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## **Dynamics of tree diversity in undisturbed and logged subtropical rainforest in Australia**

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### **Abstract**

In subtropical rainforest in eastern Australia, changes in the diversity of trees were compared under natural conditions and eight silvicultural regimes over 35 years. In the treated plots basal area remaining after logging ranged from 12 to 58 m<sup>2</sup> per ha. In three control plots richness differed little over this period. In the eight treated plots richness per plot generally declined after intervention and then gradually increased to greater than original diversity. After logging there was a reduction in richness per plot and an increase in species richness per stem in all but the lightest selective treatments. The change in species diversity was related to the intensity of the logging, however the time taken for species richness to return to pre-logging levels was similar in all silvicultural treatments and was not effected by the intensity of treatment. These results suggest that light selective logging in these forests mainly affects dominant species. The return to high diversity after only a short time under all silvicultural regimes suggests that sustainability and the manipulation of species composition for desired management outcomes is possible.

**Key words:** Australia, disturbance, species richness, logging, rainforest, silviculture, subtropical, tropical

## **Introduction**

Various systems of rainforest silviculture have been devised which have had as a goal the sustainable production of particular timber species (Baur, 1964, 1991, 2001; Dawkins and Phillip, 1998). More recent interest is in whether such systems can provide sustained productivity and income as well as effectively preserve biodiversity. Silvicultural systems designed to maintain the production of desirable timber species have been either monocyclic or polycyclic, and included treatments to remove non-commercial species, as well as treat vines and weedy species (Baur 1964, Lamprecht, 1989, Whitmore 1990). Currently there is an emphasis on designing systems which are low impact and minimise damage to the retained stand, although the idea of Reduced Impact Logging (RIL) is not new. Baur (1964) emphasized the importance of avoiding “careless felling and snagging”. Burgess et al. (1975) analyzed the percentage basal area damaged by different logging treatments at the site that is the subject of this study. Further, RIL should ideally be only a part of coherent productive silvicultural systems that are based on an understanding of the ecophysiological and regeneration requirements and of key species (de Graaf, 2000; Wadsworth, 2001).

Sustainable forest management systems have been criticised as being uneconomic, yielding little income on a per-hectare basis. Thus an alternative of 1) a single entry targeting high value species and 2) conversion to park or reserve status has been proposed (Rice et al., 2001). Such issues are complex, given that sustainability is as dependent on social and economic factors - including the number of commercial species found in a given forest type - as on biological and physical factors (Reid and

Rice, 1997; Rice *et al.*, 1997; Cannon *et al.*, 1998; Rice *et al.*, 2001; Montagnini *et al.*, 2001).

Where any intervention is planned an understanding of forest processes that maintain biodiversity will be crucial. While many theories have offered insight into important processes (Janzen and Vazquez-Yanes, 1970; Hubbell 1979, 1998, 2001; Connell, 1970, 1978; Connell *et al.* 1984) probably the most important from a management intervention perspective is the effect of the intensity and frequency of disturbance events. Disturbance is widely regarded as a primary mechanism for maintaining the diversity found in tropical and subtropical rainforests (Connell, 1970, 1978; Connell *et al.* 1984). Disturbance regimes are also important in the evolution of tree species life history characteristics (Sheil and Van Heist, 2000; Sheil and Burslem, 2003).

Several studies have shown a disturbance event can raise tree diversity by facilitating the introduction of pioneer species (Nicholson *et al.*, 1990; Plumptre, 1996; Cannon *et al.*, 1998; Sheil, 1998). Nicholson *et al.* (1988, 1990) examined partially logged sites in rainforest in north Queensland and concluded that the after-logging response on all but lightly logged plots was for there to be an increase in tree species numbers and the diversity index. Their argument that logging assists in maintaining diversity was subsequently debated (Saxon, 1990; Nicholson *et al.*, 1990), with the opposing point of view being that natural disturbance levels maintain highest diversity. Sheil and Burslem (2003) raise the larger and more complex question of how diversity is maintained. This requires consideration of the spatial and temporal arrangement of disturbance regimes. For example, Ferreira and Prance (1999) concluded that 40 years was sufficient time for secondary forest to return to its original structure but not

to achieve the original species richness of primary forest, although on a landscape level there was an increase in total species diversity with the presence of the two forest types.

