Trees in the landscape

Technical Report 215

Biodiversity outcomes from eucalypt plantation expansion into agricultural landscapes of southern Australia: A review

P Grimbacher

CRC for Forestry
Researching sustainable forest landscapes

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Public report

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Executive summary

In recent years, industrial eucalypt plantations in Australia have grown rapidly to about a million hectares (as at 2010). Most new eucalypt plantations are established on what was previously agricultural land, and remnant vegetation often becomes embedded within plantation estates. Given rapid landscape change, there is significant industry, community and scientific interest in how biodiversity values are affected by forestry operations within plantation landscapes. In recent years, new research has been published on biodiversity-related issues relevant to Australian eucalypt plantations, but this new research has not yet been synthesised.

The aims of this review were to:

- exclusively focus on industrial eucalypt plantations in southern Australia where most of the estate expansion has occurred
- consider the full life-cycle and variety of plantation operations
- include land-use history and landscape context
- broadly consider all the different ways in which plantations affect biodiversity
- synthesise recent research on biodiversity impacts of plantations
- test numerous perceptions held by the general public concerning biodiversity and eucalypt plantations.

The results of studies on eucalypt plantations in Western Australia, South Australia, Victoria and Tasmania were synthesised. Most of the studies compared biodiversity in plantations against remnant eucalypt vegetation and cleared agricultural land. Studies of birds dominated the literature, although a wide range of taxa were covered.

A review of the literature showed that eucalypt plantations can provide habitat for many taxa, although generally the biodiversity value of plantations falls between that of agricultural land (lowest) and native eucalypt vegetation (highest). Animals with good mobility (birds, insects with wings and large macropods) were best represented in eucalypt plantations, whereas other mammals, reptiles, amphibians and especially native plants were poorly represented.

The presence of biological legacies, especially old trees embedded within plantations, strongly improved the biodiversity values of eucalypt plantations. This is because structures such as old trees provide special resources (e.g. hollows) that many organisms require, but which are absent in young trees grown in plantations.

Close proximity of remnant vegetation resulted in moderate biodiversity gains in plantations, although organisms did not always respond consistently. Some species in eucalypt plantations may only be present due to 'spillover' from adjacent remnant eucalypt vegetation. These species only partially use plantations as habitat, probably due to the lack of some key resources in plantations.

In addition to providing habitat and resources for some species, eucalypt plantations may create other biodiversity benefits. Replacing a pasture matrix with eucalypt plantations may positively influence the biodiversity in patches of remnant vegetation for several reasons, by:

- reducing the negative effects of livestock grazing
- reducing the magnitude of biophysical and biological degradation at edges of remnants
- facilitating the dispersal of organisms between remnants, although at this time there is little supporting evidence.

By contrast, establishing eucalypt plantations on agricultural land may negatively affect species that prefer:

- grasslands or open habitats, or
- the edges of woodland or forest.

Eucalypt plantations do not appear to be a source of pests that colonise adjacent land. Rather, it is more likely that pests move into plantations from pasture or native remnants. There was no evidence to support the perception that eucalypt plantations have higher levels of predation than native eucalypt forest and thus may act as population sinks for fauna moving out of remnant eucalypt vegetation.

In some cases, gene flow out of eucalypt plantations into adjacent native forest may occur when pollen from plantation trees is transported onto the flowers of local native eucalypt species and leads to hybridisation. However, several actions can be taken to manage this risk. Less is known about the risks from wildlings arising from the seeds of plantation species being transported into adjacent native forest, although at this stage these risks are thought to be low.

Use of herbicides in eucalypt plantations reduces floral diversity, structural diversity and probably the diversity of other associated organisms. Insecticide use has a temporary negative affect on terrestrial invertebrates in eucalypt plantations but is also a threat to aquatic biodiversity, off site, if mobilised via water courses, although there is little evidence for this occurring.

Eucalypt trees in plantations use more groundwater than grasses or annual crops. In landscapes where groundwater is saline, lowering water tables may lead to positive biodiversity outcomes. But if eucalypt plantations occupy too much of any particular catchment, they may have an adverse effect on stream-flows and could potentially affect aquatic biodiversity. Planning regulations aim to prevent this. Eucalypt plantations do not appear to have a negative impact on water quality.

The negative effects of harvesting on organisms residing in eucalypt plantations may be minimised by spatially interspersing plots to be harvested at a similar time. This would allow organisms in plots being harvested the opportunity to disperse to neighbouring unharvested plots, thereby minimising the stand-level effects and hydrological effects across the landscape.

At the scale of individual plantations, the net outcome of eucalypt plantation establishment may not always be positive as the factors that influence biodiversity vary considerably, particularly the amount of native vegetation retained during plantation establishment. However, this review has shown that at a larger landscape scale the net outcome of eucalypt plantation establishment on former agricultural land is likely to be positive.

Although a considerable amount of new literature has been published in the last few years, many gaps remain in our knowledge of the biodiversity values of eucalypt plantations. The establishment of large-scale eucalypt plantations is relatively recent, so there has been limited time for biota to respond to landscape changes. The impacts of harvesting are still unknown, including whether this resets the 'successional clock' in subsequent rotations. In this regard, no studies have tracked biodiversity changes in eucalypt plantations through a full rotation. Thus, longer term studies are needed to complement the growing body of short-term studies.

Past reviews have suggested that management actions that maximise structural diversity (e.g. promoting an understorey) may lead to biodiversity gains in plantation landscapes. However, these strategies are probably incompatible with most eucalypt plantations that employ short-term rotations (10 to 15 years). Given the disproportionately positive contribution of remnant vegetation and biological legacies to landscape-level biodiversity values, management efforts that aim to maximise landscape-level biodiversity may be better focused towards conserving and rehabilitating remnant vegetation within plantation estates rather than maximising structural complexity within plantations.

1. Introduction

1A. The need for a review of biodiversity and eucalypt plantations

In Australia, forestry plantations comprise hardwoods (predominantly *Eucalyptus* species) and softwoods (predominantly *Pinus* species), with roughly a million hectares of each genus (as at 2010) (Parsons *et al.*, 2006; Gavran & Parsons, 2010). The area of softwood plantations has remained the same for the last decade. However, the area planted with hardwoods has been increasing steadily since the mid-1990s and through the 2000s, although in recent years the expansion of the eucalypt plantation estate has dramatically slowed (Parsons *et al.*, 2006; Gavran & Parsons, 2010). Both the forestry industry and the general public are eager to understand how biodiversity is affected by the rapid expansion of the eucalypt plantation estate.

In recent years there have been several international reviews of biodiversity in forestry plantations (Hartley, 2002; Carnus *et al.*, 2006; Stephens & Wagner, 2007; Brockerhoff *et al.*, 2008; Felton *et al.*, 2010), as well as some focusing more specifically on Australia's eucalypt and pine plantations (Lindenmayer *et al.*, 2003; Lindenmayer & Hobbs, 2004; Lindenmayer & Hobbs, 2007). Munro *et al.* (2007) recently reviewed faunal use of revegetated areas (including farm forestry) in agricultural landscapes of Australia. Salt *et al.* (2004) produced a thorough review of biodiversity and tree plantings with guidelines for farmers and practitioners to maximise biodiversity values of plantings. The present review will:

- exclusively focus on industrial eucalypt plantations in southern Australia where most of the estate expansion has occurred (Parsons *et al.*, 2006)
- consider the full life-cycle and variety of plantation operations
- include land-use history and landscape context
- broadly consider all the different ways that plantations affect biodiversity
- synthesise recent research.

A review is needed because none of the reviews mentioned above have considered all of the five points listed, and in the last few years significant new research has been published on biodiversity-related issues relevant to Australian eucalypt plantations but has not yet been synthesised. Also, numerous perceptions are held by the general public concerning biodiversity and eucalypt plantations (e.g. 'plantations are biological deserts') that need to be tested by reviewing the scientific evidence.

The overall aim of this report is to review biodiversity outcomes (positive and negative) of eucalypt plantation expansion into agricultural landscapes of southern Australia. All published information has been reviewed, organised under themes and explored in the contexts of landscape and forestry operations. The broad nature of this topic and the variety of approaches taken in the literature make a traditional style of review more appropriate than a meta-analysis.

1B. What is biodiversity and why is it important?

Biodiversity can be described as the diversity and variety of living organisms arranged at the genetic, species and ecosystem levels (Noss, 1990). But biodiversity is more important than just the variety of microbes, insects, vertebrates and plants. The interactions and patterns that these organisms collectively form are critically important to maintaining ecological processes (Noss, 1990; Pimentel *et al.*, 1997; Hooper *et al.*, 2005). There is considerable global concern that biodiversity in tropical and temperate forests is being lost through the mass extinction of species at an unprecedented rate (Pimm, 1995).

Biodiversity can be measured in many different ways (Magurran, 1988). Simple measures of biodiversity such as the total number of species present in a given unit, however, may not be particularly informative. For example, keystone species (e.g. microbes that fix nitrogen) may make a unique contribution to particular ecosystem processes (Hooper *et al.*, 2005). The habitat association of species (e.g. old-growth forest or unforested pasture) is also important and this is likely to be driven by the need for specialist resources or environmental conditions (e.g. see Catterall *et al.*, 2004), but may not be represented in a simple biodiversity measure such as species richness. The social values attributed to species (e.g. koalas or rabbits) are also likely to differ depending on whether species are native or exotic, their degree of charisma, and their level of rarity or the degree to which they are endangered. Thus the identity of species and their relative abundance is the preferred way to measure biodiversity rather than simply the total number of species present. The many different ways that biodiversity is measured and reported in the literature (Magurran, 1988) creates challenges in the interpretation, comparison and synthesis of different studies.

There are several reasons why the plantation forestry industry should monitor and manage biodiversity. First, biodiversity is integral to the growth of plantation trees. For example, some species may be pests, while other species such as ectomycorrhizal fungi may be beneficial or essential. Second, there are increasing community expectations that biodiversity value be included in the management of forestry operations within plantation landscapes. Finally, consideration of the impact of forestry operations on biodiversity is a requirement of forest certification schemes such as the Forest Stewardship Council (FSC Australia, 2008) and the Australian Forestry Standard (AFS, 2007) for which many forestry companies already have received, or are pursuing accreditation.

