 of the 195 species of birds recorded have declined in range and/or abundance since the region was developed for agriculture. Most of these losses are due to loss of habitat and fragmentation of the remainder (Saunders and Ingram 1995). Regional loss and decline of bird species has been found in other States such as New South Wales (Barrett et al.1994) and Victoria (Loyn 1986).

More individual birds were found in the younger chained and bulldozed sites than in the older ringbarked or control sites. It is again relevant here to highlight the ecological characteristics of these sites, because of their significance to the ecological impacts of any green-wood harvesting regime. The higher bird abundance sites are the younger sites (15-30 years) with more recent and greater levels of disturbance, and were generally within a landscape with a mosaic of patch types. These patch types ranged between "open" (in between regenerating tree stands) or "dense" (within regenerating stands) and tended to be more structurally complex (see the left hand photograph of Figure 9.21). The older sites (50-100 years) were even-aged stands, whose tree stocking densities were usually approaching the maximum for the site and which were less structurally complex, providing a different level of resources (see the right hand photograph of Figure 9.21). It might therefore be more appropriate to recommend that green-wood harvesting scenarios focus on these 50-100 year old sites.


Figure 9.21 The 15 year-old pulled/chained site (on left) has more structural complexity with fallen timber and shrubs and greater bird abundances, in comparison to the 100 year-old site (on right), where tree stocking densities are getting to their maximum for the site resources.

Two other habitat variables were significant in the models: density of dead trees and canopy cover. The basal area of dead trees is significantly correlated to increased plant species richness. It is possible that a higher plant diversity will provide a greater abundance of resources for a bird in terms of structure, food availability and nesting sites. Dead trees can provide nest hollows, if the tree age is sufficient to have allowed hollow development, and provide perches in an open area.

Canopy cover is generally lower at the younger sites, where trees have not yet reached their full site stocking capacity. A significant relationship between bird abundance and canopy cover may again reflect the higher structural complexity in these younger sites with a greater diversity within the patch/interpatch matrix.

Management considerations for birds in dry sclerophyll forests

The implications of shrub cover for different aged stands under the green-wood scenario are important. The basal area of trees was negatively correlated with habitat complexity in the wetter forests of south-eastern New South Wales (Catling and Burt 1995) and our results appear to show the same trend in these drier forests. A mid-aged 50-100 year-old, densely stocked regrowth stand in these forests is less likely to have large areas of shrub cover. The exceptions are areas where site resources are not being completely dominated by the trees e.g. the forest edge, areas of higher fertility soils, or aspects sheltered from solar radiation, winds and temperature extremes. In contrast forest stands with higher shrub covers tend to be the younger highly disturbed sites (ie harvested) and the oldest sites where widely spaced large old trees form a woodland rather than an very dense regrowth forest.

It is also important to consider the natural distribution and abundances of shrubs within the Southern Tableland vegetation communities in terms of forest management. Large areas of the landscape on the lower slopes and plains, with deeper and more fertile soils, were predominately Yellow Box/Red Gum Grassy Box Woodlands. The grass and forb understorey of these woodlands was highly diverse, however shrubs were not a predominant component and their abundances were typified by their patchy occurrences (NSW National Parks and Wildlife Service 2002). These hill communities may have been characterised by a higher shrub component and bird species richness, however large scale changes to forest structure that increase the shrub component are not necessarily desirable. The natural intrinsic variation in the structural component of the vegetation communities indicates that forests with a mosaic of ages, structural and species diversity are the preferred management goal.

It is also clear that for all bird species, large areas of one habitat type or forest age type creates a sub-optimal habitat (Williams et al. 2001). A even-aged, more homogeneous stand has different foraging, nesting, refuges and breeding resources available to forest birds (Wardell-Johnson and Williams 2000) than those provided by more heterogeneous environments. Many 50 to =100 year old sites are characterised by even-aged stands with their corresponding lack of stand structure, whereas the 15-30 year old younger sites are characterised by the patchy nature of their regeneration and a shrub/groundcover layer providing a different level of structural diversity.