This study compares changes in patterns of tree diversity under various silvicultural regimes in subtropical rainforest over 35 years. We consider what effects one-time logging events and associated silvicultural systems have on tree diversity. In particular, we compare the effect of silvicultural systems characterised by a range of logging intensities. The patterns of dominance and diversity in this rainforest are described and possible explanations for those patterns are offered.

### **Site description and history**

Subtropical rainforest is found in southeast Queensland and northern New South Wales, most of it having been cleared but with a remaining stand in Lamington (Queensland) and Border Ranges National Parks (New South Wales) (Floyd, 1990). In New South Wales, Baur (1991) estimated that about three quarters of the original one million ha of all types of rainforest was cleared for agriculture. In 1985, there was 100,000 ha of subtropical rainforest, mostly on public lands. Of the four types of rainforest in New South Wales (subtropical, dry, warm temperate, and cool temperate) subtropical is the most complex. It resembles lowland tropical rainforest and is characterised by mixed species composition, no clear dominance by one tree species, and is floristically related to south-east Asian rainforest. Vines, epiphytes, and trees with buttresses are common. Webb *et al.* (1972) argued that this was true “virgin” forest, in that aboriginal people in the area did not practice agriculture per se, or

systematic tree felling. They also believed that fires had not taken place in these rainforests, so that the main disturbances would come from individual tree falls, wind and cyclones.

At the time the experiment was established the area was part of Wiangaree State Forest, just south of the border with Queensland. The area has subsequently become a national park (Border Ranges National Park). Within the National Park elevation ranges from 600-1200 m elevation above sea level, annual rainfall is approximately 3000mm and subtropical rainforest is the dominant vegetation type. The soils are fertile, being classified as kraznozems or ferrosols (Isbell, 1998) derived mainly from tertiary basalt flows. Turner *et al.* (1989) studied biomass accumulation in the undisturbed forest at this site over 16 years and estimated that the steady-state above-ground biomass for this forest is 350 tons ha<sup>-1</sup>.

The forests are characterized by an emergent canopy of large overstorey trees greater than 30 m in height, many of which are buttressed. Some of the most common species are *Doryphora sassafras* (Monimiaceae), *Caldcluvia paniculosa* (Cunoniaceae), *Sloanea woosii*. (Elaeocarpaceae), *Heritiera* (*Argyrodendron*) *actinophylla*, *Heritiera trifoliata*, (Sterculiaceae), and *Geissois benthamii* (Cunoniaceae). Floyd (1990) divided the subtropical rainforests of NSW into four alliances and 20 suballiances. The Border Ranges forest is classified as Suballiance 11, *Caldcluvia- Cryptocarya erythroxylon-Orites-Melicope octandra-Acmena ingens* in the *Caldcluvia* (corkwood) alliance. Detailed descriptions of tree species are found in Floyd (1989). The area was logged in the period 1965-1975, with most treatments described in this study taking place in 1965-66. Up to 17 species were typically taken

out of these forests. The volume of just three of these – *Argyrodendron trifoliolatum*, *Geissois benthami* and *Araucaria cunninghamii*– constituted nearly 50% of the amount cut (Shugart et al. 1980). Previous papers written on growth, composition and ecological aspects of these plots include Burgess *et al.* (1975), Pattemore and Kikkawa (1975), Shugart *et al.* (1980), Horne and Gwalter (1982), Horne and Hickey (1991) and Turner *et al.*, (1989).