1C. Forestry operations in eucalypt plantation landscapes

Most areas of plantation estates are devoted to eucalypt trees grown for fibre. But eucalypt plantations almost always contain areas of native vegetation within their estates or have native vegetation nearby (Barbour *et al.*, 2008; Greening Australia, 2008; Figs. 1, 2). These areas of native vegetation range from single old trees through to remnants of native vegetation of significant size and conservation value (Greening Australia, 2008). These remnants may include non-forest areas of native wetland, shrubland, sedgeland or grassland. The biodiversity of this native vegetation can significantly influence the biodiversity values of nearby or surrounding eucalypt plantations (Salt *et al.*, 2004). Because the native vegetation within plantation estates is managed by forestry companies, the biodiversity values of native vegetation and eucalypt plantations within the one landscape need to be assessed and managed concurrently.

Of the eucalypts grown in plantations in southern Australia, two species form the majority of plantings (81% at 2005; Parsons *et al.*, 2006)—blue gum or *Eucalyptus globulus* (Labill.), and shining gum or *E. nitens* (Deane and Maiden) . These two species of eucalypts have been transported well beyond their natural distribution (Potts *et al.*, 2003). Historically, *E. globulus* was distributed in Tasmania, the Bass Strait islands and southern Victoria, with an isolated population in South Australia, whereas now this species is used in plantations in Tasmania, Victoria, New South Wales, South Australia and Western Australia (Tibbits *et al.*, 1997; Vaillancourt *et al.*, 2001; Parsons *et al.*, 2006; Barbour *et al.*, 2008). *Eucalyptus nitens* originates from Victoria and NSW but is now used in plantations in higher rainfall areas of NSW, Victoria and Tasmania (Tibbits *et al.*, 1997; Barbour *et al.*, 2002; Parsons *et al.*, 2006). Thus, there are now areas of Australia where the predominant species of eucalypts in plantations are not native to the region.

Figure 1. Eucalypt plantation estates often include remnants of native vegetation. This example in a blue gum plantation from south-west Western Australia shows the close proximity of remnant and plantation eucalypt trees (Photo: Peter Grimbacher).

There are three phases in the rotation (life-cycle) of eucalypt plantations in southern Australia: establishment, growth and maintenance, and harvesting. Each of these phases takes a different amount of time to complete and has varying impacts on biodiversity. Overall impacts of the establishment phase of eucalypt plantations differ among regions in southern Australia depending on the previous land use. In the states of Western Australia, South Australia and Victoria, plantations tend to be established on ex-agricultural land that was used to graze sheep or cattle over the last 50 to 200 years. Much of this land was previously forest, but was cleared and replaced with native or introduced grasses during European settlement. The recent establishment of many eucalypt plantations on ex-agricultural land in effect re-converts these areas to tree-dominated ecosystems (Fig. 2). In Tasmania, about half of the monoculture plantations of *E. globulus* and *E. nitens* established over the last decade have been on pasture land, and the other half on areas previously supporting native forest (Forest Practices Authority, 2009a). In this context the net outcomes for biodiversity from plantation establishment in Tasmania are negative compared to plantation establishment in mainland Australia. Depending on the pre-existing land use, the plantation establishment phase may require physical clearing of vegetation or the application of herbicides to kill existing plants, occasional minor earthworks (ripping or mounding), followed by planting and fertilising tree seedlings (Dep. Ag. and CALM WA, 2005a; Jenkin & Tomkins, 2006; PIRSA, 2007). The process from land acquisition to tree planting may take up to several years.

The second phase of operations simply involves encouraging tree growth through minimal intervention. This may involve suppressing other competing plants by applying herbicide, pruning, thinning, maintaining fire breaks, controlling pests, and sometimes applying fertiliser (Jenkin & Tomkins, 2006; PIRSA, 2007). Competing plants such as grasses are also sometimes suppressed by grazing sheep or cattle for short periods once the trees are established. The intended final wood product (woodchips for pulp or sawlogs for timber), soil fertility, moisture availability and specific management activities will dictate whether this growth phase lasts for 10, 20 or more years (Anderson, 2003; Dep. Ag. and CALM WA, 2005a; Noble, 2009).

Harvesting completes the rotation; a process in which all the trees are cut and removed with heavy machinery, leaving residues or 'slash' of bark, branches, and leaves. In areas in which forest practice requirements have changed since the initial plantation establishment, more extensive habitat retention or restoration may be required, such as the addition of reserves around features of high natural or cultural value (e.g. riparian buffers or karst reserves).

Resprouting or coppicing from the stumps of harvested trees marks the start of the second rotation for *E. globulus* trees. The trees grow rapidly as they already have an established root system, and the many resprouting stems need to be pruned back to one main stem within the first year. Alternatively, if new tree genetics are incorporated, considerable site preparation may be needed to kill resprouting plantation trees and prepare the ground for new seedlings (Dep. Ag. and CALM WA, 2005a; Noble, 2009; PIRSA, 2010). *Eucalyptus nitens* on the other hand typically does not coppice and therefore control of coppice is not required for second-rotation establishment.

Figure 2. Aerial view of a blue gum plantation in south-west Western Australia, showing legacies of past land use. Reasonably large areas of remnant native vegetation can be seen (coarser textured patches) as can farm dams and creek lines (image from Google Earth).

2. Biodiversity in eucalypt plantations

2A. What should plantation biodiversity be compared against?

It is important that the biodiversity values of plantations are assessed in the context of alternative and previous land uses (Stephens & Wagner, 2007; Brockerhoff *et al.*, 2008; Bremer & Farley, 2010). For the landscapes of southern Australia, eucalypt plantations are expanding onto land that previously supported management-intensive treeless styles of agriculture, such as pastures used for livestock grazing. However, in certain parts of Australia (e.g. Tasmania) some native vegetation is cleared for plantation establishment (see Section 1C). Therefore assessments of eucalypt plantation biodiversity should include comparisons with the biodiversity in previous land uses (agriculture) in addition to the biodiversity of remnant eucalypt vegetation, which collectively provide a benchmark for the biodiversity values that could exist. Assessments of the biodiversity in eucalypt plantations typically include comparisons of relatively young plantations against 'old' remnant eucalypt vegetation. But such comparisons encompass many factors that influence biodiversity including the strong effects of stand age. Ideally, biodiversity assessments of eucalypt plantations should include contrasts with young native forest, such as revegetated sites, as well as old native forest. Most of the studies reviewed compared eucalypt plantation sites established on agricultural land with both agricultural land and remnant native forest. Some studies from Tasmania compared eucalypt plantation sites established on cleared native forest land with native forest alone.

How phylogenetically similar plantation tree species are to the native vegetation strongly influences biodiversity in plantations (Bremer & Farley, 2010). Plantation tree species that are more closely related are able to support more native species of animals and plants, and assemblages that are more similar to those found in native vegetation. In southern Australia, because species of eucalypts can be found in most native forests and woodlands, plantations of eucalypts (even of species not native to the area) will support at least some native animals, microbes and possibly plants. This has important implications for insect pests (see Section 4C) as it has been shown that the more similar plantation tree species are to the native vegetation, the greater the ability for insect pests to colonise plantations (Bertheau *et al.*, 2010). Because eucalypts are largely endemic to Australia, in other parts of the world eucalypt plantations are only ever likely to support a small subset of the indigenous biota (see Samways *et al.*, 1996; Barlow *et al.*, 2007; Larranaga *et al.*, 2009; Reino *et al.*, 2009; Zahn *et al.*, 2009; Proenca *et al.*, 2010). However, for this same reason we would expect eucalypt plantations in Australia to have higher biodiversity values and contribute to the maintenance of biodiversity at the landscape level.

Plantations can support a large number of insect, plant and vertebrate species (Brockerhoff *et al.*, 2008). The biodiversity values of individual plantations can vary considerably, and are influenced by several key factors in addition to phylogenetic similarity between plantation trees and native vegetation. These include structural complexity and spatial factors of the landscape (Lindenmayer & Hobbs, 2004; Salt *et al.*, 2004; Brockerhoff *et al.*, 2008) which are discussed in further detail below.

2B. Comparisons of biodiversity among eucalypt plantations, native forest and agriculture

Several studies have assessed the biodiversity values of eucalypt plantations in southern Australia, with some comparing the biodiversity of plantations to patches of remnant eucalypt bush and sometimes to other agricultural land uses. Various groups of flora and fauna in eucalypt plantations have been studied and compared using a variety of statistical techniques. An overall summary of the results of these studies are synthesised below and grouped according to taxonomic group.

Fungal communities in eucalypt plantations have been assessed in Western Australia (Lu *et al.*, 1999) and south-eastern Australia (Kasel *et al.*, 2008). Lu *et al.* (1999) found that ectomycorrhizal fungal diversity in *E. globulus* plantations established on agricultural land increased with plantation age, and although the number of fungal species in the oldest plantations was less than jarrah forests, a number of species found in the jarrah forests were also found in plantations. More recently, Kasel *et al.* (2008) compared soil fungal communities in *E. globulus* plantations to pasture and to native eucalypt forest. They found that the fungal communities in *E. globulus* plantations were more similar to native forest than to pasture in spite of the relatively recent conversion of pasture to eucalypt plantations. Both these studies suggest fairly rapid colonisation of eucalypt plantations by native fungal species.

Invertebrates constitute the majority of species (Hamilton *et al.*, 2010) and not surprisingly feature quite prominently in the literature concerning biodiversity in eucalypt plantations. A study of the invertebrates (velvet worms, snails, millipedes and carabid beetles) inhabiting *E. nitens* plantations converted from native forest in Tasmania found many forest species present in plantations but not as many as in native forests (Bonham *et al.*, 2002). Some forest species found in plantations were previously thought to be threatened by plantation establishment. However, the authors of this study noted that all the plantations with many forest species were either directly adjacent to native forest, the assumed source of the species colonising the plantations, or had biological legacies in the form of windrowed coarse woody debris from earlier native forest (see Section 2C). Another study in Tasmania compared beetle assemblages in remnant eucalypt forest with more extensive tracts of native eucalypt forest and with adjacent plantations converted from native forest (Grove & Yaxley, 2005). They found that small forest remnants sustained a significant number of forest species not always present in plantations. Overall, beetle assemblages in plantations were significantly different to those found in surrounding native forest.

Powell investigated beetle biodiversity in forest plantation landscapes within the Green Triangle region (Powell, 2010). She sampled beetle populations using pitfall traps in patches of remnant eucalypt vegetation surrounded by either pasture or *E. globulus* plantations established on ex-agricultural sites, as well as sampling beetles within pasture and within plantations. Beetle assemblages in *E. globulus* plantations were similar to beetle assemblages in native eucalypt vegetation but different from those in the pasture matrix; suggesting that the plantation matrix provides an extension of potential habitat.