Biological and physical edge effects may occur on boundaries between harvested forest stands and have a particular influence on bird distribution and abundance, so are an important consideration in forest planning (Wardell-Johnson and Williams 2000). This study did not specifically address these issues, however they must be considered as influential factors on current and future forest stands. It is likely that the site at Frogmore with the greatest species diversity was impacted by an adjoining small open grassy plain with a permanent creek, and that sites such as 50 year old Bredbo and 60 year old Murrumbateman sites, with only eight bird species, were influenced by their location within very large even-aged stands and not being adjacent to other edge types (see Figure 9.21).

It would be incorrect to surmise that these forests are not important in terms of their biodiversity. On a compositional level, these sites have a level of bird species richness and abundance that is of some significance. However, they are likely to the lower diversity end of the scale, compared to other areas of remnant woodland, gullies, open depressions or other forest types present within the dry sclerophyll forest matrix.

9.9.2 Small ground-dwelling mammals

The distribution and abundance of small ground-dwelling mammals in south-eastern Australia's "wet sclerophyll" forests has been intensively researched (Catling and Burt 1994; 1995; 1997, Catling et al. 1998; 2000) and, due to the absence of similar work in the lower rainfall, dry sclerophyll forests, is used to provide a framework for this discussion of small mammals. In brief, these studies found that abundances of small mammals were positively correlated with habitat complexity and although there was a significant relationship to nutrient status (magnesium) of the foliage, the overall nutrient status was not the highly influential factor found for arboreal fauna (Braithwaite 1983, Braithwaite et al. 1983, Braithwaite et al. 1984). Modelling in relation to habitat variables found abundance of *Antechinus* and *Rattus* species related strongly to season, and identified their preferences for undisturbed forest, particularly the moist, structurally complex sites with dense understorey and specific eucalypt communities.

Both the Yellow-footed Antechinus (*Antechinus flavipes*) and the Agile Antechinus (*Antechinus agilis*) are widespread in a variety of habitats, and so are amongst the most probable of small mammals to be found at the case study sites, with the exception of the exotic house mouse or *Rattus* spp. It is clear the dryness of the forests contributes to lower overall abundance of small ground dwelling mammal fauna compared to abundances found in the wetter forests (Catling, personal communication.). Generally, the case study areas are characterised by their skeletal soils and low nutrient status and a lower level of mammal diversity. A previous survey of *Antechinus* spp. in the Southern Tablelands woodlands and forests found low overall densities of only 1-2 individuals ha $^{-1}$ (Dickman 1980). Where higher densities of *Antechinus* occurred, they were significantly related to whatever cover is available, ie. gullies, tree-log complexes and logs. These figures correlate closely with the species and the abundances recorded at the Murrumbateman and Frogmore sites during this study. The lack of dense shrubby groundcover, which is positively correlated with increased basal area (Catling et al.2000), would be a major contributor to the low densities of small ground-dwelling mammals. Although the sparseness of the data precluded analysis, numbers were similar across all sites, with the highest abundance recorded in the moister gully.

These findings are reinforced by a parallel CSIRO study being undertaken in the Murrumbateman dry sclerophyll forest where some of the case study sites are located (CSIRO 2003). In that study small mammal surveys using tracking tunnel techniques were undertaken throughout the forest to determine areas of greatest abundances. It found that the wetter gully areas on the lower parts of the slopes had the highest densities in comparison to the drier upper slopes (Steve Henry, personal communication). These findings complement those mentioned in an earlier paragraph from Dickman (1980) and reinforce the appropriateness of current forestry guidelines which exclude drainage lines/gullies from any harvesting, which are obviously vital habitat within these lower rainfall areas. These can be compared in many ways to the value of refugia in semi-arid and arid habitats – an area in which a species or suite of species persist for short periods when large parts of their preferred habitats become uninhabitable because of unsuitable climatic or ecological conditions e.g. drought, flooding or biologically-driven collapses in food supply (Morton et al 1995). In these dry forests drainage lines provide the presence of relatively dependable supplies of moisture and nutrients so may provide refugia for animals dependent upon regular plant production for persistence in the uncertain Australian climate.

