IV

Rainforest Biodiversity Associated with Tropical and Subtropical Reforestation




11. Rainforest timber plantations and the restoration of plant biodiversity in tropical and subtropical Australia

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Abstract

We compared the species richness, growth forms and assemblages of vascular plants in five types of rainforest reforestation with pasture and forest reference sites in tropical and subtropical Australia. These types include unmanaged regrowth, young and old monoculture plantations, young rainforest cabinet timber species plantations and plantings designed to restore natural rainforest communities. Patterns of species richness across these reforestation types differed between the tropics and subtropics, although all reforestation types supported fewer species than natural rainforest reference sites. In the tropics similar numbers of introduced (i.e. non-native) species occurred in all types of reforestation (with the exception of old plantations which included few introduced species) and pasture reference sites. This contrasts with the subtropics where the greatest numbers of introduced species were associated with cabinet timber plantings. Greater diversity of growth forms (including epiphytes and vines) occurred in rainforest reference sites than in any type of reforestation. The assemblages of canopy trees (including both planted species and recruits) varied in their resemblance to rainforest reference sites in the different types of reforestation in the two regions. However, there was a tendency for young plantations to be most dissimilar to rainforest reference sites. On the other hand, old (ca. 60 years) plantation sites in the tropics were similar to natural rainforest reference sites. This was due to their close proximity to remnants and low intensity management regimes.

Because species richness and growth form obscures the importance of particular species in reforestation, we targeted eight common species (four native and four introduced) as exemplars of the possible biodiversity future under the different types of reforestation. These species demonstrated the individuality of species behaviour under different types of reforestation. Rainforest timber plantations can lead to increased biodiversity if they are designed to facilitate the colonization of rainforest taxa, and managed to favour processes associated with the development of a rainforest environment. Negative outcomes for rainforest biodiversity follow the establishment of non-rainforest species or processes (e.g. persistent high understorey light levels) not associated with a rainforest environment. Management and designs to minimize the need for ongoing intervention will be important economic considerations in future reforestation efforts aimed at restoring biodiversity.

Introduction

Australian rainforests are notable for their high biodiversity and their distribution over a large latitudinal range (Adam 1994, Barlow 1994, Webb and Tracey 1994). Northern Australian rainforests are noted at a world scale for their diversity of vascular plants, especially trees (Hyland *et al*. 2003). The extent of the rainforest cover in Australia, already diminished to refugial status by long-term climatic trends, was greatly reduced by European settlement (Floyd 1990, Winter *et al*. 1987, McDonald *et al*. 1998, Erskine 2002, Catterall *et al*. 2004).

Remaining areas of rainforest are now well represented in the reserve system, while in recent years there has been increasing interest in reforestation within former rainforest landscapes. This interest has been generated by the twin beliefs that the reserve system is insufficient in itself to conserve biodiversity (Adam 1994, Goosem and Tucker 1995, Turner 1996), and that reforestation can reverse some of the environmental damage caused by clearing (Lugo 1997, Parrota *et al*. 1997, Lamb 1998, Hartley 2002).

Approaches to reforestation in former rainforest landscapes have varied with time (Kanowski *et al*. 2003). The earliest reforestation programs resulted in extensive areas of monocultures of fast-growing native timber trees (Fisher 1980). Most of these plantations were of hoop pine (*Araucaria cunninghamii*). In more recent times, mixed-species plantations using rainforest species which are well-known from native forests for their production of potentially high value, appearance-grade cabinet timbers, and diverse mixtures of trees and shrubs to restore rainforest to cleared land have also been established in humid regions of north and south east Queensland as well as New South Wales (NSW) (Kooyman 1991, Lamb *et al*. 1997, Parrotta *et al*. 1997, Tucker 2000, Lamb and Keenan 2001, Catterall *et al.* 2004).

These have typically been on a relatively small scale (Catterall *et al*. 2004). In addition, extensive areas of cleared land have been allowed to revert to secondary forest, particularly following the decline of the dairy industry in some regions (Kanowski *et al.* 2003). These approaches to reforestation vary widely in their primary objective, costs, potential economic returns and presumably in their use by rainforest fauna (Lugo 1997, Lamb 1998, Harrison *et al*. 2000). Until recently there has been no comprehensive attempt to assess the capacity of different types of reforestation to restore biodiversity and ecological functioning to cleared rainforest lands. However, recent research has provided a framework to allow such an assessment (Kanowski *et al*. 2003, Catterall *et al*. 2004).

In this chapter, we examine the restoration of components of plant biodiversity in a variety of different types of reforestation in former rainforest landscapes in eastern Australia. We compare species richness, growth form and assemblage patterns of plants in different types of reforestation to determine trends in the restoration of plant biodiversity. We also examine the behaviour of particular species (native and introduced) following reforestation. We evaluate the extent to which these different reforestation styles and management systems have produced differing plant biodiversity outcomes.

Methods

Rainforest reforestation plantings

We examined plant biodiversity in the range of sites where reforestation approaches have been used. These approaches include: unmanaged regrowth, monoculture timber plantations, mixed-species cabinet timber plantations and plantings established to restore natural rainforest communities (referred to subsequently as ecological restoration plantings). The methodology used to investigate biodiversity values of reforestation has been described by Wardell-Johnson *et al.* (2002), Kanowski *et al.* (2003) and Catterall *et al.* (2004).

Reference sites of pasture and natural rainforest were used to benchmark the extremes of the rainforest clearing-restoration spectrum, and thereby to provide a context for restoration plantings. All are referred to collectively as land cover types and are described in Kanowski *et al*. (2003). Some attributes of these different approaches and sites are shown in Table 1.

Table 1 Summary of main attributes of the reforestation types and site conditions.

Type of Reforestation or Site	Age (years)	Tree density at planting (trees per ha)	Dominant tree height (m)	Other notes
Pasture reference (P)	80–120	NA	NA	Largely treeless reference sites
Regrowth (RG)	10-20 in tropics; 30-40 in subtropics	NA	< 10	Large variation in composition
Young monoculture plantation (YP)	5-15	1200	< 8	Entirely *Araucaria cunninghamii*
Old monoculture plantation (OP)	38-70	1200	< 35	Mostly *Araucaria cunninghamii* or *Flindersia brayleyana*; many acquire a high understorey diversity if near intact forest
Cabinet timber (Mixed species) plantation (CT)	5-10 years	> 1200	< 8	Most planted with 6-20 tree species per site
Restoration planting (ER)	6-22	4-6000	< 10	Most planted with 20-100 tree and shrub species per site
Rainforest reference (F)	Unknown	NA	30-40	c. 100-150 species per ha.

NA =not applicable

The study was conducted in the extensively modified agricultural areas of two rainforest regions of eastern Australia: the Atherton Tablelands, an upland plateau in tropical north-east Queensland; and the lowland subtropics of south-east Queensland and north-east New South Wales. The two study areas experience a similar climate (due to the lower altitude of the subtropical sites), and rainforests in both areas exhibit structural and floristic affinities (Webb 1968).

The two regions differ with respect to their management history (southern areas cleared for longer), the proportions of the landscape in different land cover types (more plantation in the south) and the extent of rainforest clearance (more cleared in the subtropics (Kanowski *et al.* 2003). Details of the study design and most land cover types are presented in a companion paper (Kanowski *et al.* Chapter 12). We also surveyed unmanaged regrowth that had developed on abandoned farmland. In the subtropics this regrowth was 30-40 years old and dominated by woody weeds, notably camphor laurel, *Cinnamomum camphora*, and broad-leaved privet, *Ligustrum lucidum.* In the tropics, patches of regrowth 10-20 years old comprised dense clumps of vines and scramblers, including the weed species *Lantana camara*, growing amongst pasture with a few rainforest trees and shrubs.

In the tropics, cabinet timber plantings were represented from Community Rainforest Reforestation Program (CRRP) plantations. Cabinet timber plantings in the subtropics were established by a number of different individuals and organisations, but those surveyed in this study resembled the CRRP plantations in terms of plant selection, spacing and management focus (see Harrison *et al*. 2003, Glencross and Nichols Chapter 7).

Sampling procedures

At each site, we conducted surveys of a wide range of ecological attributes, including plants, lizards, birds, soil and litter invertebrates, and forest structure (see Wardell-Johnson *et al*. 2002, Kanowski *et al*. 2003, Catterall *et al*. 2004). Surveys were conducted between 2000 and 2002 on a standardised 0.3 ha transect at each site.

Vascular plants were surveyed on five 78.5 m^2 quadrats at each transect, with species recorded as present or absent in each quadrat in each of three strata: canopy (top 1/3 of the canopy height), midstorey (2 m to 2/3 height of canopy) and understorey, or ground (< 2 m high). Only plants rooted in the quadrat, or growing on plants rooted in the quadrat, were counted. A frequency index (0-5, the number of quadrats in which a species was present) was used to describe the occurrence of each species in each stratum in each site. We also developed a database of ecological attributes (such as seed size, dispersal, growth form and origin) of each species.

Analytical approach

We compared the various land cover types of the two regions in terms of species richness, growth forms and assemblages. Species richness is presented separately for introduced (i.e. non-native) and native taxa, recorded at each of the three strata considered at each site. In this case, data were the total list of each site, presented as a mean and standard error for each land cover type. These land cover types are considered a proxy for successional stages of rainforest restoration from pasture through various planted forests, to reference forest.

Growth form was separated into five categories - canopy tree, shrub, vine, epiphyte and ground story (includes both herbs and low shrubs). In this case, data were the total list of species occurring in at least one site in at least one land cover type. Associations between growth form and land cover types were compared with χ^2 tests of independence.

We considered the assemblage pattern of canopy trees, where the data used were the sum of frequencies across all three strata for all species of canopy tree (including tree seedlings). Some species of shrubs, vines and epiphytes were sometimes detected in the canopy (particularly in regrowth and young reforestation sites). However, in this case, only species within the life form 'tree' and capable of occurring within the top third of the canopy of mature rainforest were included. A maximum score of 15 can be obtained by a canopy-growing tree occurring in the canopy, midstorey and ground layer of all five subplots. Pasture sites contained few canopy trees, and therefore were not included in this analysis. Multidimensional Scaling (MDS) ordination (Shepard 1962, Belbin 1991) was used to summarise and present patterns of similarity between sites in terms of plant-species composition. Kanowski *et al*. (2003) and Catterall *et al.* (2004) outline detailed analytical procedures associated with the use of ordination in this study.

As species can vary in their influence on a site, on the occurrence of other species, or on ecological processes, we consider individually four native species (*Flindersia brayleyana*, *Elaeocarpus grandis*, *Castanospermum australe* and *Argyrodendron trifoliatum)* and four introduced species (*Cinnamomum camphora*, *Panicum maxima*, *Ligustrum sinense* and *Lantana camara*) in relation to their frequency of occurrence in different land cover types.

The mean and standard error of the frequency of each species in each land cover type were plotted and compared by one-way ANOVA, with post-hoc LSD tests to determine significant pairwise differences between land cover types.

Results

Species richness and stratum

When considered over all strata, plant species richness per site was greatest for the natural rainforest reference sites (Figure 1). In the tropics and subtropics species richness was greater in ecological restoration, old plantation and forest reference sites than other land cover types, but the differences were less pronounced in the subtropics than in the tropics.

Overall, there was a trend for increasing species richness of native plants from pasture through young plantations, ecological restoration plantings and old plantations towards rainforest reference sites. Few introduced species were encountered in forest reference sites in either the tropics or subtropics or in old plantations of the tropics. However, there were many introduced species in all other land cover types in both regions. Although similar numbers of native and introduced species occurred in young plantations in the tropics, pasture was the only land cover type where introduced species generally outnumbered native species.

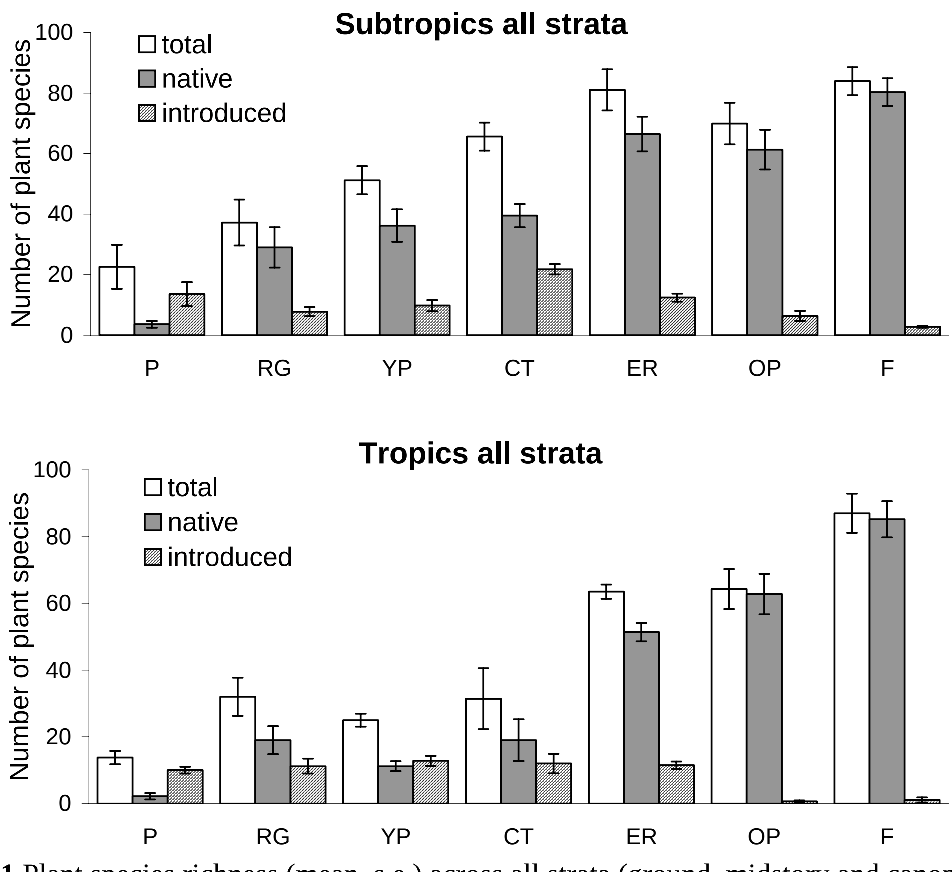

Figure 1 Plant species richness (mean, s.e.) across all strata (ground, midstory and canopy) of all introduced, native and total species in different styles of reforestation. Abbreviations are: Regrowth - RG; Young plantation – YP; Cabinet timber – CT; Ecological restoration – ER; Old plantation – OP; rainforest (F); and pasture (P). Totals include a few (< 1 %) taxa whose origin is uncertain.

When the canopy alone was considered, natural rainforest reference sites were more species rich than any other land cover type, and included fewer introduced species, in both the tropics and subtropics (Figure 2). On average, the canopy of ecological restoration plantings included more species in the tropics than the subtropics. Conversely, the canopy of cabinet timber plantings was more species rich in the subtropics, being similar to ecological restoration plantings in the same region. There was a paucity of species occurring in the canopy of old monoculture plantations, particularly in the subtropics. Relatively few introduced species were associated with the canopy of most land cover types (exceptions were regrowth in the subtropics, and young plantations and cabinet timber plantings in the tropics). Unmanaged regrowth included more canopy-occurring species in the subtropics than in the tropics.

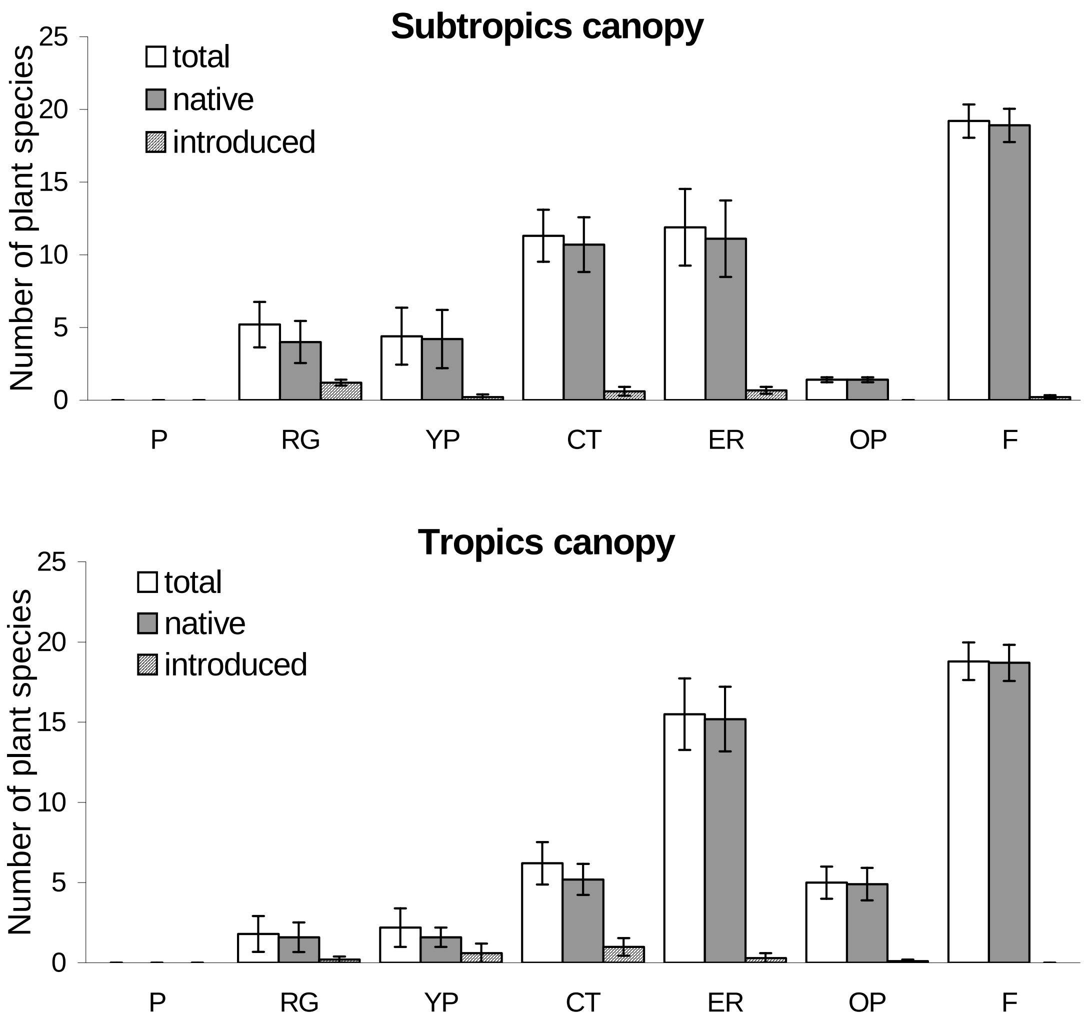

Figure 2 Plant species richness (mean, s.e. of all growth forms combined) in the canopy, of introduced, native and total species in different styles of reforestation Abbreviations are: Regrowth - RG; Young plantation – YP; Cabinet timber – CT; Ecological restoration – ER; Old plantation – OP; rainforest (F); and pasture (P). Totals include a few (< 1 %) taxa whose origin is uncertain.

Origin and growth form

When species were pooled across all sites in each land cover type, the pattern of species richness across different land cover and plantation types followed a similar trend to that observed on a site by site basis (Table 2).