Invertebrate studies on *E. globulus* plantations in south-west Western Australia have shown that eucalypt trees rapidly acquire a functionally diverse insect fauna of considerable biomass (Abbott *et al.*, 1999; Cunningham & Murray, 2007). However, the functional makeup (feeding ecology) of insect assemblages in plantations may differ from native eucalypt forest. For example, Cunningham and Murray (2007) studied beetles and found proportionally more herbivores and fewer decomposers in eucalypt plantations, compared to native eucalypt woodland.

A study comparing seven-year-old *E. globulus* plantations established on ex-agricultural sites with jarrah forests and pasture in south-west Western Australia found that plantations supported some of the biodiversity of soil and leaf litter mites, with some but not all forest species present (Adolphson & Kinnear, 2008). There were more mite species in native forests, intermediate numbers in plantations, and least in pasture. Overall, mite assemblages in plantations were more similar to pasture than to native forest, which is to be expected given the young age of the plantations studied and the low vagility of mites.

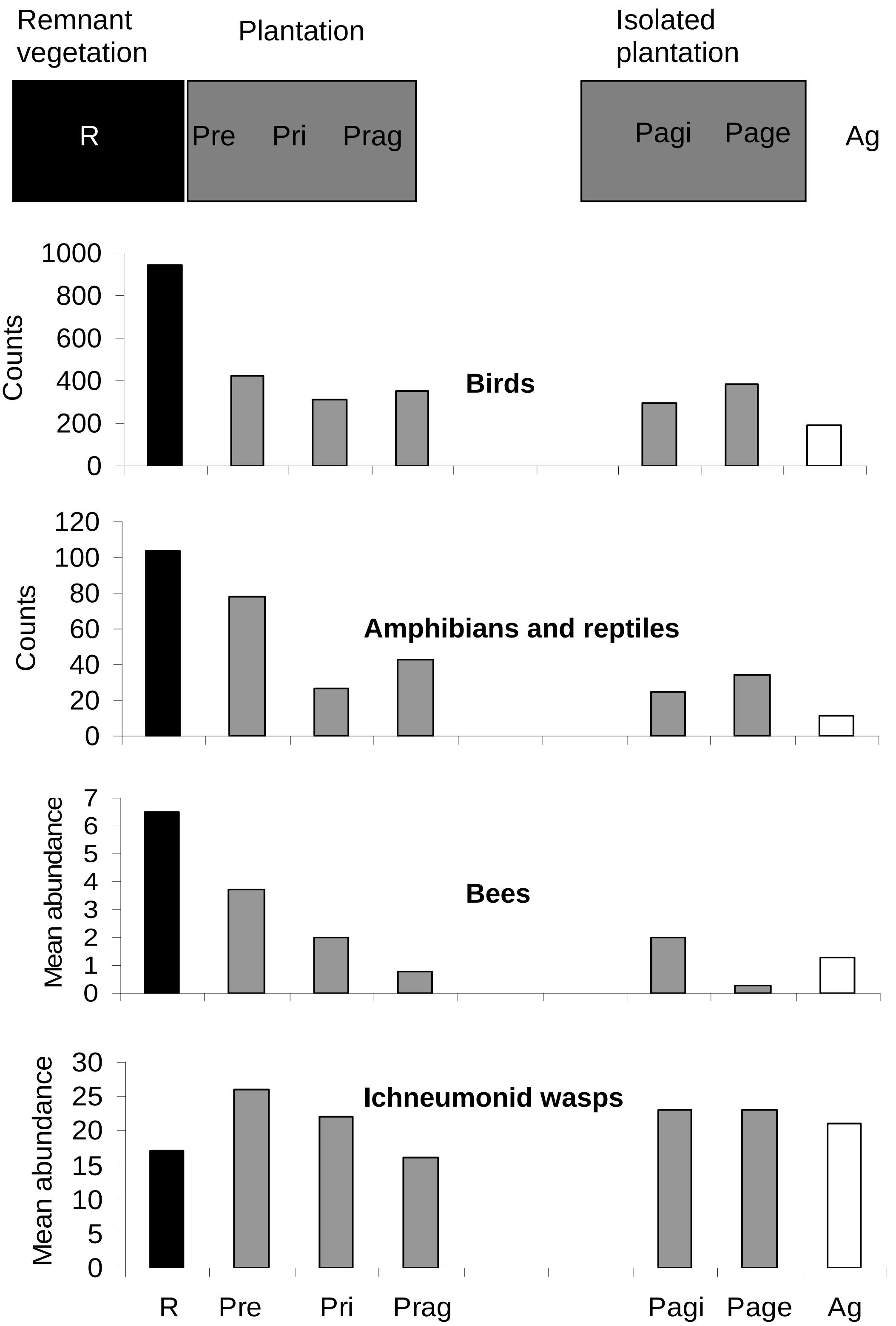
One of the most intensive studies on the insect biodiversity (assemblages of Coleoptera, Lepidoptera and Hymenoptera) of eucalypt plantations was conducted in south-west Western Australia (Cunningham *et al.*, 2005; see Box 1). The authors of this study compared insect biodiversity in *E. globulus* plantations established on ex-agricultural sites, against remnant native eucalypt woodland and pasture, and identified many forest species in plantations but concluded that insect fauna in plantations was distinct from native eucalypt forest and from pasture (Cunningham *et al.*, 2005). Insect assemblages in plantations were essentially a subset of the assemblages found in native forest but differed in that they had fewer species, had a few distinct species not found in native forest, had some species found in pasture, and tended to be dominated by a few abundant pest species.

The insect study of Cunningham *et al.* (2005) was based on the study design of Hobbs *et al.* (2003) who studied the vertebrate fauna at the same set of sites (see Box 1). Arboreal and ground-dwelling mammals (except for the western grey kangaroo) were largely absent from eucalypt plantations and agricultural land. There were slightly more amphibian and reptile species and individuals in eucalypt plantations than on agricultural land, but many more species and individuals were found in remnant eucalypt woodland. In general, vertebrate use of eucalypt plantations in south-west Western Australia was well below that found in remnant eucalypt woodland and only slightly more than that found in pasture.

Loyn *et al.* (2009) recently conducted a study of vertebrate use of *E. globulus* plantations established on ex-agricultural sites in the Green Triangle region of South Australia and Western Victoria. Vertebrates were surveyed in 13 cleared farmland sites alongside 22 *E. globulus* plantation sites and 30 native forest sites. With the exception of macropods, other mammals, particularly hollow-users, were largely absent from eucalypt plantations. A large multi-taxa survey was undertaken by Kavanagh *et al.* (2005) to establish faunal use of eucalypt plantings that had been planted for conservation purposes on ex-agricultural sites in northern Victoria and southern NSW. Although the plantings were not production plantations, the results are still relevant to this review. The authors compared vertebrate densities across eucalypt plantings (60 sites), remnant eucalypt vegetation (50 sites), and cleared agricultural land (10 sites). Vertebrates such as arboreal mammals, nocturnal birds and reptiles were rarely found among eucalypt plantings in comparison to patches of remnant vegetation (Kavanagh *et al.*, 2005). These results corroborate studies showing that reptiles and arboreal mammals are reluctant to use planted or regrowth vegetation compared to remnant eucalypt woodland (Cunningham *et al.*, 2007).

In addition to the frequent use of eucalypt plantations by several species of macropods, brushtail possums, cockatoos and parrots (Marks & Moore, 1998; Bulinski & McArthur, 2003), several mammals of significant conservation status have been reported to use eucalypt plantations, including the koala (Schlagloth *et al.*, 2008). Anecdotal reports of diggings and bi-catch from browsing control operations suggest that the Tasmanian bettong may make significant use of mid-rotation *E. nitens* plantations in north-eastern Tasmania (T. Wardlaw, pers. comm.). However, it is important to note that many of these vertebrates may not actually reside in eucalypt plantations but simply use these areas for foraging (see Section 2E).

Box 1. Faunal use of *E. globulus* plantations in south-west Western Australia



The experimental design (top panel) and selected results of fauna among remnant eucalypt vegetation, agriculture and *E. globulus* plantations established on ex-agricultural sites in different spatial configurations from the south-west of Western Australia (redrawn from Hobbs *et al.* (2003) and Cunningham *et al.* (2005)). In these studies, vertebrates and plants (Hobbs *et al.*, 2003) and selected insect groups (Cunningham *et al.*, 2005) were surveyed among four replicate sites of

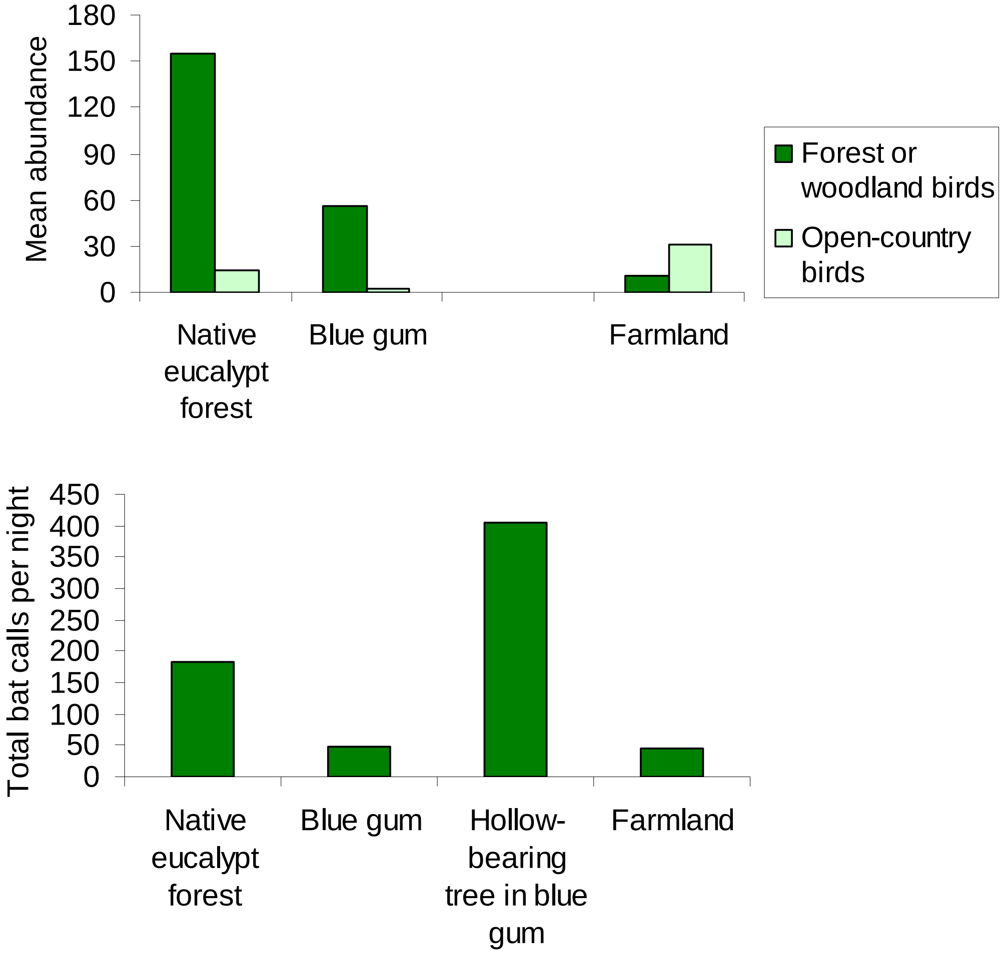
Box 1 (continued)