It was expected that the Bredbo property, with its extensive landscape mosaic from dense shrubby sites to open forest, would reflect some preferences of small mammals for the sites with higher habitat complexity. However, with only one capture across all sites, the overall abundances were extremely low. Low spring abundances have been shown to relate to the unique pattern of reproduction in *Antechinus* species, where all males die naturally in late winter (Catling et al.1998). However the case study was undertaken during autumn, so it would appear that low abundances were reflecting a generally low nutrient status (Catling, personal communication.) and the effects of the previous drought year.

Based on this research and the study results, in terms of forest management, ground-dwelling mammals seem to represent the remnants of a more diverse fauna which have declined because of extensive clearing, a lack of source-areas to recolonise from, disturbance (fire, grazing and forestry activities), feral predators, forest with generally low nutrient status and lack of structural complexity. A statement from NSW National Parks and Wildlife Service 2003, from the Mundoonen Nature Reserve on koalas may best summarise the potential ecological impacts of harvesting:

Few old growth trees remain. While it is thought that dense thickets of trees may thin naturally over time, there are some indications that in some instances these may become locked by limited nutrients and trees may not grow. Field observations during the koala survey indicate that these dense thickets provide sub-optimal koala habitat. Koala scats were found under trees with a broader cleared area around, potentially for added protection from predators. The management of these regrowth areas may become more active in the future.

The forests sampled during the case study have large areas of “sub-optimal thickets” which support regrowth that is very different from the original forest. Silvicultural management in these areas, such as harvesting for firewood, could benefit the abundance and diversity of ground dwelling mammals if it results in a diverse mosaic of habitats providing a complexity of areas which include dense shrub and groundcover, multi and even aged stands within the dry sclerophyll forest environment. Although abundances will never be comparable to the wetter forests it is possible that, with forest management, densities of *Antechinus* could reach least 1-2 individuals ha^{-1} and the same species diversity may be feasible under the green-wood scenario.

9.9.3 Plants

Plant species richness could be considered as relatively low with an average of 8 species per site. Thinned sites had the lowest plant diversity across all sites and was almost half the average species number in comparison to all other sites highlighting Field and Banks (1999) comments on low diversity of these sites. However pulled/chained sites and control sites had on average 3 more species, while bulldozed and ringbarked species had on average 4 more plant species. Plant diversity is highest in ringbarked sites, which are actually those traditionally viewed as potentially least diverse. However differences between the four treatments, other than thinned treatment, vary only by one species, and it is likely the trend towards significance in the analysis is only reflecting a difference between ‘extremely low’ and ‘low’ plant diversity and that overall these sites are characterised by low species diversity across these sites. In terms of impacts of firewood harvesting on these sites it is therefore difficult to draw strong relationships to ‘treatment’ and is perhaps more useful to draw some comparisons to other vegetation types within this dry sclerophyll forest matrix, such as those sites at Picaree Hill.

Flora surveys undertaken at Picaree Hill (Gould 2003) found the woodland species richness to be 20-30 species per plot, in comparison to 30-50 species in other areas e.g. open depression, shrubby grassland and cleared woodland. This indicates that these case study sites, with high basal areas, high canopy cover, extensive competition for resources, thick leaf litter and often hydrophobic soils, are at the lower end of the plant diversity spectrum, in comparison to other more diverse sites within close proximity. If firewood harvesting was to be undertaken in these forests these particular type of forest stands, exemplified by the case study sites, are likely to be those most suitable in terms of impacting on plant species diversity although the combined effects of the drought and autumn sampling are also likely to be strong contributors to this studies low plant diversity. However, the results also appear to reflect differences in distribution of flora species and species richness across the different areas of forest.

The potential rate of any change in plant diversity should also be considered. As discussed further (Section 9.10.2) these forest stands will self-thin at some stage with subsequent changes to plant diversity. If firewood harvesting is undertaken that "mimicked" this self-thinning process what would be the response of these stands in terms of plant diversity? It may be that the capacity of the sites to respond to changes is relatively slow when the silvicultural treatment is thinning. Plant species richness at sites where the basal area had been reduced by 60% (Fields and Banks 1999) did not increase in the 8 years post-treatment, even at sites where post-harvest treatments such as direct seeding, chisel ploughing and burning were applied. Similar results were found for understorey composition in a dry-to-moist sclerophyll (*Eucalyptus sieberi*) regrowth forest after thinning 50% of tree basal area (Bauhus et al.1999), where the species richness did not change significantly after six years. Previous work (Bauhus et al.1999) had shown that understorey species are better adapted to larger scale disturbances such as clear-felling and burning, but further research is required on the effects of intensive silviculture on these forest types.