Table 2 Summary of number of sites surveyed, and total numbers of plant species in each of seven different land cover types.

	Subtropics							Tropics						
	P	RG	YP	CT	ER	OP	F	P	RG	YP	CT	ER	OP	F
Number of sites	5	5	5	10	9	10	10	5	5	5	5	10	10	10
No. of native species	12	175	134	165	248	237	282	6	60	34	61	188	230	282
No. of introduced species	40	21	32	70	37	35	19	22	32	29	40	34	4	10

Characteristics of land cover types are shown in Table 1.
Abbreviations are: Regrowth - RG; Young plantation – YP; Cabinet timber – CT; Ecological restoration – ER; Old plantation – OP; rainforest (F); and pasture (P). Totals include a few (< 1 %) taxa whose origin is uncertain.

The two regions showed similar patterns of species; more native than introduced species, and the greatest numbers of native species in forest reference sites. However, because this is complicated by different sampling effort between land cover types or regions, we compared proportions between types and regions. In general, there were proportionally similar numbers of species in different growth form categories within similar land cover types between the two regions (Figure 3).

There was a significant association between growth form and land cover type for native species in both the subtropics (Figure 3, $\chi^2 = 171.5$, $p < 0.001$, d.f. = 24,), and tropics ($\chi^2 = 144$, $p < 0.001$, d.f. = 24). Thus, there were proportionally more epiphytes in regrowth and forest reference sites in the subtropics, and less in the young plantations and cabinet timber plantings in both the tropics and subtropics.

For all land cover types, except regrowth, there were more introduced (i.e. weed) species in the subtropics than the tropics (Table 2). This is especially true for young cabinet timber plantings, although this comparison is complicated by greater sampling effort in the subtropics than the tropics. There was a significant association between growth form and land cover type for introduced species in the subtropics ($\chi^2 = 34.93$, $p < 0.01$, d.f. = 18), but not in the tropics ($\chi^2 = 21.49$, $p < 0.256$, d.f. = 18).

Assemblage of canopy trees

A total of 258 species of canopy trees were detected in the 104 sites surveyed, including 147 species in the subtropics and 163 in the tropics (52 species were in common between the areas). This comprised approximately one quarter of all species recorded in the study (1088). Ordination analysis (Figure 4) showed that of all five reforestation styles, only old plantations and young plantations in the subtropics showed no trend towards difference from one another. The three reforestation approaches; cabinet timber, young monoculture plantation and unmanaged regrowth all trended

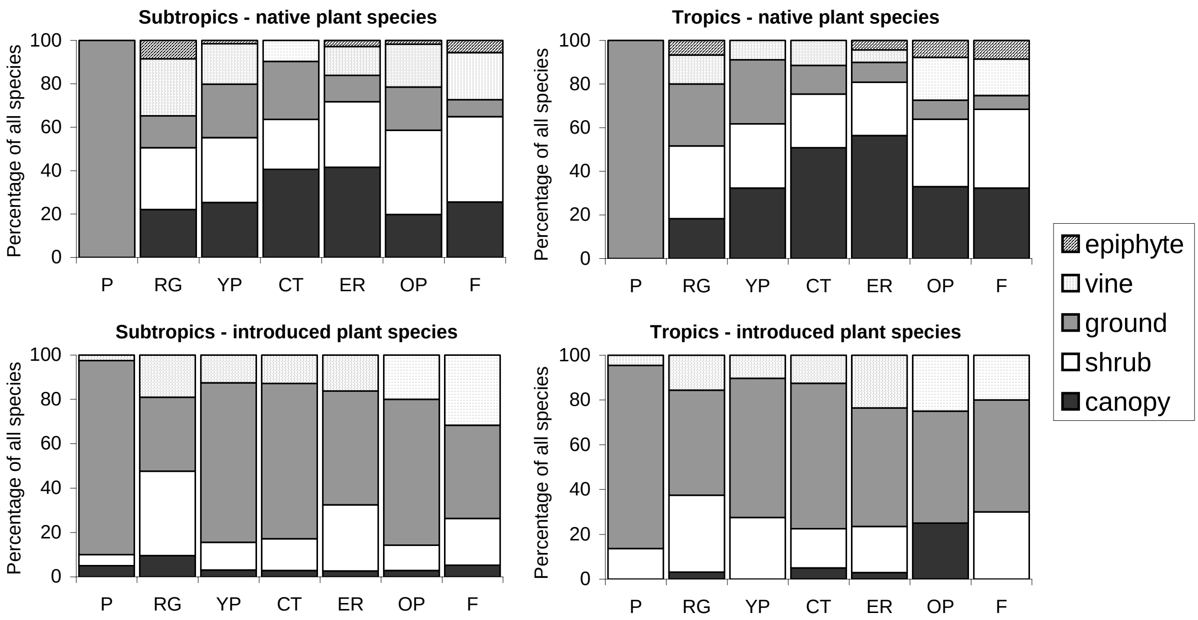

Figure 3 Proportions of species of different growth form in different land cover types Abbreviations are: Regrowth - RG; Young plantation – YP; Cabinet timber – CT; Ecological restoration – ER; Old plantation – OP; rainforest (F); and pasture (P). There are a few (< 1 %) taxa of unknown growth form that were not included in analysis.

differently in MDS ordination (Figure 4) although few canopy tree species were present in many of the tropical regrowth or young plantation sites.

The assemblage of canopy trees in the rainforest reference sites were well separated in ordination space from all other land cover types in both the subtropics and tropics (Figure 4). Old plantations in the tropics provide an exception. Although the canopy of old plantations was dominated by a single species (usually *Araucaria cunninghamii*) in both the tropics and the subtropics (Appendix 1), sites in the tropics were much less separated in ordination space from rainforest reference sites than sites in the subtropics. In the tropics, the assemblages of canopy species in both regrowth and young monoculture plantations were very different from rainforest reference sites. Cabinet timber sites varied widely in their assemblages of canopy trees in the tropics.

Although very different from rainforest reference sites, unmanaged regrowth in the subtropics was more similar in its canopy tree assemblage to rainforest than the unmanaged regrowth was in the tropics. Unmanaged regrowth sites were dominated by high frequencies of a few introduced trees and tall shrubs in the subtropics (particularly *Cinnamomum camphora*, *Ligustrum sinense* and *L. lucidum*) and by fewer individuals of the same species (and others such as *Psidium guajava*) in the tropics. However, the subtropics regrowth sites also included more native canopy tree species, particularly in the ground layer, than the younger tropical sites.

Overall species richness, proportions within growth forms, and assemblage patterns do not account for the influence of individual species within a community. Hence, we considered individually four native and four introduced exemplar species in terms of their frequency of occurrence in different land cover types (Appendices 1 and 2). Each of the native species considered in this paper (Appendix 1) are tall rainforest trees widely used in a variety of reforestation types in both the tropics and subtropics of Australia, while the four introduced species (Appendix 2) exemplify four different growth forms and two main dispersal mechanisms.

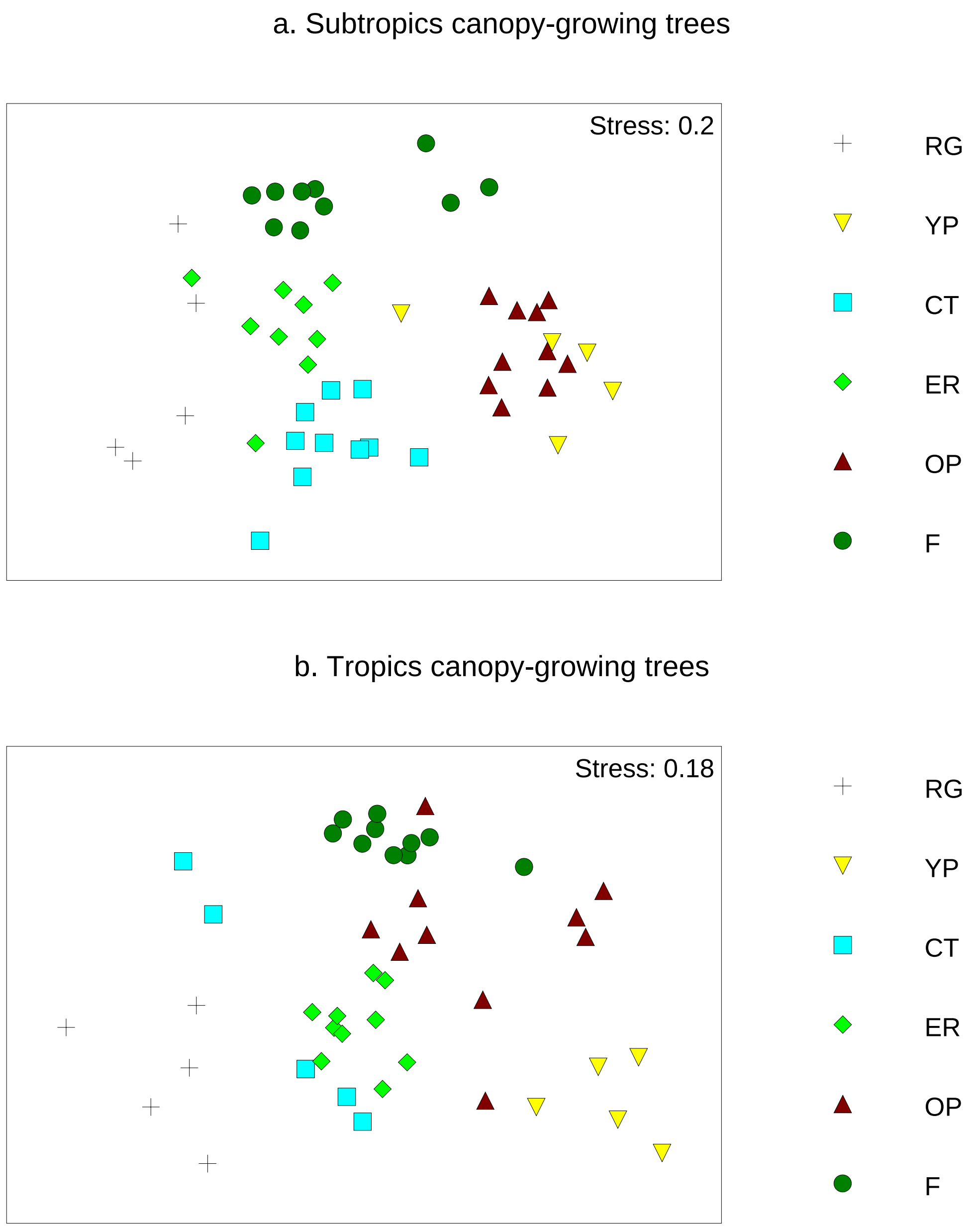

Figure 4 Ordination (SSH MDS, Bray Curtis metric, Stress = 0.2), showing trends based on frequency (0-15) of canopy tree species composition in all strata, in different styles of reforestation and rainforest reference sites (F) in the subtropics (a) and tropics (b). Reforestation styles are as follows: Regrowth - RG, Young plantation – YP, Cabinet timber – CT, Ecological restoration – ER and Old plantation – OP.

Discussion

Species richness and stratum

Differences in plant species richness between the different types of reforestation are not surprising since the various plantings differed in their silvicultural objectives; the older plantation monocultures were established and managed solely to provide timber, the restoration plantings were established largely to facilitate biodiversity recovery while objectives of the cabinet timber plantings included both timber production and biodiversity benefits.

While the plantings are still young, (even the oldest monoculture plantations are less than 70 years old) in comparison with natural rainforest reference sites, there are currently large differences in species richness between the different land cover types. Differences in plant species richness between the different forms of reforestation may have ecological consequences (e.g., in terms of their habitat for fauna: Kanowski *et al.* Chapter 12). These differences may be more associated with particular species or with forest structure. Thus, plantings differed in the composition of the species planted, e.g., ecological restoration plantings typically use many more fleshy-fruited species than timber plantations, which will influence their attractiveness to seed dispersing fauna.

The plantings are also very different from one another structurally (see Kanowski *et al*. 2003). These structural differences will influence the extent to which different types of reforestation facilitate the development of rainforest processes, such as ecological succession. For example, differences in canopy cover between timber plantations and restoration plantings, and the resultant differences in light levels, humidity and temperature may create an environment conducive to the recruitment of rainforest species.

Origin and growth form

Weed species are of particular interest in the establishment and maintenance of timber plantations since their control is both a major management expense and a conservation concern. Certain introduced species of particular growth form are widely regarded as environmental weeds that are capable of fundamentally altering the structure and function of ecosystems (e.g. Werren 2003, see also Groves and Burdon 1986, Bridgewater 1990, Michael 1994). For example, the relatively few species of woody weeds introduced to rainforest, particularly vine, tree and large shrubs, may have substantial capacity to transform this vegetation type.

There is a much greater proportion of tree species relative to species of grass, shrubs and herbs in the native assemblage than in the introduced assemblage in this study, and this is true also for the wet tropics and south-east Queensland bioregions in general (ANPWS 1991, Werren 2003). Invasive woody plants, particularly trees, are a threat to the biodiversity of the tropics in general (Binggeli *et al*. 1998). In the subtropics and tropics of eastern Australia, these species come from a taxonomically diverse group (Bebawi *et al*. 2002), but of the potentially serious invasive tree species listed by Werren (2003), only *Cinnamomum camphora* was detected in our sites. This species is widely regarded as being highly invasive in subtropical rainforests, but has received relatively little attention in the tropics, despite its establishment in the Atherton Tablelands.

Shrubs include taxa that are among the most invasive plants in subtropical rainforests (ANPWS 1991, Appendix 2). These include *Lantana camara*, *Ligustrum lucidum, L. sinense* and *Psidium guajava,* which are also highly invasive in tropical systems. *Lantana camara* is particularly associated with relatively high light levels within forests, while the remaining species are capable of colonizing intact rainforest. These shade-tolerant species have the potential to cause management concern even in reforestation that has facilitated rainforest processes (e.g., those with high levels of canopy cover).

Of the 19 introduced species that are considered particularly invasive of subtropical rainforest, in Australia by ANPWS (1991), 70 percent are vines. While none of these species were recorded in quadrats in this study, they are likely to increase considerably in abundance in the short term as the prognosis under the climate change scenarios currently considered (Hughes 2003) is for a greater spread of these transformer weeds.

Woody weeds, including vines, are considered of greatest significance in affecting biodiversity recovery in restoration of rainforest (ANPWS 1991). Other growth forms, in particular several grasses including *Panicum maximum*, are widely recognised as a threat to nature conservation values (Werren 2003). Grasses are primarily weeds of rainforest margins and of disturbance corridors, but also proliferate along riparian zones and other areas of natural disturbance. *Panicum maximum* is a relatively shade tolerant species and is particularly associated with edges and open canopies. It has the capacity to greatly increase fuel loads, thus rendering what are generally fire-sensitive communities more fire-prone (Werren 2001, 2003). This has particular importance in sites marginal for the growth of rainforest or potential rainforest sites for which open-forest processes have been facilitated.

The two study regions differ in composition and timing of establishment of introduced taxa, which in association with the management regime, will influence the trajectory of rainforest restoration. By comparing regions, this study has provided a prognosis for areas of different history. For example, distributions of the four environmental weeds that we targeted as exemplars (Appendix 2) suggest that these species may be able to thrive throughout the landscape within the two regions considered. It is therefore likely that the potential limits of the currently recognised environmental weeds will considerably expand. For example, the frequent occurrence of *Cinnamomum camphora* in pasture in the subtropics and its relatively recent occurrence in parts of the Atherton Tablelands suggest a dominant medium-term future for this species in both regions. We therefore suggest that the design and management of restoration programs will have a major influence on plant biodiversity outcomes by the way they promote or suppress, environmental weeds. These issues should be considered even while environmental weeds are not yet a major economic concern (as in the tropics in comparison with the subtropics).

Assemblage of canopy trees

We argue that the consideration of canopy tree species in all strata reflects the development of plant biodiversity in different rainforest types. This is because it includes recruitment of tree species which will in turn determine forest structure (see Tucker and Murphy 1997, McKenna 2001, Kanowski *et al*. 2003) and may influence the recruitment of other taxa (see Catterall *et al*. 2004). Patterns based on all canopy tree species also provide a context for biodiversity recovery in those reforestation styles which target only overstorey species (monoculture and cabinet timber plantations).

While 179 species of canopy-growing trees were included in CRRP cabinet timber plantations, these plantings mostly relied on a narrow genus pool, based largely on *Eucalyptus, Flindersia, Araucaria, Agathis* and *Elaeocarpus,* and the numbers of species in each planted area was relatively small (on average 11 species per site; see Lamb *et al*. Chapter 9). The relative contribution these young plantings currently make to plant biodiversity at any particular site is necessarily modest, and presently represents only a small increase over monoculture plantations in comparison to that found in intact forest. However, a possible long-term worth of all reforestation land cover types to biodiversity recovery may lie in the extent to which they can facilitate conditions and processes associated with a rainforest environment.

This study has demonstrated the considerable differences between rainforest reference sites and all reforestation land cover types examined in this study. Nevertheless, some types have a greater similarity of outcome to rainforest than others over the time frame examined. In particular, types which encourage rainforest processes or facilitate the colonisation of rainforest components tend to be most positive for biodiversity recovery. In this study, old plantations in the tropics were located adjacent to remnants of rainforest and most resembled rainforest reference sites (see also Keenan *et al*. 1997). Some of these sites received little stand management through thinning or pruning programs which tend to increase light levels, and weed control which removes understorey regeneration. Of the young reforested sites in the tropics and sub tropics, ecological restoration plantings most resembled the reference rainforest. Compared to other reforestation types, ecological restoration plantings were established using a diverse range of species, including many fleshy-fruited species attractive to seed dispersing fauna. Management of these sites included efforts to provide early canopy closure to minimise the need for longer-term weed management.

Characteristic timber plantation methods such as weed control in the establishment phase and allowing increased light in the lower canopy through pruning and thinning, may discourage successional rainforest processes, and thus may be less successful at promoting biodiversity in rainforest reforestation. Also, a reliance on a small number of predominantly wind-dispersed tree species, of limited value to seed-dispersing animals, limits the value of current timber plantation designs in biodiversity restoration (Tucker *et al*. 2004, Catterall *et al*. 2004, Kanowski *et al*. Chapter 12).

The role of plantations in enhancing plant biodiversity

The recovery of plant diversity on a local scale is enhanced by facilitating early canopy closure to exclude weeds (see also Florentine and Westbrooke 2004). The initial tree spacing of the cabinet timber plantings was typically around 3 - 4 m (equating to between 600 - 1100 stems/ha), which is less dense than ecological restoration plantings where trees are typically spaced 1.5 - 2 m apart (2500 – 4400 stems/ha). The choice of planting density involves trade-offs (see Catterall *et al*. Chapter 13). Wider spacing reduces the costs of establishment (including cost of supplying and planting the seedlings), but on the other hand, it increases the time taken for canopy closure, resulting in a greater maintenance effort to reduce competition from weeds. At the tree spacing of 3 - 4 m, weed control might be needed for up to three or four years before canopy closure occurs. This period is reduced when planting densities are greater (i.e. spacings are less). Without adequate weed control plantations can fail. For example, CRRP records indicate that, by 1998 when planting under that program was coming to an end, at least 15 % of plantations had ceased to have economic timber yield potential (Harrison *et al*. 2003). Several reasons were identified for this, but a lack of early weed control was the most important. Once canopy closure occurs most light-demanding weeds are excluded, although some can continue to thrive under more wide spaced plantations, or under open-crowned species such as eucalypts.