remnant eucalypt woodland (R), agriculture–pasture (Ag) and *E. globulus* plantations from edges adjacent to remnant vegetation (Pre), the interior of plantations adjacent to remnants (Pri), the agricultural edges of these plantations (Prag), as well as interior (Pagi) and edges (Page) of plantations isolated from remnant vegetation (refer to these papers for more details about the methodology). These two eucalypt plantation biodiversity studies are among the few in which different taxa were surveyed on the same set of sites. For each taxa, the unit of measurement is abundance or a proxy for individuals.

The results selected from these studies illustrate that for some taxa (amphibians and reptiles, birds and bees, but not ichneumonid wasps) the abundance of these animals was generally highest in remnant eucalypt vegetation, lowest in agriculture, and intermediate in *E. globulus* plantations. They also show that for this set of sites, some taxa (amphibians, reptiles and bees) showed 'spillover' from remnant vegetation to the adjacent plantation edge (but not plantation interiors), while other taxa (birds and ichneumonid wasps) appeared unaffected by proximity to remnant vegetation.

A few studies have surveyed bat assemblages among eucalypt plantations. Hobbs *et al.* (2003) surveyed bats among *E. globulus* plantations in south-west Western Australia and found that the species richness and density of bats was highest in remnant vegetation but much lower in eucalypt plantations and open farmland, which were fairly equivalent. Law and Chidel (2006) studied whether insectivorous bats used eucalypt plantings on the same ex-agricultural sites as studied by Kavanagh *et al.* (2005) in northern Victoria and southern NSW. All bat species were recorded in revegetated sites. Bat activity and species richness in revegetated sites was equal to that in agricultural sites, but only a third of the levels recorded in remnant vegetation. High tree density and the low height of trees and shrubs (a product of tree age) appeared to inhibit bat movement. These results suggest that only old or thinned eucalypt plantations would provide suitable foraging environments for bats. Loyn *et al.* (2009) also surveyed bat activity among *E. globulus* plantations established on ex-agricultural sites in the Green Triangle region of South Australia and Western Victoria. Bat activity and diversity was similar in farmland and eucalypt plantations but higher in native forest. Interestingly, bat activity (measured by calls) was even higher near hollow-bearing trees present in eucalypt plantations than in native forest (see Box 2).

Box 2. Faunal use of *E. globulus* plantations in the Green Triangle



Comparisons of bird (top panel) and bat (bottom panel) faunal use of blue gum plantations established on ex-agricultural sites, cleared farmland and remnant eucalypt vegetation in the Green Triangle region (redrawn from Loyn *et al.* (2009). Birds and bats were surveyed in multiple sites (see Loyn *et al.* (2009) for full details on methodology). The results of the bird survey reflect a general trend found in other studies that eucalypt plantations tend to support forest or woodland birds at the expense of open-country birds. For the bat survey, one bat detector was placed next to a retained hollow-bearing eucalypt in a blue gum plantation (Loyn *et al.*, 2009) in addition to the other site types. Although not replicated across multiple sites, this result illustrates the importance of biological legacies in supporting biodiversity in eucalypt plantations.

Use of eucalypt plantations by birds has been the focus of several studies. One of the most comprehensive studies of bird community use of eucalypt plantations was conducted by Loyn *et al.* (2007). They surveyed the birds at 25 agricultural sites, 58 sites in which eucalypt plantations had been established on ex-agricultural land and 22 native forest sites in two regions of rural Victoria (north-east and central Victoria). The density of forest and woodland birds was higher in eucalypt plantations than in cleared farmland, and only marginally lower than in native forest, although particular guilds of birds responded differently. Some guilds of birds were under-represented in eucalypt plantations relative to native forest (e.g. bark-foraging insectivores, nectarivores and predators), while others were well represented (e.g. insectivores that forage in the canopy and among tall shrubs). Open-ground foragers (including several species that have declined in woodlands) were more abundant in plantations than in native forest. Loyn *et al.* (2007) remarked that the 'density of forest birds in the plantations emerge as remarkably close to native forest, despite some important differences in species composition'. Nevertheless, the authors emphasised that many of the birds that were common in plantations had habitat requirements outside of plantations (e.g. hollow-bearing trees for nesting; see Section 2C). In a similar study conducted in the Green Triangle region of South Australia and Western Victoria (see Box 2), birds were surveyed in cleared farmland sites, *E. globulus* plantations and native forest sites (Loyn *et al.*, 2009). The authors found that forest or woodland birds tended to benefit from plantations at the expense of open-country birds (Loyn *et al.*, 2009).

In another study conducted in northern Victoria, Klomp and Grabham (2002) surveyed bird communities in *E. globulus* plantations established on ex-agricultural sites, pastures and nearby remnant eucalypt woodland. They found that the abundance and species richness of birds was lowest in pasture, intermediate in eucalypt plantations, and highest in remnant eucalypt vegetation. Kavanagh *et al.* (2007) reported the results of bird surveys on the same set of eucalypt planting sites as Kavanagh *et al.* (2005) in northern Victoria and southern NSW. Eucalypt plantings supported many forest-dependant and woodland-dependant species. Larger eucalypt plantings (>5 ha) tended to have more woodland bird species than smaller plantings, and this result implies that the large size of eucalypt plantations may lead to more positive biodiversity outcomes.

Working in south-west Western Australia, Hobbs *et al.* (2003) found that the abundance of birds in *E. globulus* plantations was between that in remnant eucalypt woodland (highest) and pasture (lowest; see Box 1). Species found in remnant woodland were also detected in eucalypt plantations but often at reduced densities, although a few bird species used eucalypt plantations to the same extent as remnant eucalypt woodland. Unlike Loyn *et al.* (2007), no functional groups stood out as contributing most to differences in bird use among eucalypt plantations, remnant eucalypt woodland and pasture.

Archibald *et al.* (2010b) surveyed birds in seven plantations of *E. globulus* established on ex-agricultural sites and seven native eucalypt remnants that were surrounded by these plantations in south-west Western Australia. Of 32 bird species found in the remnants, 19 were also detected in plantations.

The species richness or composition of plants in plantations has received less research attention than animals (but see Bremer & Farley, 2010). Munro *et al.* (2009) surveyed revegetated sites (including woodlots but not industrial plantations) in Gippsland, Victoria, for habitat structural features in addition to plant diversity. They found that habitat structure, but not native plant species richness, increased with site age. With the exception of weeds, native species were poor colonisers of revegetated sites. Hobbs *et al.* (2003) surveyed the vegetation in *E. globulus* plantations in south-west Western Australia. They also found that native species were very poor colonisers of

E. globulus plantations, with plantations on average supporting less than one native plant species other than *E. globulus* trees in 20×20 m quadrats, compared to more than seven species in native eucalypt vegetation, although it must be noted that agricultural sites had even fewer species of native plants than plantations. No doubt, the use of herbicides in eucalypt plantations severely restricted the ability of native plants to colonise plantations (see Section 4D). These local observations of plant biodiversity in plantations broadly concur with global patterns whereby conversions of agricultural land or degraded land into plantations favour native plant species at the expense of exotic plants (Bremer & Farley, 2010).

Collectively the studies conducted on microbes, animals and plants consistently show that agricultural land uses, such as pasture for livestock grazing, support fewer native species and different species assemblages than eucalypt plantations and native eucalypt vegetation. Consequently cleared land uses often have comparatively low biodiversity values. By contrast, the evidence reviewed supports the notion that eucalypt plantations support more forest species and different species assemblages than agricultural land uses. However, it is also evident that eucalypt plantations do not provide biodiversity values as high as remnant eucalypt vegetation and it is clear that eucalypt plantations are no substitute for native eucalypt vegetation. This conclusion concurs with Lindenmayer and Hobbs (2004) who stated that eucalypt plantations are unlikely to support the same number of forest species compared to patches of native vegetation.

Because the total number of studies quantifying biodiversity of eucalypt plantations in southern Australia is limited, few generalisations can be made. Nevertheless, it appears that the groups that were best represented in eucalypt plantations (birds and winged insects) are also good dispersers. By contrast, poor dispersers such as mammals, reptiles, amphibians and especially native plants were not as well represented in eucalypt plantations. But not all organisms conformed to this pattern. For example, bats are good dispersers but were poor colonisers of plantations. It has been suggested that plantations favour species that are good colonisers and that are generalists (Pawson *et al.*, 2008). Some but not all of the taxa using eucalypt plantations in southern Australia fit this generalisation. For birds, forest species are likely to be favoured over open-habitat species (Lindenmayer *et al.*, 2008a; Loyn *et al.*, 2009). It should be remembered that the ability of eucalypt plantations to support different taxonomic groups will vary, and even closely related species may respond differently. Furthermore, several other factors will influence the biodiversity values of eucalypt plantations, and these are discussed below (see Sections 2C–D).

2C. How does habitat structure influence the biodiversity in plantations?