9.9.4 Coarse Woody Debris

As well as its importance for ecosystem functioning a site with undisturbed coarse woody debris load often has relatively intact ecological variables such as plant diversity, regeneration and litter cover (see results of the literature review; Section 9.2.1). Undisturbed sites, for example see the left hand photograph in Figure 9.22) are not common and were difficult to locate. In over 70 field sites selected across the MDB we considered only 10 of these sites to be consistent with undisturbed (not harvested) coarse woody debris.

Figure 9.22 The site in the left hand photograph is one of the oldest sites (=100 years-old) in the case study area. It is relatively undisturbed and has over 35 t ha-1 of coarse woody debris. The right hand photograph shows a regenerating 30 year-old forest stand, with significantly low loads of coarse woody debris measured at 2.5 t ha-1. Both sites are at Frogmore.

After analysis of coarse woody debris with other forestry/habitat variables including birds (Section 9.8.1 and 9.8.7) the only significant predictor of coarse woody debris load was found to be Treatment. However, from the forest growth and yield modelling (Section 6) we know that there is a strong relationship between coarse woody debris load and live tree stem wood biomass, which in turn depends on the Treatment, through what appears to be a simple straight line relationship. This is not unexpected because larger trees have larger limbs and the dead standing trees are larger. This type of result was expected for this analysis.

This suggests that, regardless of all the other forest variables, the most significant influence in terms of coarse woody debris load present now, is human management history i.e. Treatment, and we can see quite clearly at the site level that there are significant differences between coarse woody debris loads. It is useful to highlight and examine the differences in light of the forest histories of the sites and their treatment over time.

Sites with a fallen coarse woody debris load = 10t/ha

There were seven sites across different properties which had a coarse woody debris load of less than 10 tonnes per hectare. The site names and locations are located in the coarse woody debris Table 9.24. These sites fell into three readily classifiable groups:-

Thinned sites

In the Bungendore sites, which were part of an experiment specifically designed to examine the effects of thinning treatments, all fine and coarse woody debris was removed during the thinning process. A coarse woody debris load of 1-2 t ha^{-1} represents all coarse woody debris fallen over ta period of 10 years since thinning. For the control sites, where no thinning treatment had been applied, coarse woody debris was 9 and 25 t ha^{-1} respectively.

Bulldozed sites

A typical method of clearing forests and woodlands for pasture during the 1970's was using a bulldozer to push trees into 'windrows' ie bulldozed trees pushed together into rows of stacked wood. The survey plots on the two bulldozed case study sites did not have any windrowed piles and consequently the coarse woody debris consists only of fallen timber from the regeneration, usually very low amounts as trees are in a active growth stage rather than shedding or losing old branches, or small amounts of coarse woody debris left behind after bulldozing.

Sites =100 years old

It would be expected that sites of this age would have accumulated at least 20 t ha^{-1} of coarse woody debris. However, further investigations into site history revealed that selective timber removal for fencing and firewood had occurred, as well as frequent low intensity autumn burns to reduce potential high fuel load build-up which resulted in lower coarse woody debris levels.

Implications for management

Coarse woody debris loads less than 10 t ha^{-1} appear to be well below the level expected for dry sclerophyll forests and, in the absence of any other data we presume this is not at a sufficient level for long term ecological processes, we recommend that harvesting under the dead-wood harvesting scenario is inappropriate where such low loads occur.

Sites with a fallen coarse woody debris load between 10-30 t ha^{-1}

Judging from the field work and coarse woody debris modelling undertaken by this project, it would appear that 10-30 t ha^{-1} is under-representative of the average coarse woody debris loads for dry sclerophyll forests (see Table 9.24 for site names). However it is difficult to benchmark these figures by comparing them to any of the 15 published dry sclerophyll forest coarse woody debris studies for two reasons summarised here but addressed in specific detail in Section 9.2 . Firstly, the range of values provided by the studies (2-130 t ha^{-1}) is so wide, and secondly, the vegetation communities are not the same across the studies and so may not be comparable.