## Methods

In 1965 a set of 11 plots were established in Wiangaree State Forest to assess the impact of various silvicultural systems on subtropical rainforest. Plots were of various sizes, up to 3.9 ha each. In each a central experimental sub-plot of 60.4 x 60.4 m<sup>2</sup> (0.3648 ha) was laid out. Three of the plots were left as controls. Eight of the plots were treated with various silvicultural systems and logged. The intensity of logging ranged from removal of 70% of the overstorey basal area to light single tree selection. The systems included practices from New South Wales and Queensland in Australia current at the time and silvicultural systems developed in the tropics and studied by George Baur (Baur, 1964). Logging treatments took place at various dates between 1965 and 1975, in combination with other silvicultural treatments of climber cutting, poisoning and enrichment planting. The treatments were:

1. *Selective logging – NSW*

Retain 50% of the canopy by logging of merchantable stems and poisoning of non merchantable stems

2. *Heavier selective logging – NSW*

Retain 50% of the canopy by logging of merchantable stems

3. *Group Selection System – Queensland*

Cut vines and undergrowth. Log all merchantable stems over 60cm dbh except where required as seed trees, all damaged stems unlikely to survive 20 yrs, surplus smaller stems in dense patches and low value species overtopping preferred saplings. Poison unmerchantable remnants. Liberate regeneration after 5 years.

4. *Clear cut system – Malaya*

Log all merchantable stems poison all non-merchantable stems. Liberate regeneration after 5-10 years.

5. *Pre-exploitation shelterwood – Nigeria*

Cut vines and poison smaller un-merchantable stems to aid regeneration. Log all merchantable stems 5 years later. Poison non-merchantable, coppice and damaged stems.

6. *Logging with enrichment*

Log all merchantable stems. Remove non-merchantable understorey to form a shelterwood. Enrichment plant with fast growing species. Over 5 years progressively poison non-merchantable stems. Liberate regen after 5 and 10 yrs.

7. *Improvement treatment – Congo.*

Cut vines and poison all non-merchantable stems. Log all stems 10 years later.

8. *Post-exploitation shelterwood – Trinidad*

Cut vines. Log all merchantable stems one year later. Poison selected non-merchantable stems to form shelterwood. Liberate regeneration at least once in 5 years after logging. Poison remaining non-merchantable stems 6 years after logging.

The timing of all silvicultural treatments in relation to the logging date as well as the intensity of the logging events are detailed in Table 1. For further discussion on the treatments the reader should refer to Baur (1964).

All stems  $>10\text{cm}$  dbh were measured at different dates between 1965 and 1999 with assessments being done in 1970/71, 1973, 1975/76, 1980/81, and 1987.

Buttressed trees were measured at a reference height of 2, 3 or 4 m height above the ground to avoid the buttressing. At each measure recruited trees that had attained 10cm dbh were recorded. The three control plots were last remeasured in 1995 and the treatment plots in 1999.

The study suffers from limitations common in studies of this forest type where resources were limited. The plot size is smaller than ideal and the measurements were not taken annually (Clark and Clark, 1999). Although there was only one plot established with each treatment there are eight treatments of varying intensity allowing comparisons across the range of intensities to be made. To assess the limitations of plot size the species area relationships are examined. Species area relationships were calculated in the six plots with pre-logging measurements. The mean number of species was calculated for all possible combinations of one, two, three, four, five and six plots. A further limitation is that before logging in 1965 there was a measure of all trees  $>20\text{cm}$  dbh in the gross treatment plot. Individual trees were not identified and it was not comparable to the measure of all trees  $>10\text{cm}$  dbh in the smaller measurement plot post logging.