Habitat structure and complexity have been shown to have profound effects on biodiversity in plantations, with more complex habitat structure supporting more forest species (Lindenmayer & Hobbs, 2004; Salt *et al.*, 2004; Carnus *et al.*, 2006). Similarly, it is thought that plantations with older trees support more species and more forest specialists because they have particular habitat features such as hollows that only develop after many decades (Loyn *et al.*, 2008; Vesk *et al.*, 2008). Consequently, plantations will never be able to support all forest species when plantation rotations are short, as they are for pulpwood. Intensive management strategies will produce plantations of monocultures with simplified structural complexity and are likely to result in plantations that are only able to support a subset of the species found in native vegetation. However, such structurally simple sites may be preferred by specific taxonomic groups. For example, Law and Chidel (2006)found that the density and species richness of insectivorous bats among replanted eucalypt sites was highest in sites that were the most open, without an understorey.

It has been suggested that management activities that provide resources such as woody debris and leaf litter (through pruning), may encourage particular faunal groups such as invertebrates (Bashford, 1990; Bonham *et al.*, 2002) that use those resources. The potential ability for plantation managers to manipulate structural diversity and improve biodiversity values of plantations has been extensively discussed in the literature (see Hartley, 2002; Lindenmayer & Hobbs, 2004; Salt *et al.*, 2004; Brockerhoff *et al.*, 2008). Management options to improve the biodiversity values of plantations may create interesting production trade-offs (Salt *et al.*, 2004; Kanowski *et al.*, 2005).

Recently it has been demonstrated that within-species variation in genetics, expressed through morphological variation and plant chemistry, strongly influences the biodiversity associated with *E. globulus* trees (Barbour *et al.*, 2009a; Barbour *et al.*, 2009b; Barbour *et al.*, 2009c; Barbour *et al.*, 2009d). Variations in bark volume and roughness, and in leaf litter and soil properties under trees, alter invertebrate assemblages (Barbour *et al.*, 2009a; Barbour *et al.*, 2009b). However, this within-species variation in the biodiversity associated with plant parts was not observed for macro-fungi associated with the logs of trees from these provenances (Barbour *et al.*, 2009d). These results provide plantation managers with additional ways to promote biodiversity, although this may result in further production trade-offs if trees are selected that divert growth to non-harvestable plant parts such as bark.

The absence or presence of particular guilds of animals in eucalypt plantations is by inference evidence of the importance of structure and resources. Birds that feed on nectar are poorly represented in eucalypt plantations presumably because there are few eucalypt trees of sufficient maturity to provide the necessary quantity of flowers needed to sustain specialist nectar feeders (Loyn *et al.*, 2009). Likewise Loyn *et al.* (2008) found that treecreepers were absent from plantations, most likely because these birds forage on rough-barked eucalypts. Current plantations of *E. globulus* and *E. nitens* tend to be dominated by smooth-barked trees sourced from certain provenances. However, some provenances of both species are rough-barked (Wardlaw, 1999; Barbour *et al.*, 2009b) and if they were more widely used in plantations they could presumably support specialist birds associated with bark. By contrast, canopy-foraging insectivorous birds are almost as common in eucalypt plantations as native forest probably because there is ample foliage in plantations for these birds to forage through (Loyn *et al.*, 2009). Whole groups of animals such as reptiles and arboreal marsupials may be missing or rare in planted or regrowth vegetation because of a simplified structure and the absence of key resources such as large trees and woody debris (Cunningham *et al.*, 2007).

Historical legacies from a previous land use can strongly influence the species found in ecosystems following disturbances such as the establishment of plantations (Franklin *et al.*, 2000; Salt *et al.*, 2004; Cawsey & Freudenberger, 2008). Sometimes during eucalypt plantation establishment structural features such as stumps and logs (from previous native vegetation), mature living trees (from paddocks; Fig. 3) or even rocks and boulders can be retained and embedded within plantations. These biological legacies and physical features often have particular species still associated with them that are otherwise absent from plantations. For example, invertebrates of significant conservation status detected in Tasmanian eucalypt plantations by Bonham *et al.* (2002) were likely to be present due to the legacies of old logs from previous native forest, piled together in windrows. There are also cases where plantation establishment may not destroy the soil seedbank or rootstock resulting in the survival of non-plantation plants in plantations (Kanowski *et al.*, 2003). In another example Pharo and Lindenmayer (2009) found that bryophyte diversity in pine plantations was equivalent to that of native eucalypt vegetation because of the high number of moss species surviving on logs within plantations where the logs pre-dated plantation establishment.

Biological legacies in the form of retained hollow-bearing trees are particularly important for birds and bats in eucalypt plantations in South Australia, Victoria and southern NSW (Grabham *et al.*, 2002; Kavanagh *et al.*, 2007; Loyn *et al.*, 2007; Loyn *et al.*, 2008; Loyn *et al.*, 2009; see Box 2, Fig. 3). Koch *et al.* (2009) also showed that remnant mature eucalypt trees embedded within eucalypt plantations in Tasmania provide important resources (nesting hollows) for birds, which are likely to improve the overall biodiversity values of eucalypt plantations established on ex-agricultural land. Tree hollows are a key resource used by more than 300 species of native Australian animals (Gibbons & Lindenmayer, 2002). However, the long-term viability of these biological legacies and their associated biodiversity is questionable given that they are not always being renewed (Manning *et al.*, 2006; Fischer *et al.*, 2009). Effective restoration of remnants that results in the regeneration of these biological legacies (see Section 3B) may address some of these concerns.

2D. Are the biodiversity values within plantations improved when they are near patches of native vegetation?

Many of the species that colonise plantations are assumed to originate from remnant eucalypt vegetation. Spatial barriers to the colonisation of plantations may exist if species are poor dispersers or if native vegetation (a source of colonising species) is not nearby. The amount of native vegetation remaining in the landscape may also influence the colonisation of plantations if probability of dispersal is linked to population size. Dispersal theory suggests that plantations adjacent to existing native vegetation may have minimal spatial barriers to colonisation, resulting in more species associated with native vegetation colonising plantations, and thereby higher biodiversity values. Two studies conducted in south-west Western Australia using the same site network of remnant native vegetation, pasture sites and *E. globulus* plantations that varied in their proximity to native vegetation, have tested whether this was the case for vertebrates, plants and insect groups (Hobbs *et al.*, 2003; Cunningham *et al.*, 2005; see Box 1). Although plantation edges nearest remnants were used the most by vertebrates, species showed many different response patterns and faunal use of plantation edges and interiors was not consistent (Hobbs *et al.*, 2003). Invertebrates showed only weak differences in assemblage composition when plantations were isolated or adjacent to remnant vegetation or pasture (Cunningham *et al.*, 2005). Both studies found only minor support that eucalypt plantations near existing remnant vegetation had higher biodiversity values.

Insectivorous bats also appear to be insensitive to proximity and amount of remnant vegetation in the landscape, presumably because they travel long distances and use different habitat types (Law & Chidel, 2006). However, a related study of birds from the same set of sites did show significant responses to landscape patterns of distance from remnant forest with fewer bird species in more isolated patches (Kavanagh *et al.*, 2007). It is possible that factors other than proximity to remnant vegetation, such as habitat structure and suitability, are more important in determining the biodiversity values of plantings (Grimbacher & Catterall, 2007).

Figure 3. Old retained trees such as this one contain hollows which are used by many organisms for nesting. Likewise the coarse wood debris in the foreground provides shelter for many organisms. These features do not have the time to develop in trees grown in eucalypt plantations. Biodiversity values of plantation landscapes are heavily reliant on these biological legacies. This tree was part of a shining gum plantation in northern Tasmania (Photo: Peter Grimbacher).

2E. Faunal ‘spillover’ into plantations from adjacent native vegetation and partial use of plantations by species

When interpreting the results of biodiversity surveys in eucalypt plantations, it is important to distinguish whether species detected in eucalypt plantations at a certain point in time are permanent residents, transitory migrants, or only partially use plantations as habitat. The occasional use of plantations may occur if there are resources for foraging but not refuge. Also, some species in eucalypt plantations may only be present because they are being subsidised by ‘spillover’ from adjacent remnant eucalypt vegetation. Such situations are effectively source–sink population dynamics (Dias, 1996). However, determining source and sink habitats requires long-term demographic studies that identify species’ needs as well as dispersal patterns. No studies have been conducted in eucalypt plantations that meet these criteria. However, it is clear that faunal interactions between remnant eucalypt vegetation and eucalypt plantations do occur (Strauss, 2001).

Some species detected in plantations may only be present because native vegetation is nearby. For example, the koala is known to use eucalypt plantations and feed on young blue gum trees when plantations are adjacent to remnant native vegetation (Schlagloth *et al.*, 2008). As koalas strongly prefer large eucalypt trees, it is unlikely that they use young eucalypt plantations as primary habitat. Rather, Schlagloth *et al.* (2008) speculate that koalas use blue gums as an extension of their range. Other vertebrates also use plantations in a similar manner. In Tasmania, Bennett’s wallaby, the Tasmanian pademelon and the common brushtail possum are thought to reside in adjacent native forest and use plantations for browsing, sometimes causing major tree damage (Bulinski & McArthur, 2003). Several species of birds are also thought to forage in eucalypt plantations but are unable to nest or breed in eucalypt plantation trees because the young trees lack suitable nesting features such as hollows (Loyn *et al.*, 2008). Consequently, these species may use mature eucalypt trees in nearby native vegetation remnants or even single trees in paddocks. These examples highlight why whole landscapes need to be considered when assessing the biodiversity values of plantations.

Considerable ecological information exists on vertebrates, allowing researchers to interpret the partial use of eucalypt plantations. However, for many organisms such as invertebrates, this information is unknown. Nevertheless, several invertebrate studies have shown a ‘spillover’ from native vegetation into eucalypt plantations. For the biodiversity study in *E. globulus* plantations in Western Australia described above (Cunningham *et al.*, 2005), invertebrates showed only weak differences in assemblage composition at edges or interiors of plantations adjacent to native vegetation (see Box 1). In Brazil, Zanuncio *et al.* (1998) showed that eucalypt plantations embedded within a matrix of native vegetation had greater diversity but lower densities of Lepidoptera, possibly caused by greater densities of natural predators spilling over from native vegetation. Lower psyllid densities and increased Hymenoptera predator densities have been observed at plantation edges with native vegetation in Brazil (Silva *et al.*, 2010). In another study in Brazil, Braganca *et al.* (1998) also showed that Lepidoptera densities increased in eucalypt plantations moving away from the edge with native vegetation. By contrast, densities of natural predators showed the opposite pattern. The predator ‘spillover’ effect appeared to occur up to 200 m from edges with native vegetation while the highest Lepidoptera densities occurred further into the plantation at distances of 400–600 m and possibly greater (Braganca *et al.*, 1998).