Another potential area for comparison would be estimated fuel load for sites on closely situated nature reserves; Mundoonen, Brindabella National Park and Tinderry Nature Reserve. However fuel loads have not yet been estimated for these reserves (Jo Calwell, personal communication). Other fuel load estimates, such as that of 54-76 t ha^{-1} (Cavicchiolo 1991) at the Bungendore property, are based on = 6 millimetre wood diameter, much smaller than the = 10 centimetres used in this study.

Implications for management
Based on the modelling of the relationship between live tree stem wood biomass and coarse woody debris loads (see Section 6, Figure 6.3), the coarse woody debris load predicted for these sites was between 30-70 t ha^{-1}, so the actual figures fall at the extreme lower end of the predicted range. Until comparable data becomes available, it would seem reasonable to assume that the actual coarse woody debris loads are depleted, although they are representative of typical loads currently found in dry sclerophyll forest.

Sites with a fallen coarse woody debris load between 30-40 t ha^{-1}

Of the 5 sites with the highest coarse woody debris loads (see Table 9.24), two were the Bredbo pulled/chained sites, where all standing tree and shrub material were felled in order to harvest firewood, and no windrowing was done. This process left large volumes of coarse woody debris, even after harvesting for firewood.

For the 3 remaining sites, coarse woody debris loads are higher than all the other sites, but fall within the 30-70 t ha^{-1} range which was predicted by the model and so is likely to reflect the higher live tree stem biomass sites and greater age of these sites.

Implications for management
High loads of coarse woody debris suggest that some harvesting under the dead-wood scenario could be considered, and that the harvestable amount would be significantly greater than that available at sites with loads of 10-30 t ha^{-1}.

9.9.5 Regeneration

In Section 9.3.1, the importance of regeneration as one indicator of biodiversity (at the functional level) was discussed. This is because regeneration of canopy trees relies upon the appropriate function of specific ecosystem processes which provide specific triggers for germination and the subsequent survival of seedlings. These processes include sufficient seed fall, limited ant predation, some major disturbance (usually fire), appropriate temperatures, adequate moisture, light and some open ground area as thick, undisturbed litter layer reduces the survival of seedlings. The timing of the sequence of the processes is also critical because eucalyptus seed does not survive longer than 6-12 months (Florence 1996). It has also been noted that the Southern Tablelands dry forests are a diverse group and it is *unwise to generalise their regeneration requirements* (Hamilton and Cowley 1987).

In this project, regeneration of canopy trees was found to be significantly related to the harvesting treatment. Although sites harvested by pulling/chaining had the lowest numbers of =5cm DBH regeneration forestry data reveals that up to 5,000 stems per hectare at these sites are less than 10cm DBH ie. regeneration from 1985. Several factors potentially contribute to regenerative success at sites The treatment pulls/fells whole trees, complete with seed in capsules, directly across the slope onto freshly disturbed soil surfaces created by uprooting the next tree which leaves a deep depression in the soil surface where the tree roots were previously. Microsites provided by these conditions appear to provide ideal conditions for seed fall, seed survival, germination and seedling survival.

That pulling/chaining can have regeneration at these levels contrasts with other studies on regeneration in these forests. Traditionally, cutting the coppice of epicormic shoots which develop on a stump which has been cut close to the ground has been regarded by the forestry literature as the most appropriate method for harvesting dry sclerophyll forests. Certainly the highest levels of regeneration were recorded in thinned sites (Table 9.29). Coppicing is regarded as the characteristic response to cutting of commercially harvested species in these forests, whereas in wet sclerophyll forests, the response is characterised by new seedling regeneration (Jacobs 1955, Florence 1996). Early work by the Forestry and Timber Bureau of Research (Jacobs 1955), on regeneration of fuelwood in dry forest at Black Mountain ACT, regarded coppicing as the only appropriate silvicultural techniques for these "poor quality" forests.