## Results

Original pre-logging basal areas were high, averaging 51.5 m<sup>2</sup> in the control plots, ranging up to 65.8 m<sup>2</sup> in Treatment 5, the pre-exploitation shelterwood system (Table 1). Reductions in percentage of basal area harvested ranged from 26% in Treatment 1 to 77% in Treatment 7. Retained basal areas in the treated plots ranged from 12 m<sup>2</sup> per ha in Treatment 8 to 58 m<sup>2</sup> per ha in Treatment 1. These are considerably higher than the 10 m<sup>2</sup> per ha often recommended for rainforest silviculture (Baur, 1964; de Graaf *et al.*, 1999).

We recorded no large increases in basal area, unlike de Graaf *et al.* (1999), who documented an increase of 10-18 m<sup>2</sup> per ha over the first six years after silvicultural treatments. Rather the increments were more comparable to those found in their work in Surinam in the period 6-21 years after treatment, where they recorded an increment of 0.6m<sup>2</sup> per ha per year. Throughout most of the treatments basal area increments were remarkably similar to each other, through time averaging 0.48 m<sup>2</sup> (standard deviation =0.17) increment in basal area per year from 1980 to 1999. Over this 19-year period the greatest increment was in the heavy New South Wales treatment (0.75 m<sup>2</sup> per year) and the lowest in the Malayan system (0.32 m<sup>2</sup> per year). By 1999, basal areas had returned to or exceeded pre-treatment levels in treatments 1-4. If present rates of basal area increment were to continue, projected achievement of initial basal areas would occur in the pre and post exploitation shelterwood treatments 93 and 95 years after logging, respectively (Smith and Nichols, 2000).

Species area relationship for the three control and three treatment plots where data were collected before logging (Table 1), showed the mean number of new species

found in a second plot was 14 , then 8 in a third plot and so on (Fig 1). In 2.2 ha of unlogged forest 71 species were recorded compared to 91 in 4.4 hectares including disturbed plots. The relationship shows the steepest part of the curve is from zero to the first plot after which the decline in the number of new species recorded is more rapid. The fitted relationship for the six plots (Fig 1) predicts 83 species in 4 ha compared to the 91 that were recorded over the duration of the experiment.

Richness is presented as number of species per plot and number of species per 144 stems (Figure 2). This (144) was the mean number of stems per plot over the duration of the whole experiment. Richness in the control plots was relatively stable over the 35 year period, both in terms of the number of species per plot and per 144 stems. There is considerable variation in the relationship between species per plot and per 144 stems within the control plot. Control 2 had more stems and more species per plot compared to control plots 1 and 3 (Figure 2). This is due to large numbers of several dominant species.

In the treated plots changes in species richness were correlated with the intensity of silvicultural treatment. In all but two treatments the reduction in stems from logging caused a decrease in species per plot and a corresponding increase in species per 144 stems. In two of the least intense treatments (Light and Group Selection) stem numbers stayed fairly constant over time, as did species richness (both per plot and stem). In all treatments species richness had returned to similar values to the control plots 35 years after logging and in most by 22 years after logging. The time taken for richness to return to these levels was not correlated to the intensity of the treatment. In

all eight silvicultural treatments there were more species per plot found at the end of the measurement period than at the first measurement after logging.

As expected, a few species were dominant and many more were uncommon to rare. Species dominance in Control 1 was typical of the control plots (Figure 3) and followed a pattern often seen in rainforests in a “dominance-diversity curve” (Webb and Peralta, 1998). Basal area dominance by the ten species with highest basal area in each plot varied from 65 to 93% of total plot basal area. Of 50 species that occurred in the three control plots, only 16 species were found in all three plots. Overall there were 78 species found on the four ha of plots over the thirty-five year history of measurements. Thirty-five species were represented by fewer than four trees in the four hectares of the plots, while for fifteen species there was only one tree >10cm in diameter over the entire four hectares.