Successional development of reforestation then depends on seed dispersing fauna being able to reach the plantations (e.g. Wunderle 1997). Plantations isolated from a source of seeds such as a natural forest remnant are likely to acquire new plant colonists from outside more slowly than plantations close to natural forest remnants. For example, in north Queensland, Keenan *et al*. (1997) found sites within 200 m of intact forest could contain quite high numbers of species able to colonise the plantation and grow into the canopy, but that recruitment declined with distance from the forest edge. In the same region, McKenna (2001) found that the dispersal of native plants to rainforest plantings declined rapidly with distance from intact forest, with few rainforest plants dispersed to plantings isolated from native forest by more than 500 metres.

The composition of the plantation is also likely to influence the attractiveness of the plantation to seed dispersers in a number of ways. Firstly, some plantation tree species are more attractive to seed dispersers than others. For example, our data show a positive correlation between the number of fleshy-fruited, bird-dispersed plants used in plantations and the richness of frugivorous birds (the main dispersers in Australian rainforests) inhabiting or visiting those plantations (see Kanowski *et al.* Chapter 12). In particular, more seed-dispersing birds occur in cabinet timber plantations, which include some fleshy-fruited trees (e.g., *Elaeocarpus spp.*, *Gmelina spp.*), than wind-dispersed hoop pine plantations. Secondly, the composition of a plantation will influence its structural complexity. A structurally simple monoculture plantation may be less attractive to seed dispersers than a more structurally complex multi-species plantation. The young cabinet timber plantations surveyed in this study contained trees, but no understorey, and were not particularly structurally complex. On the other hand, the greater diversity of crown architectures and fruit resources in these plantations may be more attractive to many seed dispersing fauna than simple monocultures. Recruitment of rainforest plants and subsequent successional development is likely to be faster in the ecological restoration plantings, due to the closer tree spacing and the larger variety of species planted, especially the much greater use of fleshy-fruited species (Kanowski *et al.* Chapter 12).

In the early stages of plantation development non-planted species compete with the planted trees, and in timber plantations optimised for financial return, recruits are usually removed before they can hinder growth (Keenan *et al*. Chapter 10). After canopy closure the main competition is between the planted trees themselves; and silviculturalists seek to control such competition by periodically removing the least vigorous trees or those with poor form, a process known as thinning. Pre-commercial (or uncommercial) thinning, where there is no market for the small logs removed, is often necessary. The large expense of pre-commercial thinning was one of the reasons for the high diversity of species found in the older *Flindersia brayleyana* plantations in north Queensland; in this case no thinning was undertaken because no market could be found. In the absence of thinning, plantation tree growth slows and the volume of timber produced, and hence timber value per stem declines, but successional development continues, and biodiversity increases.

It is possible that, given time and under favourable conditions, monoculture plantations, ecological restoration plantings, cabinet timber plantations and unmanaged regrowth may begin to resemble each other, especially in a landscape where remnant patches of intact forest remain nearby. However, current silvicultural practices and clearfell harvest systems characteristic of timber plantations may limit long-term biodiversity outcomes (see Catterall *et al*. Chapter 13).

This raises an interesting dilemma for managers. How should these future cabinet timber plantations be managed? Should they be pruned to enhance commercial value and thinned to increase tree growth whenever markets for thinnings can be found? (Suggested thinning schedules are given in Keenan *et al*. Chapter 10). Or should thinning be excluded and successional development be encouraged to foster enhanced biodiversity? There is obviously no single answer to the dilemma because it will depend on the objectives of the forest grower, or landowner, and on the landscape context. Tree plantations take many years to grow and landowner objectives can change. A landowner with a cabinet timber plantation close to natural forest may decide that the biodiversity value of the plantation now exceeds any timber value and they will manage for enhanced biodiversity protection. A landowner with a plantation more distant from a source of a colonist might decide that the timber values exceed the biodiversity values and manage the forest, using pruning and thinning, to increase productivity. In this case current estimates of biodiversity might be modest and will be affected by plantation management and harvesting. Alternatively, such a landholder might decide to use timber trees which are particularly valuable to wildlife, or actively enrich the planting with native species (Tucker *et al.* 2004). Other management options might used, e.g. selective logging, or incorporating both the planted trees and any new colonists with commercial value (Lamb 1998).

Conclusions

The amount of plant biodiversity present in various types of reforestation depends on the numbers and characteristics of species initially planted, the management history and the age of the plantations or reforested site. The location of the reforested area within a landscape and the history of landscape-scale disturbance will also influence the extent to which successional development occurs. Rainforest timber plantings can lead to positive outcomes for regional biodiversity by facilitating processes that promote the colonization of rainforest taxa (all elements), provided that management regimes (e.g. plant spacing, careful species selection and early weed control) favour processes associated with the development of a rainforest environment. The rate at which this biodiversity develops will not be as rapid as in ecological restoration plantings, and its outcome will be truncated by harvesting. Tradeoffs between profitable timber production (which can be enhanced by thinning, pruning and removing understorey competition) and biodiversity (which may or may not be enhanced by the aforementioned management activities) need consideration.

Irrespective of the type of reforestation, the future is likely to see an increase in weed species in the tropics. It will therefore become increasingly important to minimise the requirement for on-going management of plantings. Achieving early canopy closure is the most effective means of insuring against weed incursion. It is likely to be cost effective to design reforestation programs with rainforest species around the facilitation of rainforest processes (e.g. rapid canopy closure). It is suggested that management and design to minimize the need for ongoing intervention will be important economic considerations in future reforestation efforts.

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Appendix 1

Native species

Flindersia brayleyana is a tall canopy tree restricted in natural distribution to north Queensland rainforest between Townsville and the Windsor Plateau. It is frequently planted in cabinet timber plantings in the tropics and subtropics, but is also used in ecological restoration and old plantations in both the tropics and subtropics (Figure 5). This species was not frequently encountered in forest reference sites in the tropics and does not occur naturally in the subtropics. Several other congeneric species are also used in ecological and cabinet timber plantings in both the subtropics and tropics.

Elaeocarpus grandis is a rapidly growing canopy rainforest tree occurring from northern NSW to northern Queensland (Boland *et al*. 1992). Although uncommon in forest reference areas (where it was not detected on any of our forest transects), this species is widely planted in both cabinet timber and ecological restoration sites. It was also encountered as a recruit in old plantations in the tropics and regrowth in the subtropics (Figure 5).

Castanospermum australe is a tall rainforest tree with a wide distribution in eastern Australia from the Bellinger River in NSW to Cape York, where it most typically occurs in gallery-type rainforests. This species was frequently encountered in rainforest reference sites in both the subtropics and tropics, and was also encountered in old plantations in the tropics (Figure 5). It was not found in old plantations of the subtropics, but was detected in unmanaged regrowth in the subtropics. It is planted in both ecological and cabinet timber sites in the tropics and subtropics.

Argyrodendron trifoliatum is a tall late successional tree in rainforests from northern NSW to northern Queensland, where it may dominate stands (Francis 1981). This species was very abundant in forest reference areas (particularly in the subtropics). It was also detected in old plantations adjacent to forest reference sites in the tropics, but not in the subtropics (Figure 5). It has been used in ecological plantings and in cabinet timber plantings in the subtropics. However, it was rarely encountered in other land cover types. Several other congeneric species also occur in both the tropics and subtropics. *Argyrodendron peralatum* tends to be favoured over *A. trifoliatum* in reforestation plantings in the tropics.

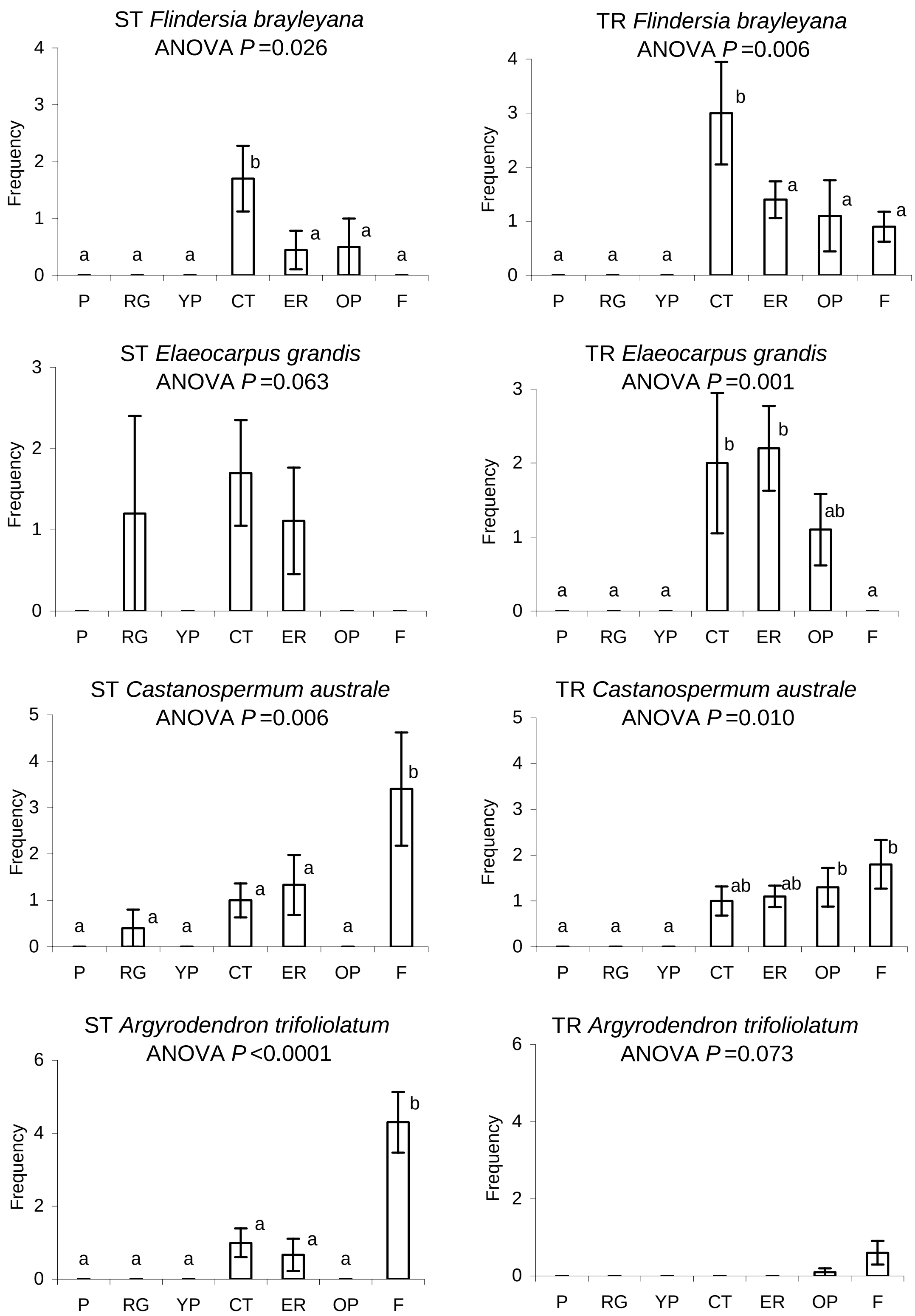


Figure 5 Mean frequency (see text for details) of four native tree species (*Flindersia brayleyana*, *Elaeocarpus grandis*, *Castanospermum australe* and *Argyrodendron trifoliatum*) in seven land cover types (Regrowth- RG, Young plantation – YP, Cabinet timber – CT, Ecological restoration – ER, Old plantation – OP; and rainforest - F and Pasture – P, reference sites) in subtropical and tropical Australia. Means with same letter are not significantly different (ANOVA, LSD).

Appendix 2

Introduced species

Cinnamomum camphora is one of few dominant canopy tree species that have become widely established as environmental weeds in the subtropics (others include *Ligustrum lucidum* and *Celtis sinensis*). This species is partly shade tolerant, is spread rapidly by birds and can persist in open pasture by resprouting following browsing. This species occurred at very high frequency in regrowth (where it was abundant in all strata) of the subtropics, and was also widespread in pasture in the same area (Figure 6). This species occurred in all but rainforest reference sites. By contrast, this species was of more limited extent in the tropics, but was encountered occasionally in unmanaged regrowth and ecological restoration. Some areas of the Atherton Tablelands include dense stands of this species, demonstrating its capacity to thrive in the region.

Panicum maximum is a tall (1.5-2 m), partially shade-tolerant clumping perennial grass native to southern Africa introduced as forage in the cattle industry. This species was not found in forest reference or old plantation sites in the tropics, but was frequently encountered in all other land cover types, particularly the relatively openly-spaced young plantations (Figure 6). It is less well established in the subtropics where it was commonly encountered only in cabinet timber sites. *Panicum maximum* was one of the few introduced species more frequently encountered in tropical than subtropical areas.

Ligustrum sinense is a tall, bird dispersed, shade tolerant shrub originating in southern China, and is one of few introduced environmental weed species to occur in forest reference sites (Figure 6); although at low frequency (others include *Psidium guajava* in the tropics). It was frequently encountered at dense regrowth in the subtropics and ecological plantings in the tropics, and also occurred at low frequency in relatively open sites (unmanaged regrowth, young plantations and cabinet timber plantings) in the tropics. However, this species was not detected in old plantations of the tropics or monoculture plantations of any age in the subtropics.

Lantana camara var. *camara* is a bird-dispersed tall sprawling shrub or vine from South America that can persist (but not thrive) in shade. This species is widespread in both the tropics and subtropics and was encountered in all land cover types except pasture (Figure 6). However, it was rare in forest reference sites in both the tropics and subtropics. It is frequently encountered in rainforest occurring in drier, more seasonal or less fertile sites in the same regions, and was frequently encountered in regrowth in the tropics but less so in the older regrowth of the subtropics. It was also frequently encountered in old plantations in the subtropics, but not in the tropics. It was frequently encountered in young plantations, cabinet timber and ecological restoration in both regions.

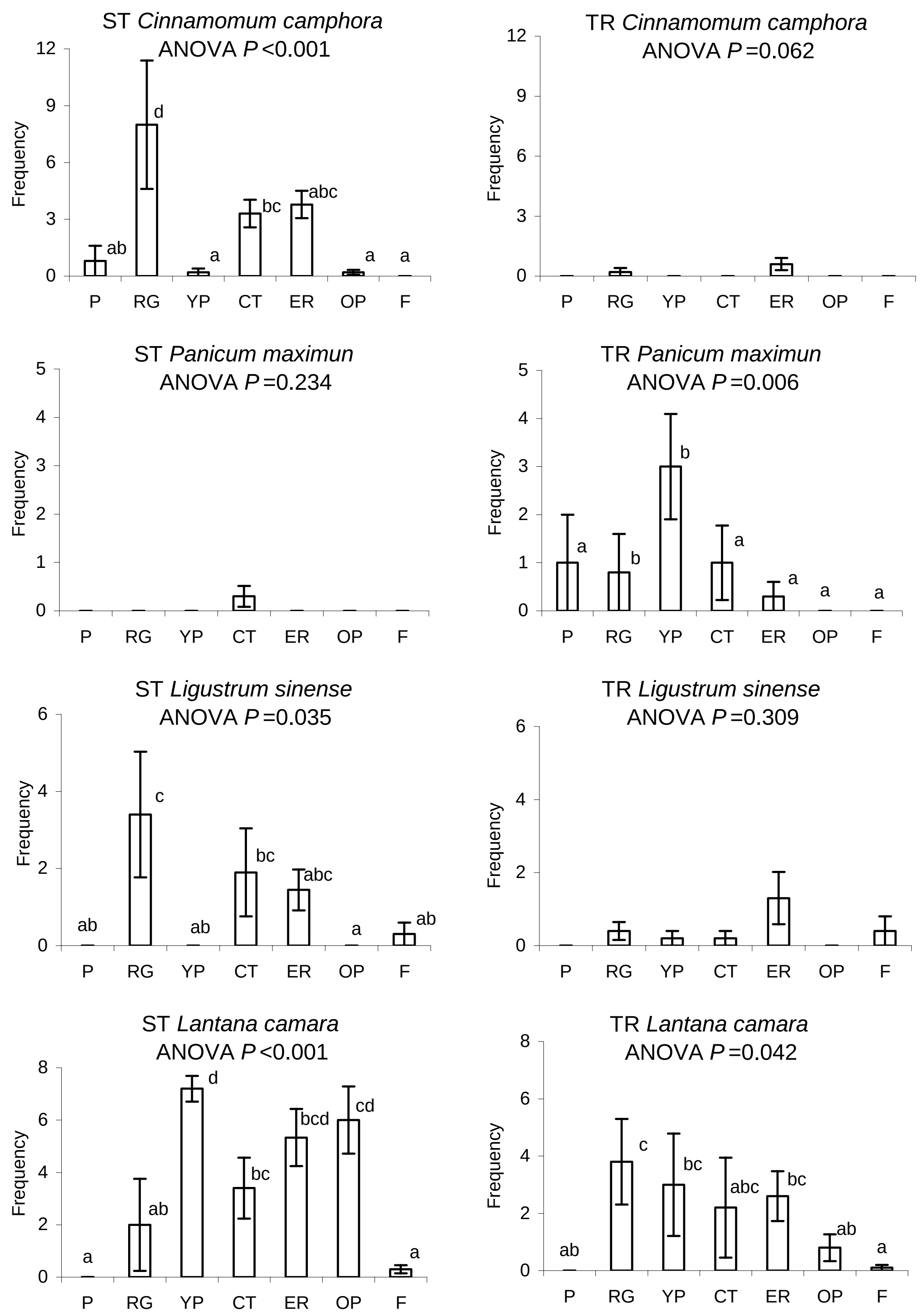


Figure 6 Mean frequency (see text for details) of four introduced plant species (Cinnamomum camphora, Panicum maxima, Ligustrum sinense and Lantana camara) in seven land cover types (Regrowth- RG, Young plantation – YP, Cabinet timber – CT, Ecological restoration – ER, Old plantation – OP; and rainforest - F and Pasture – P reference sites) in subtropical (ST) and tropical Australia (TR). Means with same letter are not significantly different (ANOVA, LSD).

12. Biodiversity values of timber plantations and restoration plantings for rainforest fauna in tropical and subtropical Australia

John Kanowski, Carla P. Catterall, Heather Proctor, Terry Reis, Nigel I.J. Tucker and Grant Wardell-Johnson

Abstract

It has been suggested that timber plantations could play an important role in the conservation of biodiversity in cleared rainforest landscapes, not only because of their potential to cost-effectively reforest large areas of land, but also because they may provide habitat for rainforest plants and animals. However, this last claim is largely untested. In this study, we surveyed the occurrence of a range of animal taxa in monoculture and mixed species timber plantations and restoration plantings in tropical and subtropical Australia. We used the richness of 'rainforest-dependent' taxa (i.e., birds, lizards and mites associated with rainforest habitats) in reforested sites as our measure of their 'biodiversity value'. We also examined whether the biodiversity value of reforested sites was correlated with habitat attributes, including plant species richness and vegetation structure and, further, whether biodiversity value was affected by the proximity of reforested sites to intact rainforest.