There is no doubt about the capacity of these forests to regenerate from coppicing. Both thinned sites at Bungendore had amongst the highest levels of regeneration recorded across the 19 sites, the majority of which was coppice growth from cut stumps. However, forestry research has also showed that the harvesting of coppice shoots was limited, from 2-3 harvests (Jacobs 1955) to a maximum of 4 harvests (Florence 1996), depending on the site and tree species. Therefore, unless there is adequate seedling regeneration, it is unlikely that harvesting coppice alone would be sustainable over the long-term.

9.9.6 Landscape Function Analysis

Landscape function analysis (LFA) was selected to assess the functioning of forest ecosystem processes after green-wood harvesting. LFA is based on the theory that landscapes function to capture, concentrate and conserve water and nutrients. Water enters the system as rainfall and is rapidly redistributed as run-off or captured as run-on by landscape patches, reserves or sinks. With runoff, nutrients are also transferred to patches as sediments and litter, where they are assimilated into the biomass of organisms in pulses of growth, then slowly recycled through death and decay (Ludwig and Tongway 1997).

The results of the LFA analysis indicate that for all sites on both properties, harvested by thinning or pulling/chaining, the landscapes are still relatively functional in terms of their stability (their ability to resist erosive force and reform), infiltration (water available for plants) and nutrient cycling (efficient soil organic matter cycling) (Tongway and Hindley personal communication).

However there were conflicting results across the sites and we are unable to definitively ascribe differences in landscape functioning to type of harvest method. In two cases the results supported our contention that landscape functioning was linked to harvest type ie. that pulled/chained sites were more functional than thinned sites. In two cases the analysis did not support it. In these two cases where there was no significant difference between the control and harvested site the reasons may be a) landscape stratification was not sufficiently efficient to capture these differences and/or b) the landscapes were inherently more resilient to start with (greater groundcover, shrubcover, soil depth and flatter slopes and sheltered aspect).

These assessments must be taken in the context of the changed nature of the soils resulting from initial clearing, ringbarking, grazing and frequent fires, which have led to substantial erosion to the upper soil horizons (Fields personal communication) and lack of comparable data prior to ringbarking.

The following sub-section details the results.

Bredbo Pair 1 - pulled/chained sites vs. control
Our observations suggested that this method of harvesting caused minimal disturbance to landscape function and that by retaining fallen timber *in situ* there was minimal leakage of soil and nutrient resources from the site. Subsequent analysis found no significant differences for the soil surface condition indicators of stability, infiltration and nutrients between the first pulled/chained site (15yr 1) and the control (15yr 1 control) . The large volumes of fallen logs, shed bark, leaf and twig materials and largely intact ground cover meant high levels of resources were captured within the sites.

Bredbo Pair 2 - pulled/chained sites vs. control
These sites did not follow the above expected trend. There were significant differences between the chained and control site indicating that pulling/chaining at this site had not led to strong resource regulation. Subsequently it was discovered that initial clearing was by chaining but limited to the outside edges of the site. The internal area of the site was cleared by bulldozer with intensive soil

surface disturbance leading to these significant differences detected between control and harvested sites.

Bungendore Pair 1- pulled/chained sites vs. control
Initial observations at Bungendore (MCthinned1) suggested thinned sites were not as functional as control sites. It was hypothesised that as two thirds of the trees had been removed only very limited amounts of residual fine or coarse woody debris remained to provide landscape patches, and site capacity to capture water and nutrients was reduced. This was confirmed when analysis demonstrated significantly reduced functionality for two of the three soil surface condition indicators, i.e. infiltration and nutrient cycling.

Bungendore Pair 2 - pulled/chained sites vs. control
However, the LFA on the second thinned site at Bungendore (MCthinned2) found no significant difference between it and the control site (MCno_thin2). In fact, the thinned site had slightly higher function values than the control site. This may be due to insufficient landscape stratification and higher standard errors than normally accepted. However, it also may be due to some inherent resilience to the thinning treatment, conferred by the site's physical and biological resources. The higher amounts of shrub and groundcover, the deeper and less hydrophobic soils, flatter slopes and protected aspect from solar radiation, winds and temperatures (Semple 1994) may have buffered the site against loss of landscape function after the thinning treatment removed protective canopy and fallen woody debris.