Powell set out to document edge effects across plantation edges with remnant vegetation and pasture and to determine how far edge effects operate in plantation landscapes (Powell, 2010). Working in the Green Triangle region, she sampled beetles with pitfall traps across remnant vegetation edges, going 100 m from edges into adjacent land uses of plantations and pasture. But the differences in beetle assemblages (measured at the taxonomic level of family) between plantations and remnant vegetation were not significant and consequently showed no edge effect.

3. Biodiversity in remnants of native vegetation: effects of replacing a pasture matrix with a eucalypt plantation matrix

3A. Consequences of reducing grazing pressure

Prior to European settlement of southern Australia, in areas now occupied by commercial eucalypt plantations, landscapes were clothed by open eucalypt woodland or forest. After settlement, large areas of native trees and shrubs were cleared and replaced with crops or grasses for pasture. Land clearing has been especially widespread in southern Australia. States with the greatest clearing of native vegetation cover (including forest and non-forest) are as follows: Victoria (63%), Western Australia (54%), South Australia (36%), New South Wales (33%) and Tasmania (20%) (Cofinas & Creighton, 2001). Open forests and especially eucalypt woodlands are the major vegetation groups most heavily cleared (DEWR, 2007). The effect of land clearing on biodiversity has been strongly negative because, not only has the overall extent of native vegetation been greatly reduced, but much of the remaining vegetation has been heavily fragmented, existing as small remnants surrounded by agricultural land. Over time, the ecological integrity of many of these vegetation remnants, particularly the smaller ones, has degraded through a combination of edge effects, nutrient enrichment from fertiliser drift and livestock use of remnants for grazing or shelter, the removal of coarse woody debris from the forest floor, the establishment of weeds, altered fire regimes, and interactions among these factors and others such as drought (Landsberg *et al.*, 1990; Yates & Hobbs, 1997; Hobbs, 2001; Jurskis, 2005; Davidson *et al.*, 2007; Close *et al.*, 2008; Duncan *et al.*, 2008; Prober & Smith, 2009; Lindenmayer *et al.*, 2010). Climate change may also contribute to tree decline in remnants (Davidson *et al.*, 2007; Sanger *et al.*, In press). Several degradation processes have thus led to the poor condition of many native vegetation remnants, often with dead, dying or stressed overstorey trees, and a lack of tree regeneration (Hobbs, 2001; Davidson *et al.*, 2007; Close *et al.*, 2008; Prober & Smith, 2009). Therefore, it is important to recognise that much of the remnant native vegetation that now abuts eucalypt plantations or is embedded within plantation estates has a complex history of disturbance and degradation that pre-dates plantation establishment.

So what are the effects on the biodiversity within native vegetation patches of replacing adjacent pasture with eucalypt plantations? Native vegetation remnants may become embedded within newly established eucalypt plantations and form part of eucalypt plantation estates managed by forestry companies (Figs. 1, 2). One likely outcome is that grazing pressure from livestock in remnant vegetation will be significantly reduced, although some grazing from the agistment of sheep and cattle for grass suppression under plantation trees, and from native macropods and other marsupials may continue. Stock grazing by sheep and cattle is particularly deleterious to the overall health of remnant vegetation because stock cause soil compaction, nutrient enrichment (through dung and urine), and introduce weed species such as pasture grasses and herbs (Close & Davidson, 2004; Davidson *et al.*, 2007; Close *et al.*, 2008; Duncan *et al.*, 2008). Elevated leaf nutrition causes problems for eucalypt trees in remnants because it predisposes trees to attack by phytophagus insects (Landsberg *et al.*, 1990). Horton has found that, in Tasmania, elevated nutrition also reduces mycorrhizal fungal biodiversity (Horton, 2011), a result that concurs with other studies (Harvest *et al.*, 2008). Elevated leaf nutrition thus decouples eucalypts from root symbionts vital in the process of uptake of water and nutrients (Close *et al.*, 2008).

The effects of plantation establishment may be similar to that of fencing native vegetation to exclude livestock, and this is likely to result in positive biodiversity outcomes (Fig. 4). Numerous studies in Western Australia (Prober & Smith, 2009), south-eastern Australia (Dorrough *et al.*, 2004; Lunt *et al.*, 2007; Briggs *et al.*, 2008) and Tasmania (Close *et al.*, 2008) have shown that reducing livestock grazing intensity leads to increases in native plant cover, plant recruitment and regeneration. Grazing intensity also affects other organisms such as invertebrates (Abensperg-Traun *et al.*, 1996; Bromham *et al.*, 1999; Lindsay & Cunningham, 2009), birds (Jansen & Robertson, 2001; Martin & McIntye, 2007) and reptiles (James, 2003) and these organisms are also likely to show positive responses to the significant reduction in grazing pressure in remnants brought about by eucalypt plantation establishment. However, not all elements of plant communities respond to livestock removal and several studies have shown that often weed cover and diversity does not reduce (Briggs *et al.*, 2008; Prober & Smith, 2009). Weeds are likely to persist in native vegetation remnants long after the cessation of livestock grazing.

Several projects have investigated the condition of remnant vegetation in plantation landscapes and how this affects biodiversity. Archibald *et al.* (2011) examined key ecosystem features and processes relating to soil chemistry, decomposition, and native tree health and regeneration within small forest remnants (1 to 4 ha) embedded within *E. globulus* plantations in south-west Western Australia. Soil nutrient enrichment was significantly associated with the scale of vegetation modification and increased in order from: 1) intact forest, 2) remnants with native understoreys, 3) remnants with weedy understoreys, 4) plantations and 5) pasture. Archibald and colleagues suspect that in this region, most remnants with understorey vegetation dominated by weeds will remain in a degraded state even after a significant reduction in stock grazing pressure, once eucalypt plantations are established in the landscape. This may be due, in part, to more rapid rates of nutrient turnover sustaining higher nutrient availability in the soil following the replacement of ligneous understorey plants (native) with annual ones (weeds)(see Prober *et al.*, 2002). To compound the nutrient problems in remnants, there was less canopy seed set and little or no seedling establishment in remnants with weedy understoreys compared to remnants with native understoreys (Archibald *et al.,* 2011). The development of a weedy understorey in remnants also has some influence on other organisms. For example Archibald *et al.* (2010b) found that bird communities in remnants with weedy understorey were significantly different from those in remnants without weeds.

Powell's work on beetle assemblages in forest plantation landscapes within the Green Triangle region (Powell, 2010) has shown that beetle assemblages respond to the quality of vegetation condition among remnant vegetation patches. O'Dwyer investigated how fragmentation and degradation of grey box grassy woodlands affects insect assemblages (O'Dwyer, 2010). She found that the number of insect species (sampled by sweep netting and pitfall trapping) in remnant vegetation was strongly correlated with the condition of that vegetation, with more species found in higher quality remnants. The effect of remnant condition on insect biodiversity was stronger than the effect of remnant size. There was also a strongly positive relationship between the time since livestock was excluded from remnant vegetation and the number of insect species sampled.

Figure 4. Remnant native vegetation adjacent to pasture (top panel) may be subjected to grazing and trampling from livestock as well as increased wind speeds and solar radiation at edges. Remnants adjacent to eucalypt plantations (bottom panel) will likely receive less grazing pressure, and the adjacent plantation trees will reduce edge effects by providing shading and reducing windspeeds. Eucalypt plantations abutting remnant eucalypt vegetation may also buffer changes in the fauna at edges (see Section 3D). These examples of edges are from the Green Triangle region (Photos: Peter Grimbacher).

3B. Restoring remnant vegetation within eucalypt plantation estates

Some of the native vegetation remnants now embedded within eucalypt plantation estates are badly degraded, with evidence of overstorey eucalypt dieback and death, and few signs of plant recruitment and regeneration despite the reduction in grazing pressure (Davidson *et al.*, 2007; Archibald & Hardy, 2009). However, the conservation and restoration of these remnants is important because research in Australia has shown that even small remnants of eucalypt vegetation can support high densities and species of native invertebrates (Abensperg-Traun & Smith, 1999; Davies *et al.*, 2001; Driscoll & Weir, 2005; Grove & Yaxley, 2005; Baker *et al.*, 2009), vertebrates (MacDonald & Kirkpatrick, 2003; van der Ree & Bennett, 2003; Brown *et al.*, 2008; Lindenmayer *et al.*, 2008a) and plants (Prober & Thiele, 1995; Ross *et al.*, 2002; Sutton & Morgan, 2009). Because some species do not use plantations as habitat, the remnant native vegetation in plantation landscapes plays an important role in maintaining biodiversity at a landscape-level

(Brockerhoff *et al.*, 2008). Even single remnant trees have significant conservation value (Manning *et al.*, 2006; Koch *et al.*, 2009; Fischer *et al.*, 2010) and for some woodland communities these single trees form a significant portion of remaining habitat (Gibbons & Boak, 2002). Thus by managing biological legacies (see Section 2C) including small remnants of native vegetation within larger plantation estates, the forestry industry can play an important role in conservation at the landscape level.

Including degraded remnant vegetation within eucalypt plantation estates may facilitate the rehabilitation of these remnants. As many impacts of grazing on plant communities are long lasting (Lunt *et al.*, 2007), possible biodiversity benefits brought by plantation establishment and reduction in grazing pressure may take some time to appear. This particularly applies to regeneration and recruitment of overstorey eucalypt trees. Management intervention may even be needed to rehabilitate remnant vegetation beyond simply removing the disturbing agents or introducing plant propagules via direct seeding or the planting of seedlings (Yates & Hobbs, 1997). Our understanding of the effectiveness of techniques to restore degraded remnants have greatly advanced since the first attempts to synthesise this topic were made by Yates and Hobbs (1997). The causes of tree decline, and hence ways to reverse tree decline, are also much better understood. For example, the importance of fire in driving tree decline, and the possible use of fire to restore vegetation health, is now apparent. Close *et al.* (2009) propose that a reduction in the frequency of fire in remnants of eucalypt vegetation since European settlement has allowed midstorey vegetation (including weeds) to flourish and leaf litter to accumulate. These structural elements effectively ‘lock up’ nutrients and intercept rainfall, preventing water from infiltrating the soil and reaching eucalypt tree roots. Studies of the relationship between fire regime and forest health conducted in dry forests (Tuart, *E. gomphacephala*) in Western Australia and wet/dry forest (gum-topped stringy bark, *E. delegatensis*) in north-eastern Tasmania showed that the long absence of fire-altered soil processes and understorey composition in a way that caused tree decline in particular forest types on susceptible sites (Harvest *et al.*, 2008; Close *et al.*, 2009; Close *et al.*, Submitted a, b).