For each plot species were classified according to whether they were: absent in 1965 then present in 1999, and also whether present in 1965 then absent in 1999 (Table 3). The results are only strictly comparable for plots where prelogging data are available (3 of the 4 heaviest disturbance treatments), however all treatments are included in the table for comparison (Table 3). Although the mean number of species in the controls was lower for both absent and present species the difference was not significant (t test of 3 controls against 3 treated plots). The results show no particularly strong pattern with treatment. A moderate treatment (Queensland Group Selection, with 32 m<sup>2</sup> left after logging), had the greatest number of species found in 1999 that were not found in 1965 (19). With the possible exception of the Enrichment treatment, the numbers

of species absent from control plots is very comparable with logged plots by the end of the experiment.

At the scale of the whole experiment (over all the plots), 13 species were not present in any plot in 1965 and present in 1999. Of these 7 appeared only in treated plots and not in control plots. Of the 6 species that were no longer present in 1999 having been present in 1965, 5 were absent from treated plots only.

## **Discussion**

This study showed two main characteristics of patterns of change in tree species diversity in relation to logging intensity. Firstly, the time for diversity to return to pre-logging levels was not related to disturbance intensity and secondly there appears a disturbance threshold below which diversity is not affected. The time for diversity to return to pre-logging levels is similar in all the plots and was unrelated to silvicultural system or logging intensity (Figure 2). The similar time for recovery of diversity in all plots suggests that most species regenerate after a disturbance or are already present as seedlings. The implication is that there is considerable scope for manipulation of species composition in the years after disturbance by stand tending. If no further interventions are made then species dominance within the stand will continue to change over a long period. Under this scenario disturbances will have little effect on species diversity at the scale of the plots if the time between disturbances is great enough. A further implication is that the spatial and temporal arrangement of disturbances will be more important than the intensity at a small scale.

That is, chance will play the most important role in maintaining diversity (Brokaw and Busing, 2000).

The rates of species turnover (species local extinctions and colonisations) in the treated plots was similar to the controls (Table 3). At the scale of the whole experiment, seven species became locally extinct, of these six from treated plots. There may be more species that became absent during the logging events that would bias this comparison, however where pre-logging data were available in 3 of the 4 heaviest treatments there was no significant difference.

The question of an appropriate scale is important in managing such diverse systems. The size of the plots in this experiment are too small, however the species area curve was still increasing after 4 hectares (Figure 1). Given that it may not be possible to express an appropriate area based on a species area curve (Williamson, et al, 2001), it may be useful to use dominance diversity relationships within the forest as a guide for management.

The level of dominance of the common species could be used as an indicator for appropriate levels of logging intensity. The appropriate level could be the point at which richness is effected. For example, in this study two of the least intense treatments, light and group selection, had very little effect on diversity. This is because up to this intensity common species are taken (assuming common species are merchantable - see below). As logging intensity increased so the species per stem increases sharply and becomes correlated with the number of stems, indicating that rare species are being removed. Similarly as the species richness recovers so the

richness per number of stems decreases indicating that the regeneration is less diverse than the retained stand, ie that it is dominated by few species. If common species are targeted the effect on rare species (and species richness) is minimised. In this study common species were not targeted. It may be possible to extend the threshold where diversity is affected by deliberately selecting only common species. This approach may not be appropriate in forests of higher diversity where few species could be described as common.

Such an approach would rely on assumptions about the adaptations of rare and more common species and on the commercial value of common species. Although the mechanisms are not well understood (Connell 1978), rare species have a high chance of local extinction and therefore must have reproductive and dispersal mechanisms for survival. Similarly the mechanisms limiting the dominance of more common species must be assumed to be density related and that these species will recover to higher densities (Connell 1978).

Some of the silvicultural systems used in this experiment were designed for either conversion of forest to maximise production of uncommon merchantable species or to maintain that production once a forest was converted. In Australia at least, management aims have changed greatly, however timber production, rehabilitation or conservation of target species are management outcomes for which elements of these silvicultural systems would be useful. In particular the independence of time to recovery of diversity indicates that post logging silvicultural treatments need to be undertaken to have most effect.