9.10 Further silvicultural management considerations for biodiversity

9.10.1 Variability of forestry attributes within dry sclerophyll forest

It would be inappropriate to recommend any firewood harvesting in dry sclerophyll forests without firstly undertaking the appropriate forest mensuration surveys to specifically locate, on a stand basis, those areas with the high stem and basal areas which might provide for both commercially and ecologically sustainable harvesting. The spatial variation in forestry attributes e.g. diameters and stems ha^{-1}, reflect the disturbance history of the forest stands and the varied landscapes they are situated within. The forest types considered most suitable for harvesting will typically be those stands which have been relatively undisturbed for 50 to = 100 years, occurring on the upper exposed slopes. Generally these areas are characterised by dense, even-aged stands, with limited regeneration, other-aged cohorts or older trees. They have substantial canopy cover, low habitat complexity and low levels of shrub and ground cover. The generally low to medium bird, small mammal and plant species richness observed in these sites reflect these attributes.

Forest measurements across these forests indicate that older trees greater than 60cm in diameter tend to be sparsely distributed, with around 1-2 trees ha^{-1}. Although the trees with largest diameters are not necessarily the most suitable as wildlife habitat, as trees with multiple hollows and dead branches in the crown are used more extensively (Gibbons et al. 2002), the probability of hollow development is greater as a tree increases in diameter over time. There also exist trees estimated to have been there since before European settlement, occurring in stands of 1-10 ha. These remnants are often found on upper slopes and ridges, but are uncommon. Retaining old trees in logged forests is critical for wildlife because they reduce the adverse effects of logging, for example on birds (Smith 1985).

9.10.2 Forest stands and the self-thinning rule

It is appropriate at this stage to consider the "self-thinning rule", which describes a density-dependent upper boundary of stand biomass for even-aged stands in a given environment. The upper limit appears to be a result of intraspecific competition for light, water and nutrients. The *maximum density* occurs where a stand undergoes substantial and continuing mortality induced by competition (Bi et al. 2000). Although subject to debate as to when and how maximum density is reached, the assumption is that, after some unknown time, undisturbed forests stands will reach the point where they naturally begin to self-thin. Some indication of the time taken may be gained from native eucalypt forests managed for silviculture in Tasmania, where the point of maximum density is reached somewhere between 80-120 years, and self-thinning will reduce an initial natural regeneration of trees of several thousand trees ha^{-1} down to 100 trees ha^{-1}. If this timing was similar for the dry sclerophyll forests of the Southern Tablelands, the case study sites in the 50 to =100 year old category should be approaching the maximum density point, where self-thinning will begin to significantly impact on the stem densities. Harvesting in these sites may possibly create a forest stand similar to that after self thinning.

Are these forest sites approaching "maximum density"?

Field and Banks (1999) suggest that typical basal areas of dense regrowth stands at maximum density at the Bungendore site is 36± 2m², although across all case study sites between 30-35m² is common. There are typically between 300 and 3,500 stems per hectare in these areas, with diameters ranging between 20 and 80 centimetres. For these sites, stand basal area could be used as a guideline for harvesting, taking into account that the basal area of a stand of a given age varies with species for a given site and with site for a given species. However, for certain species, the basal area of stands on particular sites may be reasonably constant over a considerable period of the development of the stand, particularly as the stand approaches maturity. In such circumstances, stand basal area is a good measure of the maximum occupancy of the site and thus of stand density. Stand basal area is widely used in the management of even-aged stands for a number of reasons, i.e. it is a practical index of stand density; it is easily measured, it is the natural base for deriving stand volume and volume increment and basal area increment are usually well correlated (Brack 1999).

Estimating the proportion of forest occupied at "maximum density"

The current extent of forest mapping available for spatial analysis is not yet adequate for this level of discrimination without extensive ground truthing. However it is possible to make an approximate estimation from one of the case study sites, Murrumbateman, based on field knowledge of the property. The Picaree Hill Conservation Project Area, an area fenced and mapped with all five Murrumbateman field study sites contained within its boundaries, is 452ha. The areas of native and/or pasture grassland, younger and older forest stands, ie. other than the densely stocked 50-100 year stands, have an area of around 118 ha or 26% of the total area. This result then means approximately 74% of forest area contain densely stocked stands. A neighboring adjoining forest area of 96ha, with densely stocked 50 to =100 year stands, contains approximately 18ha or 19% of forests older than 150 years, leaving around 81% as densely stocked stands. Consequently there are potentially reasonably significant areas of dry sclerophyll forest in this region that could be considered for harvesting under the green-wood and dead-wood scenario.