In general, our results showed that:

- *young timber plantations (both monoculture and mixed species) supported few rainforest taxa;*
- *Birds associated with rainforests were poorly represented in young timber plantations, but were moderately common in restoration plantings;*
- *Few rainforest lizards were recorded in young reforested sites, except in restoration plantings in the tropics;*
- *Rainforest mites were generally detected more frequently in restoration plantings than cabinet timber plantations, while the richness of rainforest mites in monoculture plantations varied between regions;*
- *The richness of rainforest birds in young reforested sites was positively correlated with plant species diversity and structural complexity, with similar correlations observed for rainforest lizards in the tropics;*
- *Rainforest mite richness was poorly correlated with measured habitat variables; and that*
- *Monoculture plantations close to intact forest tended to support more rainforest birds, lizards and mites than isolated plantations.*

These results suggest that plantations are likely to have limited value for rainforest taxa under conditions which often characterise broadscale reforestation: i.e., when plantations are established on cleared land, at some distance from intact forest and when plantations are managed intensively for timber production. Management of plantations for their faunal biodiversity values is likely to require the development of explicit design, management and harvest protocols, such as the incorporation of habitat features into plantations and/ or the reservation or restoration of native forest on part of the plantation estate.

Introduction

Rainforests cover less than 0.3% of Australia, but support around half its terrestrial biota (Adam 1994). In south-east Queensland and northern New South Wales (NSW), approximately half the area of rainforest present at the time of European settlement has been cleared for agriculture, plantation forestry and urban development (Floyd 1990; McDonald *et al.* 1998). In north Queensland, approximately one-quarter has been cleared (Winter *et al.* 1987; Erskine 2002). Forest types on arable land (e.g., floodplains and basalt plateaux) have been especially targeted for conversion to agriculture.

Following decades of community concern, the remaining areas of rainforest in Australia are now well represented in the reserve system (Adam 1994). However, the conservation of rainforests is likely to require more than the formal protection of remnants. Clearing and fragmentation have already wrought changes to the faunal composition of remnant forests (Date *et al*. 1991, Laurance 1994, Warburton 1997, Moran *et al.* 2004). Due to the complex interactions of plants and animals in rainforest dynamics, changes in the abundance of key animal species (e.g., seed dispersers, and seed and seedling herbivores) in remnant forests are likely to lead to the loss of biodiversity in rainforest remnants, even in protected areas, over the long term (Jones and Crome 1990, Lott and Duggin 1993; Turner 1996, Laurance and Bierregaard 1997, Gilmore 1999, Wright *et al.* 2002, Kanowski *et al.* 2004).

Revegetation is an important component of a strategy for rainforest conservation in Australia (Date and Recher 1991, Kooyman 1999, McDonald 1999, Catterall *et al.* Chapter 13). As yet, the extent of revegetation in cleared rainforest landscapes is small. In north Queensland, where government assistance for reforestation of former rainforest landscapes in Australia has been focussed, approximately 1,000 ha of diverse plantings established to restore natural rainforest communities (referred to subsequently as restoration plantings) and 2,000 ha of mixed species cabinet timber plantations have been established (Erskine 2002). The restoration plantings have largely been established to rehabilitate degraded remnants, enlarge the size of small remnants, or create habitat corridors between remnants (Joseph 1999, Tucker 2000). However, it may be necessary to revegetate a sizable proportion of the landscape to conserve rainforest biota over the long term (Catterall 2000, Catterall *et al.* Chapter 13). Although the scale of revegetation required is unknown, it is likely to be much larger than plantings currently established. For example, if all ‘endangered’ and ‘of concern’ rainforest types were to be restored to at least 30% of their presumed pre-European extent (the threshold below which forest types cannot now be cleared in Queensland), an additional 36,000 ha of cleared rainforest land would need revegetation in south-east Queensland alone (data compiled from Sattler and Williams 1999).

One reason why large areas of cleared land have not been returned to rainforest, even in areas where agricultural production has become marginally profitable, is the high cost of restoration. The rainforest restoration models typically practiced in eastern Australia involve the planting of a diverse range of trees and shrubs at high densities (Goosem and Tucker 1995, Kooyman 1996). These ‘ecological restoration’ plantings presently cost around $20,000 to $25,000 per ha, with little promise of a return from timber or other forest products (Erskine 2002; Catterall *et al.* Chapter 13). Without a large increase in government or private funding, or a substantial reduction in the cost of restoration (e.g., through the development of techniques such as direct seeding), the extent of restoration plantings is likely to remain small because of the costs involved in restoring rainforests.

Timber plantations have a greater potential to reforest much larger areas of cleared land. Not only are plantations much cheaper than restoration plantings (c. $5,000 to $10,000 per ha for mixed species plantations (Erskine 2002), less for monocultures), but they can also provide a return on investment when the timber is harvested.

For species with a proven production record (e.g. hoop pine - *Araucaria cunninghamii*), the expected financial return has been sufficient to encourage public investment in large-scale plantations and joint venture (government/ landholder) plantations on private land (Vize and Creighton 2001). Furthermore, plantations may recruit an understorey of rainforest plants and provide habitat for some rainforest animals (Parrotta *et al.* 1997 and references therein). For these reasons, it has been argued that timber plantations could play a major role in the restoration and conservation of biodiversity in cleared rainforest landscapes (Lugo 1997, Lamb 1998). Unfortunately, there are few data to test this claim.

Most studies of the potential value of timber plantations for rainforest biota have focussed on the recruitment of trees and shrubs to plantations (e.g., Keenan *et al.* 1997, Lamb *et al.* 1997, Parrotta *et al.* 1997). However, little is known about animal biodiversity in rainforest timber plantations (Bentley *et al.* 2000). Furthermore, most studies have been conducted in plantations established by conversion of intact forest, and/ or located adjacent to intact forest. Little is known about the potential biodiversity values of plantations established on cleared land at some distance from intact forest, conditions which would characterise the broad-scale reforestation of former rainforest landscapes.

In this paper, we present data from a research project investigating the biodiversity values of reforestation in former rainforest landscapes in eastern Australia. We survey the use of timber plantations (both monoculture and mixed species) and restoration plantings by rainforest-dependent birds, lizards and mites, and examine whether the occurrence of these taxa in plantings is correlated with aspects of plant species richness, vegetation structure and landscape context. Finally, we discuss the likely outcomes for faunal biodiversity in timber plantations under current management practices, and how these outcomes might be improved.

Methods

Study design

Our project surveyed a range of reforestation styles that are common in former rainforest landscapes of eastern Australia including monoculture timber plantations, mixed species cabinet timber plantations and ecological restoration plantings. Full details of the study design and methodology are provided elsewhere (Wardell-Johnson *et al.* 2002, Kanowski *et al.* 2003, Catterall *et al.* Chapter 13). Monoculture plantations were largely hoop pine, although in the tropics, we also surveyed some old monocultures of kauri pine (*Agathis robusta*), Queensland maple (*Flindersia brayleyana*) and red cedar (*Toona ciliata*). Monoculture plantations were established at relatively low densities (c. 1,200 stems per ha). Weeds were controlled by herbicides, slashing or grazing and trees were subject to thinning and pruning. Cabinet timber plantations typically comprised 6 – 20 species known from native forests for their potential to produce high-value appearance grade timber. Most were native rainforest species, although eucalypts (especially *Eucalyptus grandis* and *E. pellita*) and some exotics (e.g., *Cedrela odorata*) were often included. The plantations were established at similar densities and managed in a similar manner to monoculture plantations.

Restoration plantings mostly comprised a diverse mix of trees and shrubs (20–100 species, usually local species and provenances), planted at high densities (up to 6,000 stems per ha). In restoration plantings, weeds were controlled by hand or by herbicides, but trees were generally not thinned or pruned. All reforested sites were located on land which formerly supported rainforest (mostly complex notophyll vine forest in the terminology of Webb 1968). Old monoculture plantations were established by clearing and burning rainforest. Most young monoculture plantations were second rotation, except for two plantations established on already cleared land. All cabinet timber plantations and restoration plantings surveyed in the project had been established on cleared land or abandoned pasture.

We located reference sites in both pasture and intact rainforest. Pasture sites had been cleared of rainforest for 80–120 years, sown to exotic pasture grasses and subsequently grazed by dairy and beef cattle. Rainforest reference sites were selected to provide relatively undisturbed examples of complex notophyll vine forest and related forest types (Araucarian notophyll vine forest, complex mesophyll vine forest), representing the range of variation in the environments of reforested sites.

Research was conducted in tropical Australia (the Atherton Tablelands, north Queeensland) and in the subtropics (northern NSW and south-east Queensland). We obtained five to 10 replicate sites of each reforestation type and reference site type within each region to allow for variation in site history, management and landscape context. In selecting sites, we controlled for altitude and geology and major determinants of rainforest structure and composition (Webb 1968, Tracey 1982). The tropical sites were located at mid-elevations (500–850 m a.s.l.), mostly on basaltic soils, with rainfall between 1300 and 3000 mm per annum. The subtropical sites were located in the lowlands and foothills (10–400 m a.s.l.), on basaltic and metasedimentary soils, with rainfall between 1100 and 2000 mm per annum. The different site types were distributed across the rainfall gradient in each region, except in the subtropics, where monoculture plantations were mostly located in the drier parts of the study area. Replicate sites in each treatment were generally 1–10 km apart, except for monoculture plantations, where some sites were only a few hundred metres apart. However, closely adjacent sites in monoculture plantations differed in species planted or time of establishment. Most monoculture plantations were located amongst or adjacent to intact forest, whereas cabinet timber plantations and restoration plantings varied in their proximity to intact forest.

Almost all restoration plantings and cabinet timber plantations in our study areas were relatively young (one or two decades old, at most). Hence, a comparison of the biodiversity value of different types of reforestation was possible only for ‘young’ (5–22 years) plantings. Sites were constrained to be at least five years old, by which time denser plantings had usually attained canopy closure. To control for area effects on biota, we targeted sites which were greater than 4 ha, although a few sites as small as 2 ha were included to obtain sufficient replicates in some treatments. We also include data on the biodiversity values of ‘old’ (38–70 years) monoculture plantations. The resulting design is presented in Table 1.

Table 1 Attributes of rainforest plantings, pasture and intact rainforest sites surveyed in subtropical and tropical Australia.

Site type	Number of sites in: subtropics	tropics	Species planted	Age in years at survey: median (range)
Pasture reference sites	5	5	-	-
Monoculture plantations (young)	5	5	1	10 (5 – 15)
Cabinet timber plots	10	5	6 - 20	7 (5 – 10)
Restoration plantings	9	10	20 - 100	9 (6 – 22)
Monoculture plantations (old)	10	10	1	60 (38 – 70)
Rainforest reference sites	10	10	-	-

Sampling methodology

At each site, we conducted surveys of a range of taxa and ecological attributes (a full list of attributes surveyed is given in Catterall *et al.* Chapter 13). In this paper, we concentrate on results for birds, lizards and mites, and aspects of faunal habitat including floristic composition and vegetation structure. Surveys were conducted over a period of three years (between 2000 and 2002) on a standardised 100 m x 30 m (0.3 ha) plot at each site. Plots were located away from edges where possible.

Birds

Birds were assessed by recording all species seen or heard during six (subtropics) or eight (tropics) 30 minute surveys of the entire 0.3 ha plot. Only birds judged within the plot were used in analyses. Surveys were conducted at any time during daylight hours, except when hot or wet weather reduced activity levels. We were careful to rotate survey times across the different forest types. Surveys in the subtropics were conducted by a single observer, while tropical bird surveys were conducted by two observers, each of whom conducted two rounds of surveys of all sites. Two rounds of surveys were conducted every 3-4 months over the course of a year. No attempt was made to control for differences in detectability between sites (generally, visual detectability declined from pasture, through monoculture and cabinet timber plantations, restoration plantings and old plantations, to intact forest). However, most records were made from calls, which are less likely to be affected by differences in forest structure than sightings. Furthermore, the trend in visual detectability ran counter to trends in rainforest bird species richness (richness was highest in the more structurally complex plots), suggesting our results are conservative for rainforest birds.

Lizards

Lizards were surveyed by three 30 minute active searches of the entire 0.3 ha plot. If necessary, lizards were captured for identification using published keys (Cogger 2000). Surveys were carried out by a single observer (TR) on different days and over at least two different seasons.

Mites

Mites were extracted from two litres of leaf litter and surface soil collected at each plot, using a Tullgren funnel with a heat lamp operating for three days. The litter and soil was collected from a large number of ‘grabs’ from microsites (e.g., the forest floor, beside fallen logs, beside trees) located haphazardly across each plot. Mites were generally identified to family level (Walter and Proctor 2001). However, because of the poor state of taxonomy for the mite taxa Trombidioidea and Uropodoidea in Australia, these taxa were identified to superfamily rather than family. Likewise, most phoretic deutonymphal mites from the suborder Astigmata were simply identified as ‘hypopodes’ because of the difficulty of assigning them to families. Nevertheless, this level of taxonomic resolution provided about 70 taxa in each region.

Vascular Plants

Vascular plants were surveyed on five circular 78.5 m^2 quadrats, located systematically in each plot. Individuals rooted in the plot or, if epiphytes, growing on plants rooted in the plot were identified to species and recorded, if present, in each of three strata: canopy (top 1/3 of the canopy height), midstorey (2 m to 2/3 height of canopy) and ground (< 2 m high). The dispersal mode of plants was determined by reference to published sources (e.g., Tucker and Murphy 1997, Hyland *et al.* 2003) and unpublished data (D. Butler pers. comm., C. Moran pers. comm).

Species were categorised as bird-dispersed (mostly species with fleshy drupes, berries or arilate seeds), wind-dispersed (winged or plumed seeds) or dispersed by other modes. More comprehensive analyses of the plant data are presented elsewhere in this volume (Wardell-Johnson *et al.* Chapter 11).

Structural Attributes

Structural attributes were surveyed on five circular quadrats of 5-10 m radius, depending on the attributes measures, located systematically in each plot (for details, see Kanowski *et al.* 2003). Values for most of the structural attributes were strongly intercorrelated, hence for some analyses we reduced the dataset to an index of structural complexity. The index was calculated as the mean value of selected attributes at a site (see below), where each attribute was first standardised as a proportion of its average value in intact rainforest sites. The standardisation was conducted separately for tropical and subtropical sites. The structural attributes contributing to the index comprised canopy cover, basal area, canopy height, the abundance of woody stems (i) < 2.5 cm d.b.h., and (ii) > 2.5 cm d.b.h., the density of large trees (> 50 cm d.b.h.), the vertical diversity of tree heights, the abundance of special life forms (vines, epiphytes, hemi-epiphytes, strangler figs), leaf litter dry weight and an index of the volume of coarse woody debris.

Analytical approach

In this paper, we define biodiversity value as the richness of rainforest-dependent taxa recorded at a site, relative to a number of rainforest reference sites, using a standardised sampling protocol (see also Catterall *et al.* Chapter 13). This definition is based on the assumption that, from an ecological or conservation perspective, the elements of biodiversity that are of value in a former rainforest landscape are taxa potentially threatened by the clearing and fragmentation of rainforest, rather than taxa which may benefit from rainforest destruction (e.g. ‘grassland’ species). The definition is simple and readily applied to survey data, although it requires knowledge of the habitat preferences of target taxa. More sophisticated analyses of the biodiversity values of rainforest plantings (e.g., analyses which consider differences in the composition of entire assemblages) is presented in this volume (Catterall *et al.* Chapter 13).

For the purposes of this paper, we considered the ‘biodiversity value’ of rainforest plantings in terms of three faunal groups: birds, lizards and mites. For birds, habitat preferences were determined from published data (principally Kikkawa 1968, 1991, and Crome *et al*. 1994). We defined ‘rainforest’ birds as species largely associated with, or apparently dependent on, rainforest and associated wet sclerophyll forests (based on species occurrence in relatively extensive tracts of intact forest). ‘Other forest’ birds were species found regularly across a variety of forested habitats from rainforest to eucalypt woodlands, some being largely confined to eucalypt assemblages; while ‘grassland/ wetland’ birds were those species found mainly in grassland, pasture, swamps, or unforested streams, and sometimes in lightly timbered areas. A list of rainforest birds recorded in the study is provided in Table 2.

Table 2 Rainforest-dependent birds. recorded on surveys plots in rainforest plantings and intact rainforest sites.

Family	Species	Common Name
Megapodiidae	*Alectura lathami*	Australian brush turkey
	Megapodius reinwardt	orange-footed scrubfowl
Accipitridae	*Accipiter novaehollandiae*	grey goshawk
Columbidae	*Columba leucomela*	white-headed pigeon
	Macropygia amboinensis	brown cuckoo-dove
	Chalcophaps indica	emerald dove
	Ptilinopus magnificus	wompoo fruit-dove
	Ptilinopus superbus	superb fruit-dove
	Ptilinopus regina	rose-crowned fruit-dove
	Lopholaimus antarcticus	topknot pigeon
Psittacidae	*Cyclopsitta diophthalma*	double-eyed fig-parrot
Pittidae	*Pitta versicolor*	noisy pitta
Climacteridae	*Cormobates leucophaeus*	white-throated treecreeper
Pardalotidae	*Oreoscopus gutturalis*	fernwren
	Sericornis citreogularis	yellow-throated scrubwren
	Sericornis keri	Atherton scrubwren
	Sericornis magnirostris	large-billed scrubwren
	Gerygone mouki	brown gerygone
	Acanthiza katherina	mountain thornbill
Meliphagidae	*Xanthotis macleayana*	Macleay's honeyeater
	Lichenostomus frenatus	bridled honeyeater
Petroicidae	*Tregellasia capito*	pale-yellow robin
	Heteromyias albispecularis	grey-headed robin
Orthonychidae	*Orthonyx temminckii*	logrunner
	Orthonyx spaldingii	chowchilla
Cinclosomatidae	*Psophodes olivaceus*	eastern whipbird
Pachycephalidae	*Colluricincla megarhyncha*	little shrike-thrush
	Colluricincla boweri	Bower’s shrike-thrush
Dicruridae	*Machaerirhynchus flaviventer*	yellow-breasted boatbill
	Monarcha melanopsis	black-faced monarch
	Monarcha trivirgatus	spectacled monarch
	Arses kaupi	pied monarch
Campephagidae	*Coracina lineata*	barred cuckoo-shrike
Oriolidae	*Sphecotheres viridis**	figbird
Artamidae	*Cracticus quoyi*	black butcherbird
Paradisaeidae	*Ptiloris paradiseus*	paradise riflebird
	Ptiloris victoriae	Victoria’s riflebird
Ptilonorhynchidae	*Ailuroedus melanotis*	spotted catbird
	Ailuroedus crassirostris	green catbird
	Scenopoeetes dentirostris	tooth-billed bowerbird
	Sericulus chrysocephalus	regent bowerbird
Muscicapidae	*Zoothera heinei*	russet-tailed thrush
Sturnidae	*Aplonis metallica*	metallic starling

** considered a rainforest-dependent species in the tropics only*
Note: this is not a comprehensive list of rainforest-dependent birds occurring in subtropical and tropical Australia

Similarly, we defined 'rainforest' lizards as species largely confined to, or apparently dependent on, rainforest, according to published accounts (Covacevitch and McDonald 1991, Cogger 2000). A list of rainforest lizards recorded in the study is provided in Table 3.