Perhaps the most important characteristic that has changed is the merchantability of different species within a forest stand. The more species that are merchantable and can be utilised the greater the management flexibility possible in stand manipulation. While there have been important developments in the use of reduced impact logging, this doesn't completely address the issue of sustainability. An early model of the impact of logging on this forest predicted a decline in some species (notably *Caldcluvia*, the most common species) and an increase in others (Shugart et al 1980). Deliberate management of species composition will require elements of silvicultural systems similar to those used in this trial such as vine cutting, liberation of regeneration and enrichment planting if management aims to maintain levels of certain species.

Some of the diversity post disturbance will be attributable to the transient increase in diversity found post disturbance (Shiel, 1999, Nicholson, et al, 1988, 1990 ). However, the species area relationship for unlogged plots, the low species loss and colonisation rates in treated plots (Table 3), and the importance of *C. paniculosa* and *S. woolsii* (see below) suggest that this forest is recovering from a natural disturbance in the recent past (relative to stand dynamics processes). For example, rainforest in SE Queensland not far from the site of this study had low diversity of species compared to lowland tropical rainforest and adult basal area was dominated by *Sloanea woolsii* (Debski et al. 2000). This was attributed to a large-scale disturbance event as the dominance of this species indicates a forest that has not yet attained equilibrium species composition. The most dominant species at this site was *Caldcluvia paniculosa*. It is known locally as Corkwood and identified as a pioneer

species common on rainforest margins, sometimes persisting as canopy tree (Williams *et al.*, 1984).

Other pioneer species took varying amounts of time to appear in the >10cm diameter class, taking into consideration the fact that measurements were not recorded annually. *A. melanoxylon* began to appear 10 to 15 years after the treatments took place. The general tendency for *A. melanoxylon* is the same in Treatments 2 through 8, with the percentage of total stems increasing up until the last measurement, thirty-five years after the beginning of the experiment. This species appears consistently and provides a good indication of recent disturbance. *Alphitonia excelsa* (and other species) were much less common and followed a different pattern, appearing in only three of the treatments.

Rice *et al.* (2001) suggest that it may not be realistic to expect that sustainable forestry systems be implemented in the world's remaining rainforests. They argue that the components of such systems that make them sustainable – reduced impact logging with careful directional falling and designated skid trails – also make them prohibitively expensive in most situations. However, this is more an issue of economic and social frameworks than of biology and such an argument rests on economic assumptions that have been questioned (Pearce *et al.* 2003). As Sheil and Van Heist (2000) point out, equating “biodiversity” with species richness is far from adequate. We do not understand what, if any, are the keystone species, if in fruiting trees, birds, bats or microorganisms, in this forest type. Further, we assess only a one-time harvest this forest (including two treatment entries in some cases) rather than many ongoing impacts. Even if logging is done carefully, using directional falling



and designated skid road techniques, repeated logging and silvicultural treatments could be expected to create some damage to residual trees each time they take place. Nevertheless, realistically not all rainforests can be put into reserves. Vanclay (1992), for example, has argued that the working forest has a place in conservation, along with pure forest reserves. Our findings here suggest that periodic entries, removing moderate numbers of trees, may not reduce the diversity of tree species in the long run. We can even more strongly argue that a moderate one-time logging event, which produces some income, may not cause undue harm.

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## List of figures

Figure 1. Species-area curve, based on six 0.3648 ha plots before silvicultural treatments applied. The fitted relationship is compared to the total diversity found over the 4 ha and 35 years of the experiment.

Figure 2. Total number of trees >10cm dbh per plot, number of species per 144 trees and number of species per plot.

Figure 3. Dominance-diversity curve for control plot 1, showing structure typical of subtropical rainforest at Border Ranges National Park.

Table 1. Treatment details including pre and post logging interventions, the date of logging as well as the remaining basal are after logging.