9.11 Conclusions

In Section 8 (green-wood scenario), we suggest that it is feasible to meet a long term demand for firewood exclusively by thinning live trees from only those forests away from major water courses, on shallow slopes less than 15°, and from forests patches at least 100 hectares in size in regions with at least a 30% forest cover. Our modelling suggests that an exclusive harvest of live trees would eventually create mixed age stands and allow for substantial accumulation of coarse woody debris. Averaged across the entire modelled area, loads of woody debris would vary between 15-20 tonnes per hectare over the next 100 years. This would result in 5-7 times greater post-harvesting loads of coarse woody debris than under the dead-wood scenario which on average left only 3 tonnes per hectare of woody debris after harvest of dead standing and fallen timber.

In this section (9), we sought field evidence to substantiate our green-wood modelling. We sought field sites that demonstrated our modelled expectations that selective harvesting of live trees can be followed by extensive tree regeneration events, can promote mixed age forest structure and can allow for significant accumulation of coarse woody debris.

We succeeded in finding a few case study sites in dry sclerophyll forests on the Southern Tablelands of NSW where live trees have been harvested for a range of products including firewood. The ecological impact of such harvesting depends on the method of live-tree harvest and site characteristics.

We suggest that the impact of harvesting live trees for firewood can be minimised when the harvesting operation leads to greater structural complexity of the forest stand. Harvesting of forests can increase structure when it leads to tree regeneration which in time will create mixed age stands. Selective harvesting can increase structural complexity when it leads to greater loads of coarse woody debris left after the harvesting operation. Patch-scale harvesting of live trees can increase structure when the opening of the forest canopy stimulates the establishment of a greater density and diversity of understorey shrubs, grasses, forbs and orchids.

Our case studies, and many other studies, have shown that there are more wildlife species across a range of taxa in structurally complex forest stands compared to structurally simple forests. For example, our surveys found a greater species richness of woodland birds at sites with higher shrub cover. Our studies, and others, have also shown that structurally complex forests “leak” less water, nutrients and soil than structurally simple forests with little ground cover.

The harvesting of live trees for firewood will have adverse environmental impacts if it perpetuates even aged stands with few old trees with hollows. Harvesting will have adverse impacts if coarse woody debris is also harvested or burned. The effects of harvesting will be adverse if they expose the soil surface to excessive rainfall erosion and prevent the regeneration of trees, shrubs and grasses.

Our case studies suggest that the extensive cover of privately owned dry sclerophyll forests in the upper catchments of the MDB have significant potential for improved management to enhance both biodiversity and commercial values. Few of these forests exhibit “pristine” old growth characteristics. Every case study site exhibited signs of considerable disturbance from various harvesting and partial clearing events over the past 150 years. We suggest that those forest stands that have regenerated from what amounted to clear felling 50-100 years ago, have the greatest potential for patch-scale harvesting of live trees for timber products, including firewood. These medium-age stands are fully stocked and have little structural complexity, particularly little understorey and low levels of coarse woody debris (<10 t h^{-1}). The few old trees remaining in these mostly even-aged stands should be retained for their habitat values, particularly because of the high probability that they have hollows.

The method of harvesting of live trees by pulling/chaining relatively narrow belts of trees along the contour shows potential to allow for regeneration and maintenance of landscape function. We suggest that allowing the fallen green trees to dry (cure) *in-situ* for a number of years reduces the impact of post-harvest soil disturbance. *In-situ* drying maximises the amount of timber material lying on the ground and the resulting obstruction of fine-scale movement of soil material and litter, the capture of which provides ideal micro-sites for tree, and under story regeneration.

Further management and policy implications of our findings from these case study surveys are discussed in Section 11.