Table 3 Rainforest-dependent lizards .recorded on surveys plots in rainforest plantings and intact rainforest sites.

Family	Species	Common Name
Agamidae	*Hypsilurus boydii*	Boyd's forest dragon
Scincidae	*Calyptotis lepidorostrum*	a fossorial skink
	Egernia major	land mullet
	Eulamprus murrayi	a forest skink
	Eulamprus tigrinus	a forest skink
	Gnypetoscincus queenslandiae	prickly forest skink
	Lampropholis coggeri	a sun skink
	Lampropholis couperi	a sun skink
	Lampropholis robertsi	a sun skink
	Ophioscincus ophioscincus	a snake skink
	Ophioscincus truncatus	a snake skink
	Saproscincus basiliscus	a shade skink
	Saproscincus challengeri	a shade skink
	Saproscincus spectabilis	a shade skink
	Saproscincus tetradactylus	a shade skink

Note: this is not a comprehensive list of rainforest-dependent lizards occurring in subtropical and tropical Australia.

For mites, where habitat associations were not known *a priori*, we calculated the proportion of rainforest and pasture sites in which each taxa occurred. We defined 'rainforest' mites as (i) taxa detected in both study regions in rainforest, but not in pasture; and (ii) taxa detected far more frequently in rainforest than pasture in a region. For the latter, a 60% difference in frequency of occurrence between rainforest and pasture was considered meaningful: e.g., 'rainforest' mites included taxa detected in at least six of the ten rainforest sites and no pasture sites in one region, or if detected in one of the five pasture sites in a region, then in at least eight of the ten rainforest sites, and so on. The converse rule was used to identify 'pasture' mites. Remaining taxa were included in the 'other' category. A list of taxa identified as 'rainforest' mites in the study is given in Table 4.

Differences in the mean richness of rainforest taxa between different types of plantings and reference sites were analysed with ANOVA with post-hoc LSD tests. Results are reported separately for the tropics and subtropics, because of regional differences in biota as well as, for birds, differences in survey effort and observers.

To examine potential determinants of biodiversity value, we examined correlations between the richness of rainforest biota (birds, lizards and mites) and selected habitat attributes including plant richness and various structural attributes. For these analyses, we only included data from young replanted sites, as the old plantations differed considerably from the young plantings in site history (no intervening pasture or plantation phase) and landscape context (almost all the old plantations were located adjacent to intact forest), which might confound any correlation with habitat attributes.

We also conducted a preliminary analysis of the influence of proximity to rainforest on the richness of rainforest biota in young revegetated sites. In this analysis, sites were classified as 'close' if within 400 m of extensive or remnant (> 5 ha) rainforest (most were adjacent to rainforest), or 'distant' if more than 400 m from extensive or remnant rainforest (most were more than 1 km from rainforest).

These thresholds are arbitrary, but previous work suggests that small rainforest remnants tend to support only a subset of the biota of intact forest (e.g., Warburton 1997). This last comparison is made tentatively, as the number of sites in most categories was small, precluding statistical analysis.

Table 4 Mite taxa (mostly identified to family level) categorised as indicators of pasture and rainforest in subtropical and tropical Australia.

Mite taxon	'Pasture' mites		'Rainforest' mites	
	Subtropics	Tropics	Subtropics	Tropics
Acaridae		*		
Caligonellidae	*	*		
Cunaxidae	*			
Digamasellidae	*			
Erythraeidae	*			
Ixodidae	*	*		
Parasitidae	*	*		
Rhodacaridae	*	*		
Tarsonemidae		*		
Tetranychidae	*	*		
Tydeidae	*	*		
Alicorhagiidae			*	*
Bimichaeliidae				*
'hypopodes'#				*
Labidostommatidae			*	*
Penthalodidae			*	*
Rhagidiidae			*	*
Smarididae			*	*
Trachytidae			*	
Trombidiodea			*	*
Uropodoidea				*

phoretic Astigmata
See text for explanation of categories.
Note: This is not a comprehensive list of rainforest-dependent mite taxa for each region.

Results

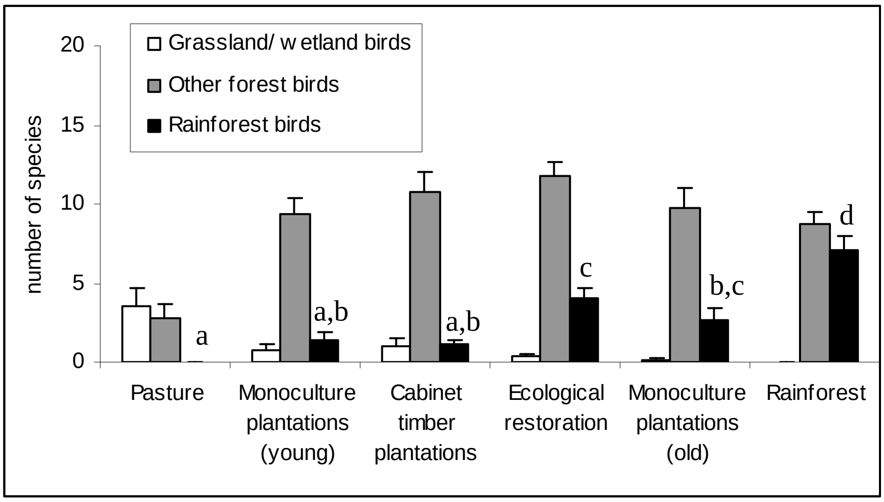
Birds, lizard and mite richness in timber plantations and rainforest plantings

Birds

Bird assemblages in young reforested sites were dominated by habitat generalists (Figure 1). Rainforest-dependent birds were relatively uncommon in young monoculture and cabinet timber plantations. In contrast, the richness of rainforest birds recorded in restoration plantings was about half that of intact rainforest. These patterns were similar in both study regions.

The richness of rainforest birds in old monoculture plantations varied between regions. In the subtropics, old plantations supported less than half the number of rainforest birds recorded in intact rainforest sites, on average. In the tropics, old plantations supported about 75%, on average, of the birds recorded in intact rainforest sites.

(a)



(b)

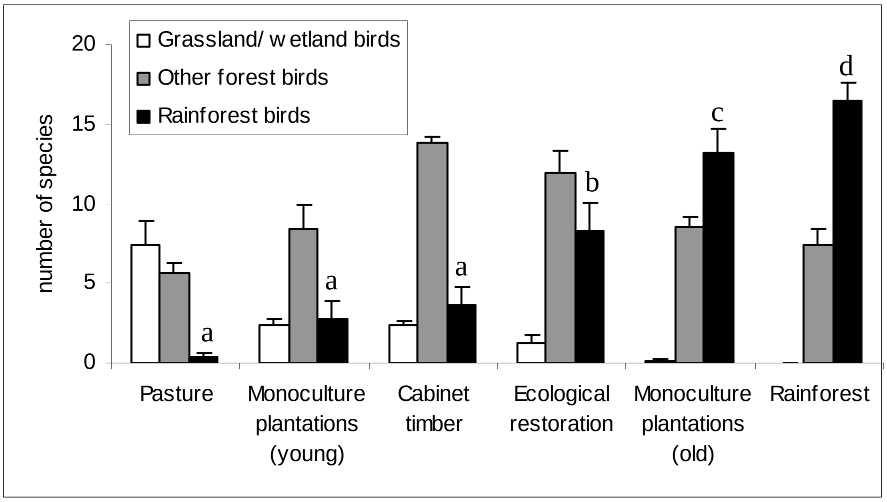
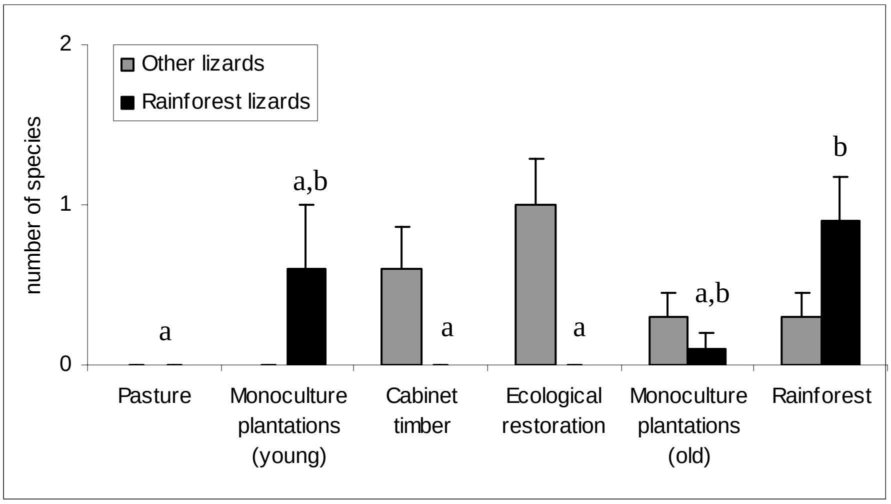


Figure 1 Bird species richness (mean, s.e.) in rainforest plantings, pasture and intact rainforest sites: a) subtropics; b) tropics. 'Rainforest' birds = species largely confined to, or apparently dependent on, rainforest; 'other forest' birds = species found regularly across a variety of forested habitats, some being largely confined to eucalypt assemblages; 'grassland/ wetland' birds = species found mainly in grassland, pasture, swamps, or unforested streams, and sometimes in lightly timbered areas. Treatments with the same letters are not significantly different in rainforest bird richness.

Lizards

Only a few rainforest lizards were recorded in this study, especially in the subtropics (Figure 2). Assemblages in young reforested sites tended to be dominated by lizards associated with open habitats, rather than rainforest. In the tropics, rainforest lizards were recorded in restoration plantings and old monoculture plantations, but not in young timber plantations. Some rainforest lizards were recorded in young monoculture plantations in the subtropics, often associated with relictual coarse woody debris.

(a)



(b)

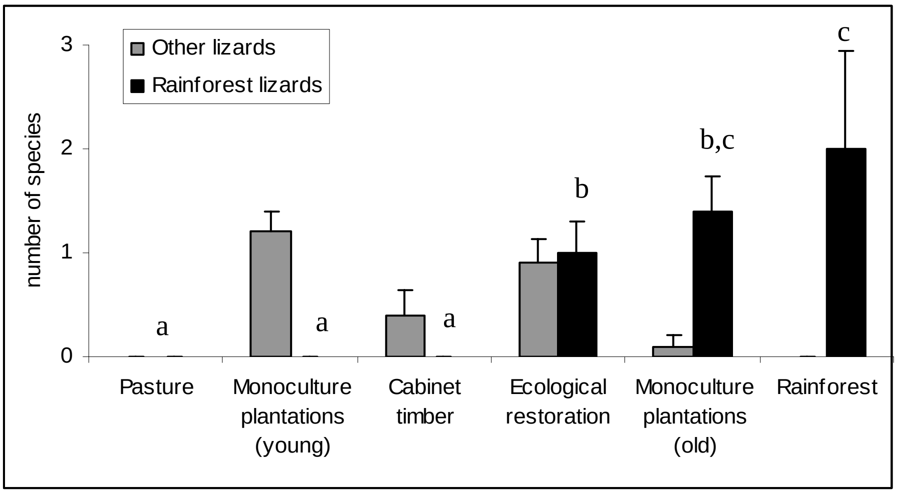


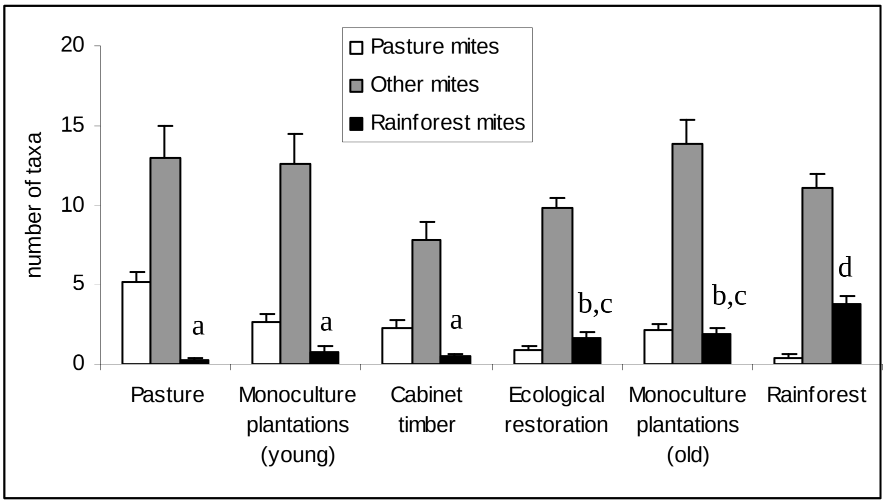
Figure 2 Lizard species richness (mean, s.e.) in rainforest plantings, pasture and intact rainforest sites: a) subtropics; b) tropics. 'Rainforest' lizards = species largely confined to, or apparently dependent on, rainforest; 'other lizards' = species found regularly across a variety of habitats. Treatments with the same letters are not significantly different in rainforest lizard richness.

Mites

Mite assemblages (mostly identified at the family level) in reforested and reference sites were dominated by taxa associated with a wide range of habitats. Nevertheless, we identified a number of mite taxa which were strongly associated with intact rainforest (Figure 3**)**. The richness of these 'rainforest' mites was generally higher in revegetated sites than pasture, but less than that recorded in rainforest. In young revegetated sites, rainforest mites tended to be least common in cabinet timber plantations and most common in restoration plantings. The relative richness of rainforest mites in young monoculture plantations varied between regions, with proportionally more rainforest taxa in plantations in the tropics than the subtropics.

The occurrence of rainforest mites in old monoculture plantations also varied between regions. In the subtropics, old plantations supported about half the rainforest mites of intact rainforest, whereas in the tropics, old plantations supported a similar number of rainforest mites as intact rainforest.

(a) Subtropics



(b) Tropics

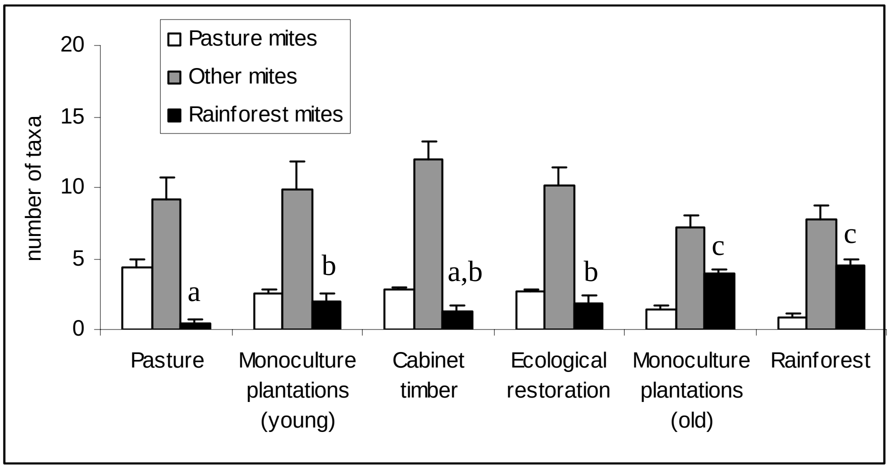


Figure 3 Mite taxon richness (mean, s.e.) in rainforest plantings, pasture and intact rainforest sites: a) subtropics; b) tropics. Mite families were categorised by habitat associations according to their frequency of occurrence in pasture and forest sites (see text). Treatments with the same letters are not significantly different in rainforest mite richness.

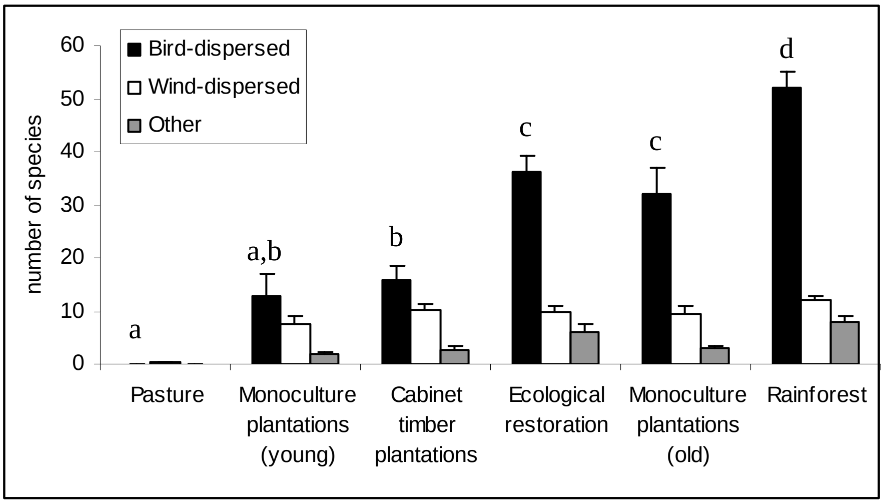
Habitat attributes of timber plantations and rainforest plantings

Plant species richness

For the purposes of this paper, we consider only plants in the canopy and midstorey (mostly shrubs, trees and vines), as most of the reproductively mature individuals which might provide resources for fauna (e.g., nectar, fruit) are in these strata. Within young revegetated sites, plant species richness in the canopy and midstorey increased from monoculture plantations, through cabinet timber plots to restoration plantings (Figure 4).

Most canopy and midstorey plants in restoration plantings were fleshy-fruited and dispersed by birds, similar to the pattern in intact rainforest. In contrast, bird-dispersed plants were relatively uncommon in the canopy and midstorey of young timber plantations, especially in the tropics. However, most old monoculture plantations supported a relatively rich flora of bird-dispersed plants.

a) Subtropics



b) Tropics

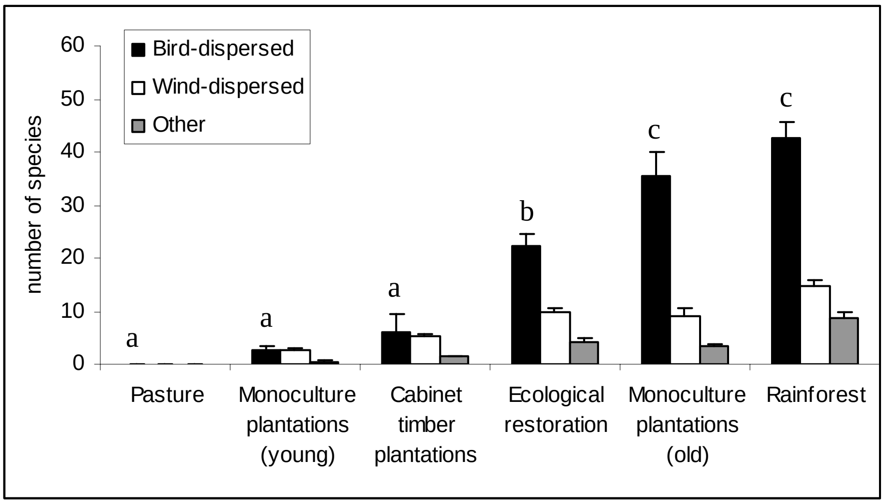


Figure 4 Plant species richness (mean, s.e.) in the canopy and midstorey of rainforest plantings, pasture and intact rainforest sites: a) subtropics; b) tropics. Plants were categorised by the dispersal mode of the seed (bird dispersed = fleshy fruited drupe, berry or arilate seed; wind dispersed = winged or plumed seed). Treatments with the same letters are not significantly different in the richness of bird-dispersed plants.