Silvicultural treatment	pre logging treatments			Logging Year - LY -	BA (m <sup>2</sup> /ha) after logging	post logging treatments		
	<i>Regen assess</i>	<i>climber cutting</i>	<i>poisoning</i>			<i>poisoning</i>	<i>liberate regen</i>	<i>enrichment planting</i>
1. NSW selective logging				1965	58	> LY		
2. Heavier selective logging				1965	34	> LY		
3. Queensland group selection		< LY		1965	32		LY + 5	yes
4. Malayan clear cut	yes			1965	29		LY+5 to 10	
5. Nigerian pre-exploitation shelterwood		LY - 5	LY - 5	1970	24	> LY	LY + 5	
6. Logging with enrichment				1965	23	LY to + 5	LY + 5 & 10	yes
7. Congo improvement treatment		LY - 10	LY - 10	1975	15			
8. Trinidad post-exp. shelterwood		LY - 1		1966	12	LY + 1, 6	LY to + 5	

Table 2. Species frequency and tentative species guilds. Authorities from Floyd (1990) and Species guilds from Floyd (1990) and Kooyman (1996). \*\*\* present in all three controls. (+) -overstorey. Guilds: P=Pioneer ES=Early Secondary LS=Late Secondary M=Mature. NB Table needs final revising and final edits.