The sclerophyllous vegetation of many parts of Australia is well adapted to fire with this disturbance agent triggering widespread re-sprouting and re-seeding. Species of eucalypts are particularly well known for mass germination events following fire. Therefore, it comes as no surprise that many restoration trials are incorporating the use of fire (Archibald, 2009; Archibald *et al.*, 2010a; Close *et al.*, 2010; Ruthrof *et al.*, 2010; Close *et al.*, Submitted a, b). It is noteworthy that re-introducing fire into eucalypt plantation estates is at odds with current management strategies of fire suppression. Other techniques are being trialled, such as understorey thinning or weeding using herbicides, the addition of mulch, understorey nurse plantings, protecting seedlings with guards, irrigation, ground ripping and fertilising (Graham *et al.*, 2009; Archibald *et al.*, 2010a; Close *et al.*, 2010; Ruthrof *et al.*, 2010). In South Australia, natural resource managers have taken a different approach to restoration and are building habitat corridors to link patches of remnant vegetation embedded within plantations, specifically to facilitate the movement of threatened species such as bandicoots (PIRSA, 2011). What is clear from these studies is that the most appropriate restoration techniques are likely to be specific to each local area. This local variation has been incorporated into a ‘Tree Decline Toolkit’ devised by Davidson and Goodwin (2010). This computer model allows land managers to identify the current state of a forest, assisted by more than 100 photographs and notes depicting how native forest remnants are affected by each of the 12 driving ecological factors associated with tree decline. The forest is then given a score out of one hundred and is put into one of six health categories. Managers can then test a number of management options and see predicted outcomes of decisions.

PhD student Tanya Bailey has been studying eucalypt seedling recruitment in native dry forest remnants in the Tasmanian midlands (Fig. 5).[1] She found that for several years following fire, 80% of seedling regeneration was adjacent to large logs. The micro-site for seedling regeneration was described and compared to average conditions on the forest floor (Bailey, Submitted). Seedling micro-sites had more ash, more charcoal, greater protection from wind and mammalian browsers, had no competing weed species (because hot fire had cooked the soil seed bank), but most importantly the surface soil was not hydrophobic (Bailey, Submitted). Most eucalypt forest soils are hydrophobic because waxes and oils leached from eucalypt leaves, and hydrocarbons leached from eucalypt litter are coated onto colloids at the soil surface causing water droplets to sit on the soil surface rather than penetrate the soil. Fire that occurs near logs and within piles of coarse woody debris is sufficiently hot to heat soils to a depth of several centimetres, removing hydrophobicity and allowing water to collect in soils at this point, providing a store of moisture for seedling growth. This water source is not available elsewhere on the forest floor. Therefore the combination of fire and coarse woody debris is essential for forest regeneration and can be mimicked to improve regeneration in degraded forest systems (Bailey, Submitted). Katarzyna Bialkowski is also conducting research into remnant restoration and soil properties but she is focusing on the ecological processes mediated by microbial communities.[2] She has set up field trials in *E. globulus* plantations near Albany, Western Australia to investigate what happens to microbial activities in soil when different treatments designed to restore remnant vegetation are trialled. Preliminary results suggest that herbicides, but not mulching, significantly improve the condition of the soil (Bialkowski, 2010).

3C. What influence does the biodiversity in plantations have on the biodiversity in native vegetation?

The land use between patches of remnant vegetation can affect the biodiversity residing within remnant vegetation, and this influence can be even stronger than the effects of patch area and isolation (Prugh *et al.*, 2008). Unfortunately there are no published examples of such effects for eucalypt plantation landscapes in Australia. The most relevant examples in the literature are from newly established pine plantations and pasture grassland matrices in landscapes in southern NSW containing remnant eucalypt vegetation. Over many years, Lindenmayer *et al.* (2008a) have documented the faunal changes occurring in eucalypt remnants and the surrounding matrix. One of the key results they report is that the replacement of a grassland matrix with pine forest favours forest species and habitat generalists at the expense of open-country or woodland species. This can lead to unusual combinations of species that these researchers have termed ‘novel assemblages’ (Lindenmayer *et al.*, 2008b). Species that did well in the pine matrix tended to increase in abundance in eucalypt fragments, a result also demonstrated for beetles in a different landscape but also in southern NSW (Davies *et al.*, 2001). One would expect to see similar ecological changes in eucalypt plantation landscapes.

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[1] Tanya Bailey is completing her PhD through the University of Tasmania. Her CRC for Forestry student profile can be accessed at http://www.crcforestry.com.au/education-outreach/students/index.html.

[2] Katarzyna Bialkowski is completing her PhD through Murdoch University. Her CRC for Forestry student profile can be accessed at http://www.crcforestry.com.au/education-outreach/students/index.html.

Figure 5. Experiments are underway in different parts of Australia on ways to restore degraded eucalypt remnants. Working in the Tasmanian midlands, PhD student Tanya Bailey has found that 95% of eucalypt seedlings regenerated in full or partial ashbeds and 80% of seedlings were within a metre of coarse woody debris (T. Bailey, pers. comm.). Further work on soil water repellency after fire has revealed that in recently burnt sites, soil that has been burnt intensely where fuel load was high (i.e. around coarse woody debris) is very hydrophilic while neighbouring soil that has been less intensely burnt is extremely hydrophobic. Thus, the top soil layer forms a wettable reservoir of moisture for newly germinated seedlings (Photos: Tanya Bailey).

PhD student Mayumi Knight[3] is conducting research into how bird and bat biodiversity in patches of remnant vegetation in the Green Triangle are affected by differing matrices (cleared farmland, eucalypt plantations and pine plantations). Preliminary results from the first year of data suggest that birds that like to forage in *E. globulus* plantations (e.g. spotted pardalotes) have increased abundances in remnants surrounded by blue gums compared to remnants surrounded by pasture and pine. By contrast, open-country species (e.g. welcome swallows) are not found in remnants surrounded by blue gums but are found in remnants surrounded by pasture or pine (Knight *et al.*, 2010).

[3] Mayumi Knight is completing her PhD through the University of Melbourne. Her CRC for Forestry student profile can be accessed at http://www.crcforestry.com.au/education-outreach/students/index.html.

3D. Do plantations ameliorate edge effects at boundaries with native vegetation?

The recent replacement of pasture with eucalypt plantations has not only increased the overall amount of forest cover in landscapes (Fig. 2), but it has altered the edge dynamics for some remnants of native vegetation (Fig. 4). In some areas, native vegetation that was previously abutting pasture now abuts eucalypt tree plantations. Some small remnants have even had their entire edges altered so that they are now completely embedded within plantations. Biophysical and biological interactions across remnant edges have therefore changed because of the establishment of plantations. At a landscape level, there are likely to have been alterations to ecological processes involving water, nutrients and fire. These changes are likely to influence plant physiology and biodiversity within native forest remnants.

Abrupt edges where forests abut open habitats such as pasture typically result in higher solar radiation, variation in temperature, vapour pressure deficit, and wind speed at edges (Murcia, 1995; Denyer *et al.*, 2006). A large body of literature demonstrates the biological responses and mechanisms of edge effects on a range of different organisms (Ries *et al.*, 2004). Because the strength of microclimatic gradients across edges varies depending on the structure of the edge (Didham & Lawton, 1999), it is thought that plantations may mediate the negative effects of abrupt edges because the physical structure of plantation trees is more similar to native vegetation than pasture or crops. Recently Denyer *et al.* (2006) demonstrated this phenomenon and showed that pine plantations in New Zealand can mediate microclimate gradients at edges with native forest. It should be noted that eucalypt plantations would not be expected to mediate microclimate edge effects for plantations adjacent to treeless native vegetation, such as grasslands or shrublands.

Replacing pasture edges with eucalypt plantations may alter the proportion of species that prefer interior or edge conditions. MacHunter *et al.* (2006) observed that the abundance of an edge-loving bird (the noisy miner) in eucalypt remnants in Gippsland, Victoria reduced over time after eucalypt plantations were established nearby. By contrast, the abundance of miners in other remnants that were not adjacent to plantations did not reduce. The noisy minor has a disproportionately large deleterious influence on other bird species due to its hyper-aggressive nature (Grey *et al.*, 1997; Piper & Catterall, 2003). For one of the remnants adjacent to eucalypt plantations, the reduction in the density of miners was accompanied by an increase in the number of other forest bird species (MacHunter *et al.*, 2006). While there were too few sites to draw robust conclusions regarding the consequences of plantation establishment, the results nevertheless suggest that plantations may buffer some biotic edge effects in remnants and may also slow down faunal losses in remnant vegetation caused by habitat fragmentation. These faunal losses often gradually occur many years after the initial loss of habitat. Importantly, MacHunter *et al.*(2006) found that those species most likely to disappear from remnants were rare species which tend to have high conservation values.

Loyn *et al.* (2010) recently compared bird assemblages in remnants of native vegetation surrounded by different matrices (cleared farmland, eucalypt plantations and pine plantations) in the Green Triangle region. The authors hypothesised that plantations would buffer edges of remnants from the edge effects found across abrupt remnant–pasture edges. They found that the abundance of forest birds was slightly greater in remnants surrounded by pine plantations compared to remnants surrounded by *E. globulus* plantations or pasture. The pine plantations were older than the *E. globulus* plantations meaning that perhaps not enough time had elapsed since eucalypt plantation establishment for positive effects to be observed. Loyn *et al.* (2010) also suggested the absence of a positive effect among remnants surrounded by eucalypt plantations may be because the remnants studied were in relatively good condition and above a degradation size threshold (10ha), and the landscape has a low density of noisy miners, who drive out forest birds (Piper & Catterall, 2003) leading to faunal degradation. PhD student Mayumi Knight is investigating this research question in the same region but in more detail. Preliminary results from her first year of sampling have found slightly stronger differences than Loyn *et al.*'s (2010) study. Remnants surrounded by blue gums tended to have greater abundances of forest birds but fewer open-country birds compared to remnants surrounded by pasture. Remnants surrounded by pine plantations were somewhat intermediate. Importantly, there were significantly fewer noisy miners in remnants embedded in *E. globulus* plantations, compared to pasture, with remnants embedded within pine plantations intermediate. These preliminary results suggest that blue gum plantations are buffering edge effects across remnant vegetation boundaries (Knight *et al.*, 2010).