Forest structure

Young monoculture and cabinet timber plantations typically had a simple structure, with an open canopy and grassy ground cover (Figure 5). Restoration plantings had a more complex structure, with a relatively closed canopy and understorey of shrubs, seedlings, herbs and leaf litter. Nevertheless, young revegetated sites generally lacked a suite of structural attributes which are characteristic of intact rainforest, including robust vines, epiphytes, hemi-epiphytes, strangler figs, large trees and large woody debris. Many of these structural attributes were well-developed in old plantations in the tropics, but not in the subtropics.

(i) a)

(b)

(ii) a)

(b)

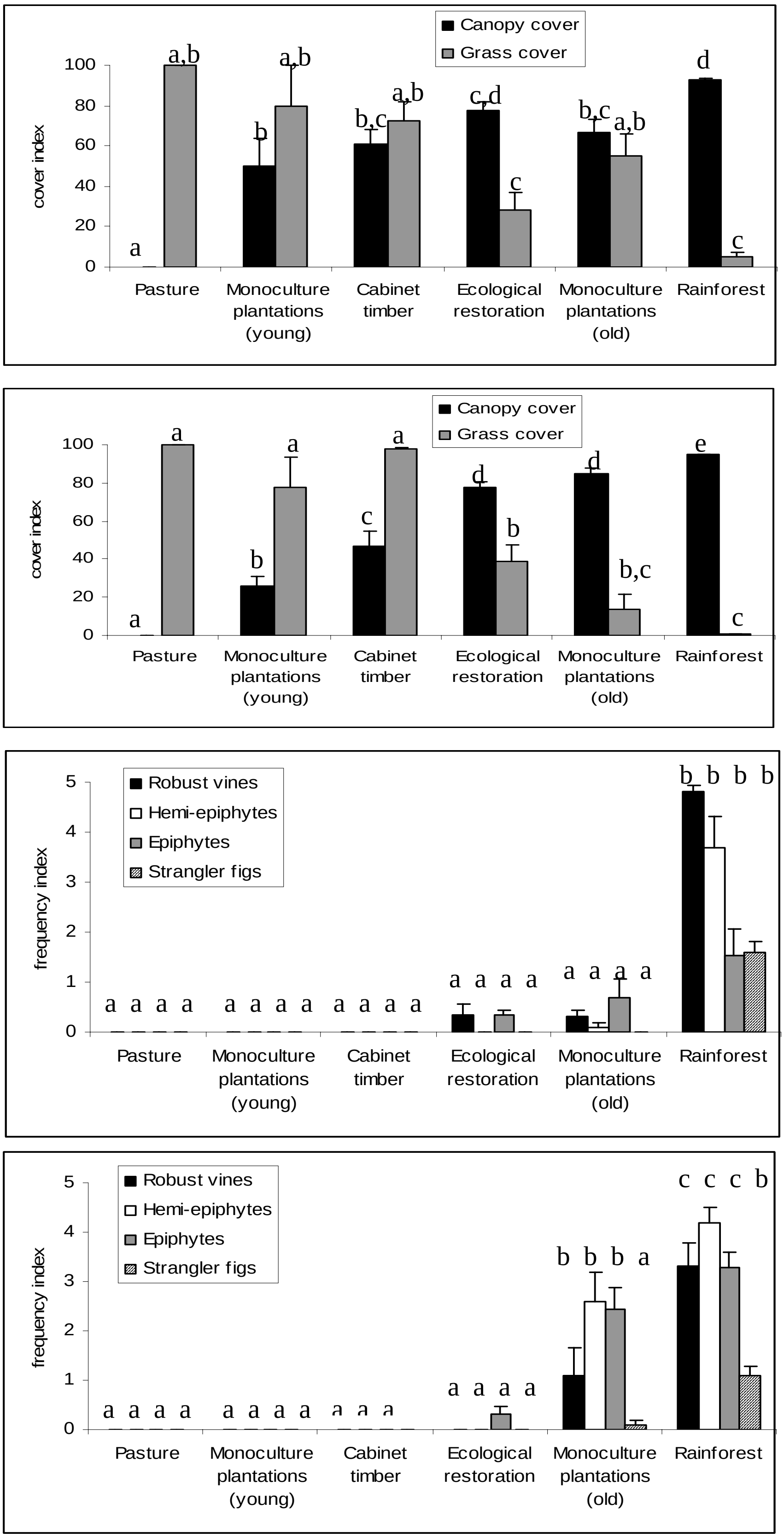


Figure 5 Selected aspects of the physical structure of the vegetation (mean, s.e.) of rainforest plantings, pasture and intact rainforest sites: a) subtropics; b) tropics. (i) Canopy cover and grass cover: a) subtropics; b) tropics. Canopy cover = projective cover of vegetation > 2 m above ground; grass cover = frequency of occurrence in twenty-five 0.5 m radius plots per site. (ii) Abundance of special life forms (Webb et al. 1976): a) subtropics; b) tropics.

Correlations between the richness of rainforest biota and habitat attributes in young rainforest plantings

The richness of rainforest birds in young revegetated sites was positively correlated with a number of habitat attributes, including plant species richness and various aspects of structural complexity (Table 5). The richness of rainforest lizards in these sites was also associated with plant species richness and some structural variables, but only in the tropics. Few rainforest lizards were recorded in the subtropics however, so the analysis had little power in that case. The richness of rainforest mites in young revegetated sites was not significantly correlated with any of the measured variables, except for a positive relationship with canopy cover in the subtropics.

Table 5 Rank correlation (r_s, P) between the richness of rainforest-dependent birds, lizards and mites, and selected habitat attributes, in young timber plantations and restoration plantings in subtropical (n = 24) and tropical (n = 20) Australia.

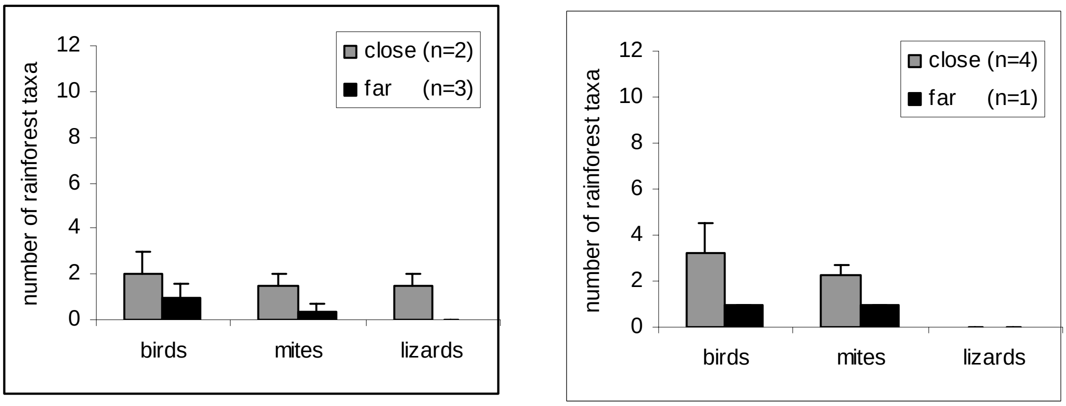
Habitat attribute	Rainforest bird richness		Rainforest lizard richness		Rainforest mite richness	
	Subtropics	Tropics	Subtropics	Tropics	Subtropics	Tropics
Plant richness in canopy and midstorey	***r* = 0.59, *P* = 0.002**	***r* = 0.62, *P* = 0.004**	*r* = -0.28, *P* = 0.18	***r* = 0.46, *P* = 0.043**	*r* = 0.15, *P* = 0.48	*r* = 0.18, *P* = 0.45
Canopy cover	*r* = 0.33, *P* = 0.11	***r* = 0.59, *P* = 0.007**	*r* = 0.21, *P* = 0.33	***r* = 0.70, *P* = 0.001**	***r* = 0.46, *P* = 0.023**	*r* = 0.04, *P* = 0.88
Abundance of woody stems < 2.5 cm d.b.h.	***r* = 0.66, *P* < 0.001**	***r* = 0.65, *P* = 0.002**	*r* = 0.19, *P* = 0.38	*r* = 0.39, *P* = 0.089	*r* = 0.12, *P* = 0.57	*r* = 0.18, *P* = 0.45
Abundance of woody stems > 2.5 cm d.b.h.	***r* = 0.59, *P* = 0.003**	***r* = 0.46, *P* = 0.044**	*r* = -0.28, *P* = 0.18	*r* = 0.43, *P* = 0.059	*r* = 0.35, *P* = 0.094	*r* = -0.01, *P* = 0.96
Index of structural complexity	***r* = 0.75, *P* < 0.001**	***r* = 0.59, *P* = 0.007**	*r* = -0.03, *P* = 0.90	***r* = 0.57, *P* = 0.009**	*r* = 0.30, *P* = 0.15	*r* = 0.01, *P* = 0.95

Note: given the number of correlations examined, at least one could be expected to be significant at $\alpha = 0.05$ by chance alone.

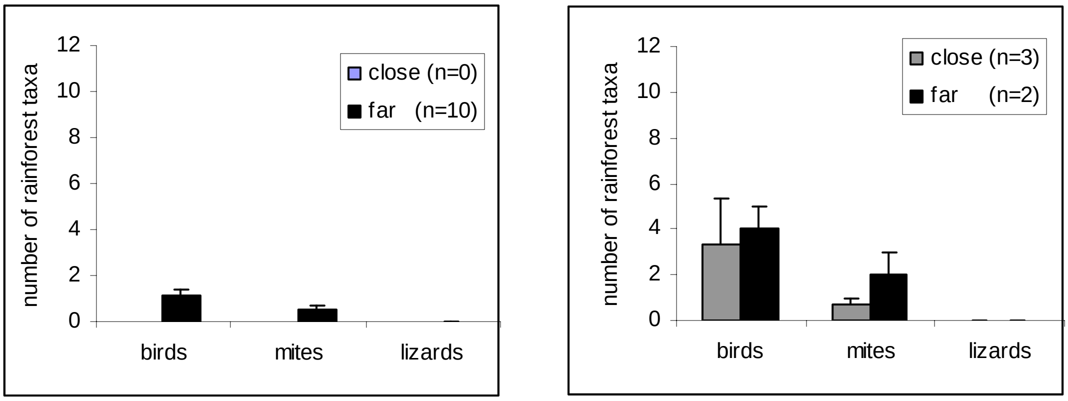
Effect of proximity to intact forest on the occurrence of rainforest biota in young rainforest plantings

In both study regions, the richness of rainforest birds, lizards and mites in young monoculture plantations appeared to vary with proximity to intact rainforest (Figure 6). Plantations adjacent to intact rainforest tended to have more rainforest taxa than sites distant from remnant or extensive forest. In contrast, the richness of rainforest taxa in cabinet timber plantations and restoration plantings did not appear to be strongly influenced by proximity to rainforest. However, sample sizes in the two proximity categories were low in most cases, precluding rigorous analysis of trends.

(i) monoculture plantations



(ii) cabinet timber plantations



(iii) ecological restoration plantings

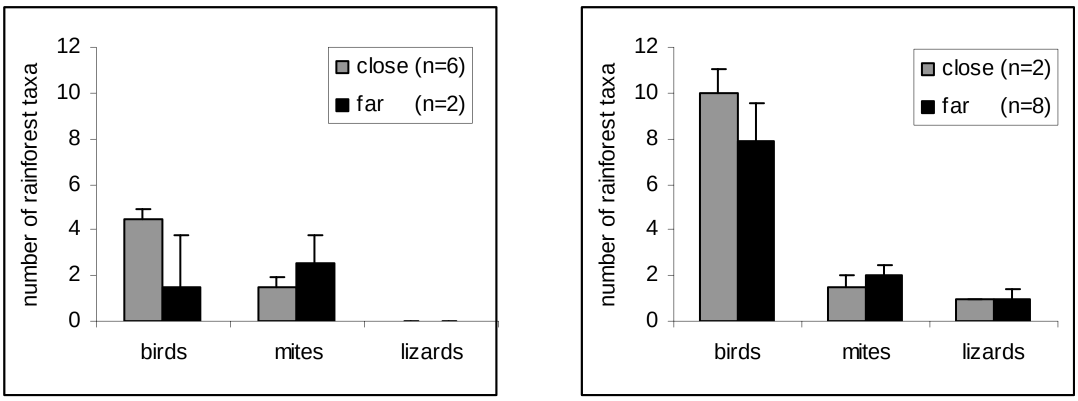


Figure 6 Richness of rainforest birds, lizards and mite taxa (mean, s.e.) in young rainforest plantings in the subtropics and tropics in relation to proximity to intact rainforest. Close = within 400 m of extensive or remnant (> 5 ha) rainforest; distant = more than 400 m.

Discussion

Biodiversity values of rainforest timber plantations

There have been few systematic surveys of faunal biodiversity in rainforest plantations (see Lamb 1998) and, until now, none which have contrasted the value of different types of plantations for rainforest biota. On the basis of the surveys conducted in this project, it is clear that timber plantations have much less value for rainforest biota, per unit area, than ecological restoration plantings, at least in the decade or two following establishment. These results are not surprising, given our limited knowledge of the habitat requirements of rainforest fauna (e.g., Kikkawa 1968, 1991 for birds), and the relatively poor development of suitable habitat in young timber plantations.

Nevertheless, it is possible that timber plantations could still make an important contribution to biodiversity conservation in former rainforest landscapes, as argued by a number of authors (Lamb *et al.* 1997, Lugo 1997, Lamb 1998), provided they recruit and maintain rainforest species over the longer-term. While some old plantations do support a rich diversity of rainforest plants and animals (e.g., those in north Queensland surveyed in this study and Keenan *et al.* 1997), the results from these plantations cannot be extrapolated to the broad-scale reforestation of cleared land, for several reasons. First, it is very unlikely that plantations established on cleared land will recruit as diverse a floristic understorey as plantations established by direct conversion of intact forest, where there is considerable regeneration from rootstocks and the soil seed bank (Fisher 1980). In contrast, the rootstocks and seedbank of rainforest trees are usually destroyed by a lengthy pasture phase (Hopkins and Graham 1984). Second, plantations established on cleared agricultural land must rely on the dispersal of seeds from forests elsewhere in the landscape to recruit rainforest plants. However, the dispersal of rainforest plants to revegetated sites is a function of their proximity to intact forest (e.g., Keenan *et al.* 1997, Lamb *et al.* 1997, McKenna 2001). Our data suggest this may also be the case for rainforest animals. For example, at even relatively small distances from intact forest, young timber plantations appear to recruit few rainforest birds, lizards and mites. Similarly, while most old plantations surveyed in this study were established adjacent to intact forest, we surveyed one old hoop pine plantation in the subtropics that was on private land, about 2 km from the nearest patch of rainforest. No rainforest birds, no rainforest lizards and only one taxon of rainforest mite were recorded in this plantation.

Third, the intensive management of plantations for timber production is likely to reduce the availability of habitat features required by many rainforest biota. For example, in north Queensland, many old plantations had not been thinned, or thinned only once, since establishment. These plantations are floristically rich and structurally complex (Keenan *et al.* 1997, Kanowski *et al.* 2003) and support a relatively rich rainforest fauna. In contrast, old plantations in the subtropics had been more intensively managed, with most sites subjected to several thinning cycles (Fisher 1980). The subtropical plantations are less floristically diverse and less structurally complex than the tropical plantations and support proportionally fewer rainforest birds, lizards and mites. However, this comparison is confounded by other differences between regions (e.g., the subtropical plantations experienced a drier climate than the tropical plantations, which may affect recruitment).

Biodiversity values of mixed species plantations

It is generally presumed that mixed-species plantations will support more biodiversity than monoculture plantations (Lamb 1998, Hartley 2002). While this notion is intuitively appealing, there are few data to test it. Our study found no evidence that mixed species plantations supported more rainforest birds, lizards or mites than monoculture plantations. There are several caveats to these results. First, the plantations were surveyed while still young. As plantations mature and the availability of resources such as fruit and nectar increases, mixed species plantations might be expected to support more rainforest animals.

However, many cabinet timber trees planted in Australia have wind-dispersed seeds (e.g., *Flindersia, Toona, Araucaria, Agathis, Eucalyptus*). The value of these plantations to rainforest frugivorous birds, at least, is unlikely to increase with maturity. Second, most mixed-species plantations surveyed in this study were located amongst cleared land, whereas monoculture plantations tended to be located adjacent to intact forest. That is, proximity to rainforest confounds our comparison of the biodiversity values of monoculture and mixed-species plantations.

The design and management of rainforest timber plantations for biodiversity conservation

The notion that rainforest timber plantations might make a significant contribution to biodiversity conservation is comparatively recent (e.g., Keenan *et al.* 1997; Lugo 1997; Lamb 1998). More traditionally, plantations have been viewed as an efficient means of producing timber. At present, plantation managers seem to hold to the traditional view. In subtropical and subtropical Australia, old plantations with high biodiversity values are currently being clearfelled. This is the most destructive of the possible options for harvesting these plantations identified by Keenan *et al.* (1997), other possibilities being selective thinning or adoption of a polycyclic silvicultural system. Although the harvested sites are being replanted, second rotation plantations are unlikely to support the same level of biodiversity as the old plantations. Not only will the soil seed bank and rootstock of native plants be depleted in second rotation plantations, but production methods have intensified in recent decades. Plantations are now managed using mechanical site preparation, the widespread application of herbicides and fertilisers, a reliance on a few superior provenances of trees and a reduction in rotation times (Fisher 1980, Constantini *et al.* 1997, Blumfield and Xu 2003). These measures are likely to reduce floristic diversity and structural complexity of plantations, and hence their value for rainforest biota.

Nevertheless, plantation managers may wish to consider the biodiversity values of rainforest timber plantations in the future, for various reasons. For example, environmental considerations are increasingly impinging upon the management of plantations, as a result of governmental regulation (e.g., the *Plantations and Reafforestation Act (1999)* NSW) and through certification schemes (e.g., DPI Forestry 2003). State government agencies and many private investors in timber plantations are also increasingly concerned to project a positive environmental image of their management practices, often for commercial reasons (e.g. Stanton 2000, DPI Forestry 2003).

Proposals for improving the faunal biodiversity value of plantations have been made by a number of authors (e.g. Catterall 2000, Boorsboom *et al.* 2002, Hartley 2002, Lindenmayer and Franklin 2002, Tucker *et al.* 2004). Common elements to these proposals can be grouped into two categories:

(i) changes to plantation design and management to increase the quantity and/ or quality of habitat features within plantations; and
(ii) the reservation of part of the plantation estate for biodiversity, either by the retention or restoration of native forest.

These potential actions are partly compensatory, in that management of plantations to promote habitat quality may reduce the proportion of the plantation estate that would need to be reserved for biodiversity, and vice versa (Lindenmayer and Franklin 2002).