Species	Family	Frequency	Overstorey	Guild
Baloghia inophylla (G. Forster) P. Green	Euphorbiaceae	***		
Caldcluvia paniculosa (F. Muell.) Hoogl.	Cunoniaceae	***	(+)	M
Cinnamomum oliveri Bailey	Lauraceae	***	(+)	M
Cryptocarya erythroxylon Maiden & Betche	Lauraceae	***	(+)	M
Diospyros pentamera Woolls & F. Muell	Ebenaceae	***	(+)	M
Doryphora sassafras Endl.	Monimiaceae	***	(+)	M
Geissois benthamii F. Muell	Cunoniaceae	***	(+)	M
Heritiera actinophylla (Bailey) Edlin	Solanaceae	***	(+)	M
Heritiera trifoliolata F. Muell	Solanaceae	***	(+)	M
Orites excelsa R. Br.	Proteaceae	***	(+)	M
Planchonella australis (R. Br.) Pierre	Sapotaceae	***	(+)	M
Polyscias elegans (C. Moore & F. Muell.)	Araliaceae	***		P
Quintinia sieberi DC.	Escaloniaceae	***		M
Randia benthamiana F. Muell	Rubiaceae	***		M
Stenocarpus sinuatus Endl.	Proteaceae	***		M
Syzygium crebrinerve (C.White)	Myrtaceae	***	(+)	M
Acacia melanoxydon R. Br.	Mimosaceae			P
Acmena hemilampra (F. Muell. Ex Bailey)	Myrtaceae			
Acmena ingens (F. Muell. Ex C. Moore)	Myrtaceae			M
Acmena smithii (Poir.) Merr. & Perry	Myrtaceae			M
Acronychia suberosa C. White	Rutaceae			
Akania lucens (F. Muell.) Airy Shaw	Akaniaceae			M
Alangium villosum (Blume) Wangerin.	Alangiaceae			
Alectryon subcinereus (A. Gray) Radlk.	Sapindaceae			
Alphitonia excelsa (Fenzl) Reiss. Ex Benth.	Rhamnaceae		(+)	ES
Araucaria cunninghamii Aiton ex D. Don	Araucariaceae		(+)	LS
Archidendron grandiflorum (Soland. Ex Benth.) Nielsen	Mimosaceae			
Austromyrtus sp. Aff. lasioclada	Myrtaceae			
Beilschmiedia elliptica C. White & Francis	Lauraceae			M
Beilschmiedia obtusifolia (F. Muell. Ex Meissner) Muell.	Lauraceae			M
Brachychiton acerifolius F. Muell.	Sterculiaceae		(+)	LS
Cinnamomum virens R. Baker	Lauraceae			M
Citronella moorei (F. Muell. Ex Benth) Howard	Icacinaceae			M
Cryptocarya glaucescens R. Br.	Lauraceae			M
Cryptocarya obovata R. Br	Lauraceae		(+)	M
Daphnandra micrantha (Tul.) Benth.	Monimiaceae		(+)	M
Decaspermum parviflorum (Lam.) A.J. Scott	Myrtaceae			
Dendrocnide excelsa	Urticaceae			ES
Denhamia pittosporoides	Celastraceae			
Diploglottis australis (G. Don) Radlk.	Sapindaceae			ES
Duboisia myoporoides R. Br.	Solanaceae			P
Dysoxylum fraserianum (Adr. Juss.) Benth.	Meliaceae		(+)	M
Dysoxylum muelleri Benth.	Meliaceae			LS
Ehretia acuminata R. Br.	Boraginaceae			LS
Elaeocarpus kirtonii F. Muell. Ex Bailey	Elaeocarpaceae			LS
Emmenosperma alphitonioides	Rhamnaceae			LS
Endiandra crassiflora	Lauraceae			
Endiandra discolor	Lauraceae		(+)	M
Endiandra muelleri Meissner	Lauraceae			
Euodia micrococca F. Muell.	Rutaceae		(+)	
Eupomatia laurina R. Br.	Eupomatiaceae			M
Ficus watkinsiana Bailey	Moraceae		(+)	M
Gmelina leichhardtia F. Muell	Verbenaceae			
Guoia semiglaucula (F. Muell.) Radlk.	Sapindaceae			ES
Halfordia kendack (Montr.) Guillaum.	Rutaceae		(+)	M
Halfordia trifoliolata	Rutaceae			
Helicia glabriflora F. Muell	Proteaceae			
Litsea reticulata (Meissner) F. Muell	Lauraceae			M
Maytenus bilocularis (F. Muell.) Loes.	Celastraceae			
Melicope octandra F. Muell Druce	Rutaceae			M
Mischocarpus anodontas (F. Muell.) Radlk.	Sapindaceae			
Neolitsea australiensis Kosterm.	Lauraceae			
Neolitsea dealbata (R. Br.) Merr.	Lauraceae			
Pittosporum rhombifolium Cunn. Ex Hook.	Pittosporaceae			LS
Polyosma cunninghamii Bennett	Escalloniaceae			
Polyscias murrayi (F. Muell.) Harms	Araliaceae			P
Rhodamnia rubescens (Benth.) Miq.	Myrtaceae			ES
Sarcomelicope simplicifolia (Endl.)	Rutaceae			M
Sarcopteryx stipata (F. Muell.) Radlk.	Sapindaceae			ES
Sloanea australis (Benth.) F. Muell	Elaeocarpaceae		(+)	M
Sloanea woollsi F. Muell	Elaeocarpaceae		(+)	M
Stenocarpus salignus	Proteaceae			M
Synoum glandulosum (Smith) Adr. Juss.	Meliaceae		(+)	LS
Synoum watkinsiana	Meliaceae			
Syzygium coryanthum (F. Muell.) L. Johnson	Myrtaceae			
Toona australis (F. Muell.) Harms.	Meliaceae		(+)	LS
Wilkiea huegeliana (Tul.) DC.	Monimiaceae			M
Zanthoxylon brachyacanthum F. Muell	Rutaceae			



Table 3. Number of species which appeared or disappeared, between 1965 and 1999 in each of the plots. Note that controls are only strictly comparable to treatments 5, 7 and 8 where pre-logging data were available

Species loss and addition	1C	2C	3C	1. Select	2. Heavy	3. Group	4. Clearcut	5. Preexp	6. Enrich	7. Improve	8. Postexp
not present 65 to present 99	3	5	10	11	6	19	6	8	14	13	11
present to not present	2	4	4	4	6	7	2	2	14	5	6