Proposed changes to the design of rainforest timber plantations to improve their value as wildlife habitat have been listed under the rubric of ‘restoration forestry’ by Tucker *et al.* (2004). These include the greater use of the timber trees valuable to wildlife, notably fleshy-fruited plants, and in particular large-seeded species which are likely to be poorly dispersed to plantations. In both tropical and subtropical Australia, there are many candidate timber species that meet these criteria (Tucker *et al.* 2004), but knowledge of their silviculture is extremely limited (although this work has begun, e.g. Lamb and Keenan 2001). Tucker *et al.* (2004) also advocate the inclusion of some ‘keystone’ non-timber species in plantations, such as figs (*Ficus spp.*). Establishing plantations adjacent to native or replanted forest would also increase the likelihood that plantations are utilised by rainforest wildlife. However, this would also increase the risk that biota, including pests, weeds and exotic genotypes, could disperse from plantations into native forest, and these risks would need to be balanced against the potential benefits from such a strategy.

Proposed changes to the management of plantations to improve their value as wildlife habitat include measures to encourage the development and maintenance of a floristically diverse and structurally complex rainforest under the plantation canopy. These measures may include limiting the intensity and frequency of thinning operations, and selective or staggered harvesting regimes (Keenan *et al.* 1997, Lamb 1998, Hartley 2002).

However, until large-scale, long-term research is conducted on production-biodiversity trade-offs in rainforest timber plantations, the development of protocols such as those listed above will, in most cases, remain in the realm of reasoned speculation (Catterall *et al*. 2004).

Recommendations

- Management of rainforest plantations for their biodiversity values will require explicit design, management and harvest protocols to promote the development of habitat for rainforest taxa, and/ or the reservation or restoration of part of the plantation estate as native forest.
- At present, we can make few specific recommendations on measures required to support particular taxa in rainforest plantations. For example, of the taxa surveyed in this study, we could only confidently suggest measures for enhancing rainforest bird species richness (e.g., “promote the development of a diverse and structurally complex understorey of rainforest plants in plantations”). These measures might also enhance rainforest lizard richness, but we have no evidence they would enhance rainforest mite richness. The development of specific measures for mites and other taxa will require better knowledge of their habitat requirements.
- The development of suitable protocols for managing rainforest timber plantations for their biodiversity values will require investment in large-scale, long-term research on production-biodiversity trade-offs.

Acknowledgements

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13. Trade-offs between timber production and biodiversity in rainforest plantations: Emerging issues and an ecological perspective

Carla P. Catterall, John Kanowski, David Lamb, Daryl Killin, Peter D. Erskine, and Grant Wardell-Johnson

Abstract

During the past two centuries there have been three major paradigm shifts in the management of Australian rainforests and the use of their timbers: from felling native forests towards growing plantations; from viewing forests and plantations as mainly providers of timber to viewing them as sources of multiple benefits (e.g. timber, biodiversity, carbon sequestration, catchment protection, recreation, regional economic development); and from timber plantations being developed mainly by government on public land towards those established by private citizens, companies, or joint venture arrangements, on previously-cleared freehold land. Rainforest timber plantations are increasingly established for varied reasons, and with multiple objectives. Landholders are increasingly interested in the biodiversity values of their plantations. However, there are few guidelines on the changes to plantation design and management that would augment biodiversity outcomes, or on the extent to which this might require a sacrifice of production.

This paper presents a conceptual framework for considering the interactions and trade-offs between biodiversity values and timber production within plantations of rainforest trees in the Australian tropics and subtropics, and discusses aspects of design and management that are likely to affect the outcomes. Three forms of trade-off are discussed: those related to plantation design and management, those connected with timber harvest cycles, and those involving landscape issues and site configurations (allocation of different areas for different primary goals).

Existing knowledge suggests that plantation design, harvesting, and management regimes which maximise timber production will make a limited contribution to sustaining rainforest biodiversity. Different designs and management regimes may be able to produce better synergies between timber production and biodiversity, but to determine this will require: (1) implementation of a greater range of plantation designs, including those which purposefully aim for differing combinations of biodiversity and production, established in different landscape contexts; (2) quantitative assessments of both biodiversity and timber production made simultaneously at a range of these sites, at an appropriate stage of their development; and (3) a built-in research component, which includes biodiversity expertise, at the initial stages of large-scale tree-planting schemes. Measurements on existing sites, such as those planted during the Community Rainforest Reforestation Program (CRRP) scheme in the Wet Tropics, are providing some useful data, but a wider range of plantation designs and management regimes also needs to be established and monitored. Further development and application of site-based methods for quantitatively monitoring biodiversity values within mixed purpose plantation projects are also needed; these include assessment methods for plantations within environmental certification (e.g. eco-accreditation) schemes.

Introduction

Historical developments in northern Australian rainforest tree plantations

Historically, the main purpose of planting rainforest trees in tropical and subtropical Australia has been timber production (Dargavel 1995). Many rainforest trees produce timber which is highly valued, especially for cabinet-making (Herbohn *et al*. 2001). There has been extensive research and development worldwide to establish techniques for assessing and managing timber yield and values in monocultures of both softwood and hardwood species. This has lead to prescriptions for planting and growing trees in a manner which improves the production of timber, including recommended techniques for propagation, tree spacing, thinning, pruning, and fertilising (Evans 1992). While research of this type is lacking for many Australian rainforest species, there have been some trials with rainforest cabinet timbers grown in both mono-specific and mixed-species plantations (e.g. Cameron and Jermyn 1991, Lamb and Keenan 2001, Keenan *et al*. Chapter 10).

Most Australian rainforest timber plantations have traditionally contained only one tree species (mainly hoop pine *Araucaria cunninghamii*; occasionally kauri pine *Agathis robusta* or Queensland maple *Flindersia brayleyana*). Additional species have received attention mainly if they are unwanted competitors (flora) or pests (vertebrate and invertebrate fauna) that potentially reduce the quality or quantity of timber, or because they play a role in increasing productivity (e.g., soil microbes or predators of pests).

More recently, there has been a growing interest in the non-timber benefits from establishing tree plantations on cleared land (e.g., Abel *et al*. 1997, Dames and Moore NRM/Fortech 1999, Bennett *et al*. 2000, Borsboom *et al*. 2002, Lindenmayer 2002, Hobbs *et al*. 2003). There is a wide variety of potential benefits associated with the establishment of tree plantations, including the maintenance or restoration of biodiversity, provision of ecosystem services such as climate regulation and water purification, regional economic development, and recreational opportunities (Table 1).

Table 1 Commonly perceived aspects of the value of plantations in rainforest regions. These are not independent of one another.

Type of value or benefit	Subcategories	Examples
Productive	Timber	Woodchip; veneer, plywood and sawn timber for furniture or building materials
	Other tree products	Fruits, seeds, rubber, drugs
Environmental	Ecosystem services	Climate regulation, water purification, land stabilisation
	Biodiversity	Variety of organisms, ecological processes, physical structure
Social	Community development	Regional economy, employment
	Individual wellbeing	Recreation, aesthetics, spirituality

However, until recently, many of these concepts have not been viewed from a utilitarian perspective. Therefore, there has been little research and development into the forms of plantation design and management that could improve their contributions to the value of plantations, or into techniques for assessing these contributions. Furthermore, most of the work cited above has involved *Pinus* or *Eucalyptus* plantations in temperate Australia.

In the early decades following European colonisation of northern Australia (c. 1850-1920), timber was readily obtained from old-growth tropical and subtropical rainforests that were formerly found in a few parts of eastern Queensland and New South Wales. However, the rainforests on arable land were soon cleared for agriculture (Adam 1994, Lamb *et al*. 2001, Catterall *et al*. 2004). By the 1980's there was rising public concern that these landscapes had been over-cleared, and that many of the remaining rainforests (mostly on mountain ranges) were being logged (e.g. Cassells *et al*. 1988). This concern was accompanied by a greater recognition of the environmental and social values of intact rainforest areas (Webb and Kikkawa 1990). These were viewed increasingly as places of beauty and grandeur that are especially rich in species of flora and fauna, and which play a role in local and global climate regulation. Rainforests may be defined by the presence of a closed canopy of broad-leaved trees of particular plant families or genera, with life-forms such as vines and epiphytes often present (see Bowman 2000, Adam 1994 for further discussion of rainforest definitions). In addition, the small area of rainforest found in Australia is noteworthy for its evolutionary distinctiveness, endemic fauna and flora, its links to Gondwanan rainforests which once covered the continent, and the high proportion of Australia's overall biodiversity that it supports (Adam 1994).

Between 1980 and 2000, various governmental initiatives either severely restricted or ended timber harvesting in native rainforests, which were by then seen mainly as a conservation resource (Catterall *et al*. 2004). Following these initiatives, most future rainforest timber supplies would have to be obtained from other sources. Candidates included imports of timber from other tropical countries, use of the existing hoop pine plantation areas, and the conversion of cleared agricultural land (e.g. pasture or cropland) to forest plantations. Since felling of timber from overseas rainforests raises environmental concerns, and the existing hoop pine plantation estate is relatively small in area, there has been increasing interest in establishing plantations on former agricultural land in rainforest regions (e.g. Lamb and Keenan 2001). Suggestions for plantation species have ranged from eucalypts (e.g. Annandale *et al*. 2002) to mixed-species cabinet timber plantations of mainly native rainforest trees (e.g. Lamb 1998). By definition, the latter represents some improvement in rainforest biodiversity, compared with either eucalypt or hoop pine monoculture (Lamb and Keenan 2001).

Also during this time, early experiments in the planting of rainforest trees to restore rainforest ecosystems began, with a view to helping sustain rainforest biodiversity (e.g. Tracey 1986, Kooyman 1991). To achieve these ends, a completely different plantation design and management system was devised, with a focus on achieving a dense canopy, establishing a high diversity of native plant species and a complex multi-layered vegetation structure, providing habitat for a diverse native fauna, and fostering the potential for successional processes resembling those of rainforest (Goosem and Tucker 1995, Kooyman 1996). Such plantings have been used to restore forest to streambanks, to provide links between remnant forest patches, and to increase the size of remnants, with no expectation that timber will be harvested from them. However, the high establishment costs for such plantings limits their use over large areas (Table 2, see also Erskine 2002).

Plantations for timber and/or biodiversity?

Clearly, it would be useful to know how to design plantations which have timber benefits together with environmental and social benefits, and to know the limits to the compatibility of these goals. It seems that many private rural landholders have planted trees during the past decade with an expectation for both timber and biodiversity benefits (Emtage *et al*. 2001). There is also a growing expectation that publicly-funded tree planting projects should contribute to achieving environmental goals.

Furthermore, the biodiversity values of plantations may soon be associated with some financial benefit for landholders. For example, this could occur through certification schemes (e.g. Nussbaum *et al*. 2001) which include an "eco-accreditation" element. Such schemes may improve access to particular markets (such as "ethical investment" programs), or may incorporate "biodiversity credits" which could be used to offset other forms of environmental damage caused by a business (e.g. Binning *et al*. 2002).

Table 2 The current spectrum of reforestation planting styles in rainforest landscapes of tropical and subtropical Australia. Note that there is considerable variation within each type of planting, and the two may intergrade in practice (after Catterall 2000, Lamb and Gilmour 2003, Catterall *et al*. 2004, Kanowski *et al*. Chapter 12).

	Main goal of planting	
	Timber production[+]	Ecological restoration
Tree species diversity	Low (typically 1-10)	High (tens - over 100)*
Species types	May include a substantial proportion of exotic species, eucalypts, and/or wind-dispersed rainforest species.	Few or no exotic species; few eucalypts; many fleshy-fruited rainforest species*.
Planting density	Lower (c. 1,000 stems/ ha or less)	Higher (c. 6,000 stems/ha)
Management	Grass and understorey suppressed with herbicide, slashing, and/or stock grazing. Fertilizers added. Stems and lower branches pruned, stems progressively thinned.	Initial herbicide followed by heavy mulching and selective weeding and/or herbicide; stock excluded. Little or no fertiliser. Native understorey allowed to develop.
Location	Often on level ground, fertile soils	Often in areas not desired for production – e.g. steep slopes, creek banks.
Cost (*circa* 2000)	$4-8,000	$20-25,000

[+] *Incudes both monoculture and mixed-species plantations whose typical occurrence has differed (e.g. industrial plantations have been monocultures over large areas; mixed-species plantings have been established as smaller areas within multi-purpose agroforestry landholdings).*

* *Native rainforest in higher rainfall areas of subtropical and tropical Australia has 50-100 tree species/ha, few or no exotics, few or no eucalypts.*

Table 2 contrasts the design and maintenance characteristics of plantations aimed at rainforest timber production with those aimed at ecological restoration. Until recently, there has been an information vacuum concerning either the biodiversity values of the timber plantations or the timber values of the ecological restoration plantings. Most past plantation research and development has focussed on maximising timber yields, and much of the available technical advice on planting and managing timber trees has remained directed towards this.

Guidelines for improving biodiversity outcomes in rainforest timber plantations have been suggested (e.g. Keenan *et al*. 1997, Lamb 1998, Lamb and Keenan 2001, Tucker *et al*. 2004, Kanowski *et al*. Chapter 12), but until very recently these were based on reasoning informed by a knowledge of ecology and natural history and by general field experience, rather than being the outcomes of systematic research and development.

The extent to which such hybrid approaches produce synergistic or compromise outcomes for either biodiversity or wood production is largely untested. Without a better understanding of how to design plantations to meet multiple goals, there is a risk that investment in plantings will not achieve the outcomes now desired by many landholders and investors.

Quantitative assessments of plantation benefits and values, for different designs and management regimes, are needed to answer these questions. However, until very recently, the notion of quantifying biodiversity values was impractical; it is barely two decades since the term "biodiversity" was coined, and conservation values were previously viewed mainly in terms of rare vertebrate species on the verge of extinction. More recently, the notion of biodiversity as the variety of all life-forms, including genetic diversity, species diversity, and ecosystem diversity, has become widely accepted in both scientific and public arenas (Cogger 1994). The protection of biodiversity, together with the processes that sustain it, is also now widely incorporated in legislation (e.g. *Commonwealth Environmental Protection and Biodiversity Conservation Act*, and State Acts relating to land, vegetation and waterway management, nature conservation, and land use planning).

Biodiversity assessment methods are being developed and improved (see for example Margules and Austin 1991, Landsberg *et al*. 1999). The "Biodiversity Values in Reforestation" project of the Rainforest Cooperative Research Centre (CRC) has begun developing and applying techniques for assessing biodiversity attributes in replanted rainforest sites (Wardell-Johnson *et al*. 2001, Catterall *et al*. 2004, Kanowski *et al*. 2003, Kanowski *et al*. Chapter 12, Wardell-Johnson *et al*. Chapter 11).

The assessment has two components:

- First, a broad range of biodiversity attributes are measured in target plantations. These include forest structure (e.g. canopy height and cover, stem densities and diameters, woody debris), biota (e.g. birds, reptiles, invertebrates, plants), and ecological processes (e.g. seed predation patterns, litter decomposition).
- Second, the numerical values of attributes derived from these measurements (e.g. the proportion of bird species that are "rainforest-dependent") are compared with those obtained from a set of reference sites, within both pasture and intact rainforest, whose background environmental properties broadly match those of the replanted sites.

This paper presents, from an ecological perspective, a conceptual framework for considering the interaction between biodiversity values and timber production within plantations of rainforest trees. It also considers aspects of plantation design and management that are likely to effect these outcomes. This treatment draws upon current knowledge (where available), but the intent is also to stimulate consideration of gaps in current knowledge and to identify future research priorities.

Trade-offs involving biodiversity and production in relation to plantation style

One form of trade-off between biodiversity and timber production is determined by stand design and management. Features that are associated with higher biodiversity values include: denser tree spacing, more tree species (including fleshy-fruited species), greater variety of life-forms and age-classes, less pruning and understorey suppression, and greater total plantation area (patch size). The context of a plantation will also affect its biodiversity, as discussed later. These issues have been discussed by Catterall (2000), Kanowski *et al*. (2003), Nakamura *et al*. (2003), Tucker *et al*. (2004), Catterall *et al*. (2004), Proctor *et al*. (2004), Kanowski *et al*. Chapter 12 and Wardell-Johnson *et al*. Chapter 11.

Figure 1 illustrates these issues by plotting the hypothetical long-term (averaged over several rotation cycles) biodiversity values of a range of differently-managed sites against their wood production values. "Biodiversity value" is more accurately termed "rainforest biodiversity value", which is defined as the development of a rainforest-like set of biota and ecological processes, which can be quantitatively assessed by taking measurements from reference sites within intact rainforests growing under specified geographical and environmental conditions (after Catterall *et al*. 2004). An underlying simplifying assumption in Figure 1 is that the sites are similar in area.

There are two forms of relationship between production and biodiversity. First, an increase in tree cover from "pasture" to "forest" is accompanied by an increase in both biodiversity and timber values, i.e. at this level the relationship is positive (Figure 1). However, if only land that is "forested" is considered, the relationship becomes negative, and different plantation styles and management regimes will involve trading off production goals against biodiversity goals. At one end of the spectrum, rainforests that are lightly harvested will retain high rainforest biodiversity values, but will have low wood production. At the other end, an intensively-managed hoop pine plantation, established far from any other forest on a long-cleared site, could maximise the production of high-value timber, but would support limited rainforest biodiversity.

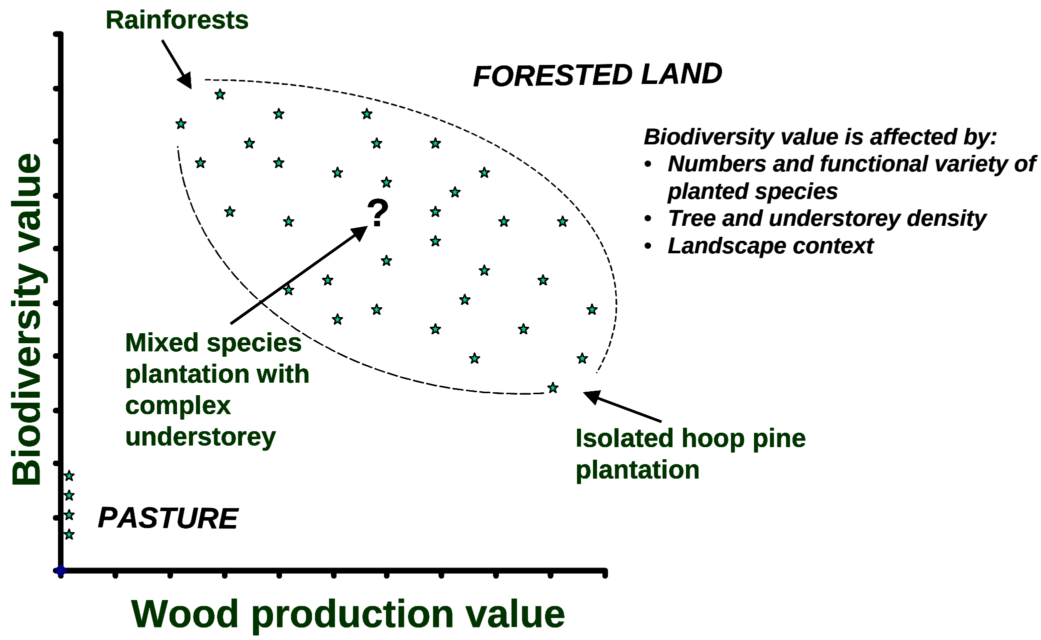

Figure 1 Hypothesized relationships between a site's long-term wood production value and its biodiversity value. Each point represents a site of fixed area and uniform type, which is characterised by a different form of tree cover and management regime (some examples are labelled). The values represent outcomes across a full harvest cycle for plantations. For simplicity, points are shown only for pasture (few trees) and relatively dense tree cover (forest), although sites with other forms of tree cover could occupy the space between pasture and forested land. "Rainforest" sites are assumed to be selectively harvested in a manner that has little impact on their biodiversity. Factors likely to strongly influence biodiversity value are listed.

Somewhere in between may lie various design and management compromises, for example, a moderately-isolated mixed-species plantation, in which the development of a complex native understorey is encouraged, may provide intermediate biodiversity value with moderate wood production. If it included non-timber trees of types that are particularly important to wildlife (such as native figs), the biodiversity would improve further, but its timber value would be reduced.

A plantation established adjacent to rainforest, without thinning or pruning, may acquire relatively high rainforest biodiversity value, but at some cost to wood production. The shape of the upper bound to the cloud of points in Figure 1 is important. The hypothetical convex edge, as drawn, allows for some plantation styles that give potential synergy between biodiversity and wood production (with a moderate increase in biodiversity value for very little decrease in production value). If the upper bound were straight, then increased biodiversity would always be traded for decreased production. If it were concave, then attempting to achieve a compromise between biodiversity and wood production through varying such design features as tree spacing or species composition would be a waste of time and resources. At present, we do not have the data to assess whether the relationships hypothesized in Figure 1 are empirically realistic. However, it should be technically possible to obtain such data, if plantations matching the range of design and management options could be found.

Trade-offs between biodiversity and production involving timber harvest cycles

A second form of trade-off between biodiversity and production lies in the harvest cycle. We know that, in some situations, old (40-70 year) plantations of rainforest timber species (hoop pine, kauri pine, Queensland maple), which were initially established as monocultures, can come to support a diverse rainforest biota which is similar in many respects to nearby mature rainforest (Keenan *et al*. 1997, Kanowski *et al*. Chapter 12). However, the overall biodiversity contribution that such plantations might make must be viewed over a complete rotational cycle. Figure 2 shows possible rates of development in both biodiversity and wood production, for a hypothetical timber plantation that is harvested at 30 years (caveats concerning the realism of the timescales are discussed later). "wood production" and "Biodiversity value" (again, comprising characteristics of rainforest) refer to the standing level of either at any given time.

In the examples shown in Figure 2, trees begin to senesce and decay after around 80 years, and the volume of standing timber hence declines. Since this provides important resources for fauna (such as food, nest hollows, and ground cover), biodiversity continues to increase after this time. Two scenarios for biodiversity development are shown: fast and slow. Under the fast scenario, more than half the rainforest characteristics present by 100 years have been acquired by around 30 years, compared with less than one-fifth under the slow scenario. The plantation's nature and context will affect both its maximum biodiversity value and the rate at which biodiversity develops. The biodiversity value of a mature rainforest is not shown at Figure 2, but is expected to exceed the plantation maximum (see for example, Kanowski *et al*. Chapter 12, Wardell-Johnson *et al*. Chapter 11).

Under a 30-year harvest rotation with slow biodiversity development (Figure 2), any given area of this plantation would never have much "biodiversity value", although its value could still exceed that of the same area of pasture or cropland. Under the rapid development scenario, the biodiversity value at harvest is better. However, while the realised wood value is determined by the harvest quantity, the overall biodiversity value is the average over the 30 years of development; still not particularly high. Figure 3 shows the joint pattern of development for biodiversity and wood; until around 40 years, the comparative rate of increase in wood value is much faster than the comparative rate of development of biodiversity value; at around 80 years both are relatively high; but subsequent increases in biodiversity value are achieved with only small gains in wood production.

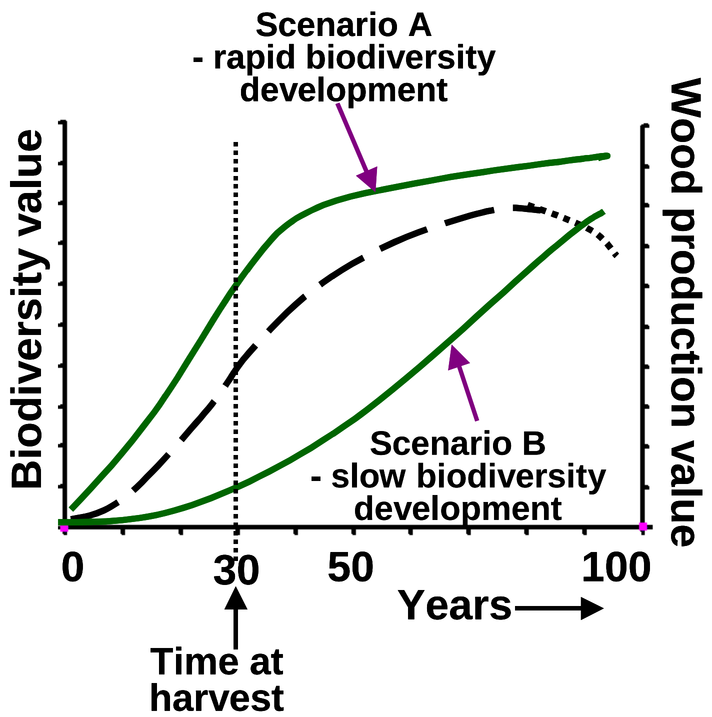

Figure 2 Possible patterns of temporal development of the value of a rainforest plantation in terms of rainforest biodiversity under two scenarios A (rapid) and B (slow) of development (solid lines); and wood production (dotted line). The timescales are hypothetical. Biodiversity value is conceptualised as the plantation's measured levels of attributes which characterise intact rainforest (and are lost or greatly reduced following clearing). Wood production is conceptualised as the value of the timber that could be obtained from clearfelling.

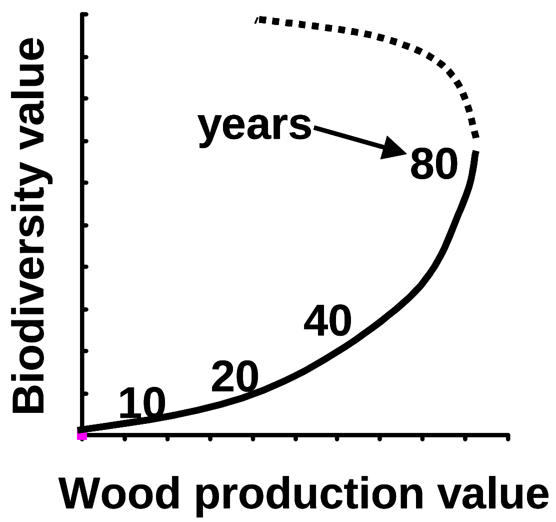

Figure 3 The relationship between the wood value and biodiversity value during growth of a rainforest plantation, according to the slower biodiversity development scenario of Figure 2. The numbers show the plantation's age.

The curves in Figure 2 are not intended to be fully realistic. In hoop pine plantations currently under management for peak wood production, the recommended rotation length is 45-50 years, at 400-1,000 stems/ha (Hogg and Nester 1991, Keenan *et al*. 1997, Lewty and Last 1998). The financial outcomes of decisions concerning the design and management of plantations are influenced by factors such as costs of establishment and management, discount rates and market prices. The value of standing timber in hoop pine plantations would continue to increase well past 100 years, while its net value over a rotation would diminish.

From a biodiversity perspective, rainforest trees typically do not begin to form large hollows until they are at least 200 years old. However, the relevant data on long-term tree performance is incomplete and widely scattered. If realistic curves were available for both production and biodiversity, optimisation techniques could suggest rotation cycles that would be appropriate for different levels of compromise between biodiversity and financial outcomes. Precise quantification could be very difficult, since many factors complicate both the value of timber and the achievement of biodiversity, and both are difficult to measure, especially if there are several tree species involved. Nevertheless, even approximate calculations may be a better decision guide than the often-incorrect assumptions currently being made by many landholders and managers.

In the example given (Figure 2), the biodiversity value would vary greatly during the course of a rotation. But if a plantation comprised stands of differing ages, this within-stand variation would be buffered at the scale of the whole plantation, even though the long-term average would not appear to differ between the two scenarios. However, the greater continuity in habitat availability could improve the overall biodiversity outcome, since older forest stages would always be present somewhere in the plantation and it would not be necessary for species to repeatedly recolonise the plantation from suitable habitat elsewhere (with possible associated time-lags in recruitment) as each cycle progressed. From a production viewpoint such asynchrony would also be desirable as it generates regular returns on investment (and is currently the practice in many plantation forests).

It could be argued that selectively harvesting individual stems would ensure that timber removal had less effect on biodiversity value than if all stems were clear-felled simultaneously but less frequently (e.g. Keenan *et al*. 1997, Lamb 1998, Hartley 2002). However, the resultant thinning of the stand might be detrimental to its ability to provide suitable habitat for rainforest-dependent fauna and fauna, depending on its effect on the plantation understorey (see previous section). Scattered patches of small-area clearing therefore may be more compatible with maximising biodiversity values than uniform stem removal, and would have an outcome similar to the asynchronous rotations discussed above, and perhaps more similar to the natural disturbance mosaic in rainforests, to which rainforest species are adapted. In fact, canopy gaps have been advocated as a means of increasing biodiversity at the plantation scale in temperate forest plantations (Spellerberg and Sawyer 1997, see also Lindenmayer and Franklin 2002). However, the lack of information on the basic biology, population dynamics, and recruitment processes of most rainforest flora and fauna species, coupled with the very large number of species involved, prevents reliable prediction of the outcome of such differences in management.

A more effective approach than such theoretical speculation would be to determine empirically the biodiversity, timber, and economic outcomes of differing forms of plantation design and management in long-term large-scale field trials.

Biodiversity, production and landscape issues

Importance of context, configuration and area to biodiversity

Both the absolute size (area) of a plantation, and its landscape context are important to its acquisition of rainforest biodiversity value, because they affect the ability of new species to colonise the site. Large natural areas nearby are likely to be a source of native colonists but small and more distant remnants may contribute little. In particular, many vertebrate species of rainforest are sensitive to patch area, to the amount of suitable habitat nearby (including the presence of direct habitat corridors), and to the amount of suitable habitat in the surrounding landscape. The recruitment of rainforest flora to plantations will also be affected by these factors, because birds and mammals are the main agents of seed dispersal for most plants of tropical and subtropical rainforest, as well as being important predators of seeds and seedlings (Willson *et al*. 1989, Kanowski *et al*. Chapter 12).

Patches less than five hectares in area, even if high-quality rainforest habitat, appear unlikely to ever support the full range of rainforest birds and mammals, no matter what their design and management, unless they are very close to larger areas of high-quality rainforest (Price *et al*. 1999, Catterall *et al*. 2004). Small patches of production forest are also likely to be less economically viable than larger patches.

Plantations can also be colonised by introduced species (including grazing stock). Both exotic and native colonists may also be influenced by the size and shape of the plantations, in ways that interact with the landscape context. For example, a narrow rectangular-shaped plantation surrounded by rainforest should acquire biodiversity values more rapidly than the same area if square, whereas the reverse is likely if the plantation were surrounded by pasture, and the maximum value reached in the latter case would also be lower. It is even possible that biodiversity values per unit area in very large plantations may fall, due to the increasing proportion of the plantation that is distance from any native forest in the surrounding landscape.

In general, a plantation would be expected to acquire greater levels of rainforest biodiversity if its context included: greater proximity to native forest, fewer weedy species in the landscape, more native forest in the landscape, and less exposure to adverse processes from adjacent areas (e.g. dry winds, insecticide drift or runoff, heavy grazing).

Plantations can also provide off-site biodiversity benefits which stem from the landscape context in which they are placed. These add further complexity to the accounting of their biodiversity outcomes. For example, a structurally simple monospecies timber plantation which borders a small remnant of intact forest may provide a buffer against exposure to wind and sun, so that the remnant experiences a more rainforest-like microclimate, and can support more of the rainforest flora and fauna that are sensitive to dryness than would be the case for an unbuffered remnant (see also Tucker *et al*. 2004). Timber plantings could also provide benefits to remnant rainforest patches by providing stepping stones or corridors of habitat between them (the quality of habitat that animals require for movement is likely to be less than that required for residency). However, negative impacts on rainforest biodiversity from such buffers or links are also possible at a broader spatial scale, for example if the plantation acted as a source of invasive exotic species, or if the plantation trees belonged to a non-local genetic race of a plant species present within the remnants.

Area also complicates the measurement of biodiversity value. Species-area curves are nonlinear; the rate, per unit area, at which new species are recorded at a site declines as the surveyed area increases (Connor and McCoy 1979). This affects estimates of the contribution of a plantation's area to its biodiversity. For example, if a 20 ha timber plantation contains 50% of the rainforest-dependent species that occur in 20 ha of rainforest, this does not mean that 40 ha of the same form of plantation would support 100% of the rainforest species, or that 10 ha of replanted rainforest would be equivalent to a 20 ha timber plantation. Even 1,000 ha of this plantation is unlikely to contain all the rainforest species that can be found in 20 ha of rainforest, because habitat elements essential to some species will always be missing.

While it is relatively straightforward, with current methods, to compare sites of the same area which differ in their type of planting, and to compare different areas whose type of planting is the same, it is more difficult to compare quantitatively the biodiversity values of two plantations that differ in both area and type of planting. Resolving these issues requires better information on species-area accumulation curves for different types and ages of plantation.

Trade-offs between biodiversity and production involving the configuration of site areas

A third form of trade-off between biodiversity and production lies in the configuration of the planted area. Most of the previous discussions have assumed that the design (e.g. tree spacing, species selection, early management) of a plantation is uniform over its entire area, and have considered the consequences of altering the design and harvesting over this area. An alternative would be to incorporate spatial heterogeneity into plantation designs, for example by designing and managing some sections of the nominal plantation area for timber production (with little expected biodiversity benefit), while other sections are allocated to restoration planting (designed for biodiversity outcomes, but with no expected timber harvesting). This option has also been discussed by Lamb (1998, 2001) and Kanowski *et al*. (Chapter 12), and more generally by Lindenmayer and Franklin (2002). The trade-off between biodiversity and timber production in this case will largely depend on what proportion of the land area is allocated for each purpose. However it could also be influenced by design and management practices within a section (including hybrid management, for example, if the "timber" section was designed to forego some timber production in exchange for incorporating some design aspects to improve biodiversity, such as fig trees or understorey development; or if the "biodiversity" section mainly comprised timber tree species with a relatively open spacing).

Without further empirical testing, it is not possible to say whether such area-based trade-offs could produce better production-biodiversity compromises than simply altering design and management within the plantation structure. The preferred type of trade-off is also likely to be affected by the context and nature of a plantation site. For example, where a previously-cleared site includes a waterway, then allocating land for ecological restoration along the waterway would be desirable because it also meets environmental goals other than biodiversity (such as those relating to water quality and streambank stabilisation). If a plantation site is far from intact rainforest, allocating a significant area to ecological restoration may be more important than if it is adjacent to a large rainforest remnant. Different forms of trade-off might suit different scales of enterprise, for example a small landholder might choose to compromise mainly on plantation design whereas a large-scale business might prefer to allocate sections of land for different purposes.

Area-based trade-offs could also be the most pragmatic from the viewpoint of balancing different primary objectives. Spatially separating the area of production forest from ecological plantings could assist in project management and accounting procedures, especially in large-scale projects that involve both private and public sector funding.

Conclusions and recommendations

During the past two centuries there have been three major paradigm shifts in the management of Australian rainforests and the use of their timbers (described in Adam 1994, Lamb and Keenan 2001). The shift has been from felling native forests towards growing plantations; from viewing forests and plantations as mainly providers of timber to viewing them as sources of multiple benefits (timber, biodiversity, carbon sequestration, catchment protection, others), and from timber plantations being developed mainly by government on public land towards those established by private citizens, companies, or joint venture arrangements, on previously-cleared freehold land.

This paper has examined and discussed, from an ecological perspective, the ability of plantations to act as a source of both timber and biodiversity benefits. The Commonwealth of Australia has committed to expanding its area of commercial forestry plantations to supply current and projected demand in pulp, sawn timber and timber products. But such plantations have limited biodiversity value.

Rainforest biodiversity in subtropical and tropical eastern Australia is a significant conservation issue at state, national and international levels, and rainforest restoration is part of the conservation strategy (Tucker *et al*. 2004, Catterall *et al*. 2004). But replanting rainforest for strictly biodiversity purposes is expensive, and unlikely to be carried out over large areas. Mixed-purpose plantations offer the prospect of some financial return, which might make reforestation more attractive to landowners, and thereby increase the opportunity to reforest larger areas of cleared land. However, motivations of small-scale private landholders are complex (Emtage *et al*. 2001), and affected by perceived threats to harvest security that may occur if a plantation's biodiversity values became high.

Current knowledge has enabled us to identify different aspects of plantation design and management which may either constrain or enhance a plantation's ability to provide both biodiversity and timber. In the future, various forms of environmentally-targeted incentives, such as environmental certification, carbon credits, salinity credits and biodiversity credits, may offer a changing economic context for privately-owned timber plantations (e.g. Binning *et al*. 2002). In this new context, changes to management or harvesting practices (such as the development of spatially heterogeneous plantations, or an extended rotation length) may become more economically attractive.

The rainforest landscapes of tropical and subtropical Australia offer an opportunity to compare the performance (for biodiversity, timber, and other attributes) of a range of different plantation designs and approaches to management, including timber monocultures, a variety of mixed-species timber plantations, and species-rich, complex restoration plantings (Table 2). Most of these are still young in successional terms (less than 20 years old) and there remains much to be learned about the rates and patterns of their biodiversity development (c.f. Figure 2). At these sites, strategically-timed ongoing monitoring can provide results to help improve the design of future plantation systems, and contribute to the development and application of environmental certification schemes.

In the Queensland wet tropics, the CRRP sites provide an opportunity to track the performance of tropical mixed-species timber plantations. On upland basalt soils, the best-developed and managed of these plantations, after 5-10 years, had a low to moderate ability to support rainforest biota (Kanowski *et al*. 2003, Catterall *et al*. 2004, Kanowski *et al*. Chapter 12). However, many of the CRRP plantings are dominated by eucalypts, rather than rainforest trees (Wardell-Johnson *et al*. Chapter 11), and most are very small in area (<5 ha, Catterall *et al*. 2004). Furthermore, their characteristics do not meet the requirements of rigorous experimental design (replication of sites with controlled variation in factors of interest). Thus they allow only limited exploration of the trade-offs involved in rainforest biodiversity and timber production.

It will be difficult to further develop a sound basis for designing plantations which provide novel combinations of timber, biodiversity, and other benefits, unless: (1) a greater range of plantation designs are established and tested, including carefully planned projects that aim to provide differing combinations of biodiversity and production, set within different landscape contexts; (2) there are simultaneous quantitative assessments of both biodiversity and timber at a range of plantation styles, at an appropriate stage of their development; and (3) there is a built-in biodiversity research component at the initial stages of large-scale tree-planting schemes.

Funding is needed to encourage a research-based approach that takes controlled risks with different forms of plantation design, management and harvest schedules. This also requires ongoing dialogue between forest restoration scientists and commercial forestry practitioners, and a wider recognition that these designs can be an investment in knowledge generation, for use in future decades rather than a few years hence.

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