There has been little direct research into the physical edge effects of eucalypt plantation establishment in Australia except for a recent study by Wright (2010). Wright studied microclimate and plant physiology on the north-facing edges of three remnant vegetation sites adjacent to *E. globulus* plantations, and three remnant vegetation sites adjacent to pasture, in the Dergholm region of south-west Victoria. Remnant vegetation at pasture edges was exposed to greater wind velocities and solar radiation than plantation edges (Wright *et al.*, 2010). It appears that eucalypt plantations adjacent to native vegetation have the effect of reducing wind velocities and provide shading, lowering temperatures and moisture stress during winter. However, in summer, this does not have the effect of decreasing moisture stress because the reduced wind speeds are thought to lower the amount of thermal energy that is removed through air movement (Wright *et al.*, 2010). Consequently, this increases temperatures in native vegetation adjacent to plantations during summer, and may actually increase moisture stress at this time.

Photosynthesis and stomatal conductance were also measured on the dominant tree (*Eucalyptus arenacea*) and shrub species (*Leptospermum myrsinoides*) in remnants (Wright, 2010). During winter, most physiological variables for these two species were lower at plantation edges, probably due to shading by plantation trees. In summer there was no difference in any of these variables between pasture or plantation edges for the shrub. However, the tree had reduced photosynthesis and stomatal conductance, suggesting competition for water with adjacent plantation trees (see Section 4F). Thus it appears that microclimate patterns across edges are complicated and as yet, no simple generalisations for edges of open-woodland ecosystems exist. The research by Wright (2010) shows that eucalypt plantations can have both positive and negative influences on the microclimate of adjacent native vegetation. It should be noted that the design of this study, especially focusing on north-facing edges in the 650 mm precipitation zone, maximised the potential water competition between plantation trees and adjacent native vegetation. Despite this, the results from the two plant species studied did not suggest that vegetation condition within remnants was affected.

Powell investigated beetle biodiversity in forest plantation landscapes within the Green Triangle region (2010). She sampled beetles in patches of remnant vegetation surrounded by either pasture or newly established *E. globulus* plantations. Beetles were captured using pitfall traps, and were identified to family level. The richness of beetle families in remnants of native vegetation surrounded by pasture decreased from the centre of the remnant towards the pasture–remnant edge, suggesting a negative edge-effect on biodiversity from the surrounding pasture that penetrated about 30 to 40 m into remnant vegetation patches. By contrast, the richness of beetles in remnants of native vegetation surrounded by plantations did not decrease towards the remnant–plantation edge. These findings suggest that recent plantation establishment within the Green Triangle may mitigate some of the impacts of habitat fragmentation, by providing a buffering effect for beetle communities living in patches of remnant vegetation.

3E. Do plantations facilitate the dispersal of species between native fragments?

Previous clearing of native vegetation in southern Australia has resulted in the fragmentation of habitat with remaining native vegetation existing as patches in multiple-use agricultural and eucalypt plantation landscapes. The spatial breaking apart of habitat into disjunct patches separated in space is one of the effects of habitat fragmentation and is different from habitat loss (Fahrig, 2003). This effect of fragmentation creates spatial barriers to the movement of genes and species across landscapes. The intervening land use between habitat patches (termed the matrix) varies in the degree to which it can provide habitat for species (e.g. Driscoll, 2008). But even if the matrix does not provide habitat for species, it may facilitate the dispersal of species (typically animals) between patches of remnant vegetation (Ricketts, 2001) or it may reduce movement of some forest species between fragments. The ability of the matrix to facilitate dispersal (often termed permeability) varies depending on the physical structure of the vegetation. A recent review by Prevedello and Vieira (2010a) found that the matrix is most successful at facilitating movement of organisms when it is structurally similar to remnant vegetation. From these results, it would be expected that forestry plantations have greater potential to facilitate the dispersal of species between remnant patches of eucalypt vegetation than all other agricultural land-use types and there is at least some evidence from other countries to support this expectation (Brockerhoff *et al.*, 2008). However, dispersal is species specific, so eucalypt plantation establishment may not improve the ability of all species to disperse between patches of native vegetation.

In southern Australia, there has been limited research into how different matrix types (land uses) influence the dispersal of organisms between patches of native vegetation. The koala is known to use *E. globulus* plantations adjacent to native vegetation and some have speculated that eucalypt plantations may be able to facilitate the dispersal of this species between patches of native vegetation (Schlagloth *et al.*, 2008). Law and Chidel (2002) also speculated that eucalypt plantation establishment may increase the dispersal of ground mammals between fragments of native vegetation. Several studies of Australian marsupials have shown that they can disperse across matrices of exotic pine plantations even when those plantations do not provide habitat (Banks *et al.*, 2005; Taylor *et al.*, 2007; Marchesan & Carthew, 2008). It has also been shown that marsupials dispersing through plantations initially use the tree rows to guide direction and facilitate movement (Prevedello & Vieira, 2010b; Fig. 6). However, replacing a pasture matrix with trees may not benefit all faunal groups. For example, Law and Chidel (2006) found that insectivorous bats avoided young trees in eucalypt plantings. A study on the effect of pine plantations on beetles inhabiting remnant patches of native vegetation in southern NSW showed that plantation establishment can even reduce dispersal between habitat patches (Davies *et al.*, 2001). Thus, the overall biodiversity benefits that eucalypt plantations might provide by facilitating the dispersal of species between remnants of native vegetation are likely to be small. However, this interpretation needs to be verified experimentally because, to date, no studies in Australia have actually tried to measure dispersal through eucalypt plantation matrices.

4. Potential negative effects on biodiversity caused by eucalypt plantation establishment

4A. Can plantation trees cross-pollinate with local eucalypt trees?

The planting of *E. globulus* and *E. nitens* outside of their natural distribution, and the establishment of plantations either adjacent to native vegetation or surrounding native remnants (see Section 1C; Figs. 1, 2) has the potential to place plantation eucalypts in contact with local species of eucalypts or with races of the same species with which there has been no prior association. Pollen flow out of plantations into neighbouring native vegetation may lead to the creation of hybrids and the introgression of genes that could compromise the genetic integrity of local eucalypt populations, especially for rare or highly fragmented species (Barbour *et al.*, 2010). This process is a potential threat to the biodiversity values of adjacent native forest (Strauss, 2001).

The risks associated with pollen-mediated gene flow out of eucalypt plantations have been comprehensively reviewed by Potts *et al.* (2003). Potts *et al.* (2003) note that in many cases the risk of such gene flow is small owing to strong barriers to hybridisation between distantly related species, differing flowering times, and poor hybrid fitness. Nevertheless, in some cases the risks are real as hybrids between local and plantation species have been observed. For example, *E. globulus* has been recorded naturally hybridising with 15 different eucalypt species (Barbour *et al.*, 2008). It is worth emphasising that hybridisation only occurs between closely related species within the genus *Eucalyptus*, and there is no risk of hybridisation between species from different eucalypt subgenera. Within a subgenus the chances of hybridisation are thought to increase with reduced taxonomic distance between the species. Therefore, one of the factors likely to determine the ability of plantation eucalypt species to hybridise depends on how taxonomically similar they are to indigenous species. A potential barrier to hybridisation is that flowering time varies greatly between plantation species and local native species, and also within species from different regions. The geographic distance between plantation trees and indigenous eucalypt trees is a barrier to hybridisation as most pollen appears to be transported only several hundred metres from a source (Barbour *et al.*, 2005; Mimura *et al.*, 2009), although rare, long-distance dispersal no doubt occurs (Southerton *et al.*, 2004). Local eucalypt trees adjacent to plantations or embedded within a plantation are at the highest risk (see Section 1C; Figs. 1, 2). Although hybridisation has been detected at distances of up to 1.6 km from plantation boundaries, most hybridisation occurs within 200 m (Barbour *et al.*, 2005).

Figure 6. Although the specific role that eucalypt plantations may provide in facilitating faunal movement between remnants of native vegetation is unknown, recent research overseas suggests that the orientation of tree rows in plantations may play a role (Prevedello & Vieira, 2010b). Note the lack of an understorey and the simplified structure in this blue gum plantation established on pasture, from the Green Triangle region (Photo: Peter Grimbacher).

So, what are the risks of genetic contamination of native eucalypts from eucalypt plantations for the plantation species *E. globulus* and *E. nitens*? In a recent study, Barbour *et al.* (2010) assessed the likelihood of pollen dispersal from exotic eucalypt plantations into all of Australia's rare native eucalypts. They conducted spatial analyses quantifying the proximity of eucalypt plantations and rare eucalypt species. For the 74 rare eucalypt species, 22 species had eucalypt plantations within 10 km and, of these, eight were within 1 km. In a separate study, Barbour *et al.* (2008) surveyed 300 *E. globulus* plantations across southern Australia in order to assess the risk of pollen-mediated gene flow from plantations into indigenous eucalypt populations. They found that few trees flowered in plantations grown for pulpwood (range of 0–20% of trees). This low incidence of flowering agrees with observations that plantations lack nectar-feeding birds (Loyn *et al.*, 2007). Barbour *et al.* (2008) also found that 52% of the plantations were not in the vicinity of eucalypt species that were compatible for hybridisation with *E. globulus*. Nevertheless, there were some important differences in the compatibility of eucalypt species between the major regions where *E. globulus* plantations are currently grown. In Tasmania only 16% of *E. globulus* plantations surveyed had compatible species nearby; for Western Australia the figure was 37%; and it was 39% in Gippsland. In the

Green Triangle, the proportion of plantations with compatible natives growing adjacent was even higher at 72%. The authors concluded that 65% of reproductive-age plantations presented minimal or low risk of hybridisation. In ten high-risk situations, open-pollinated seed from adjacent eucalypts was collected. Hybrid seeds were found at three sites (hybridising with *E. camaldulensis* and *E. ovata*), and hybrid seedlings were found near only one of the 300 plantations examined. *Eucalyptus nitens* has also been found to hybridise with neighbouring *E. ovata* in Tasmania (Barbour *et al.*, 2002; Barbour *et al.*, 2005). Even if hybrids are able to form and germinate, there is evidence of significant selection against them in the wild (Barbour *et al.*, 2006a). In Tasmania, trials measuring seedling survival of *E. ovata* × *E. nitens* hybrids showed that in field conditions they had higher levels of mortality, poorer plant health and reduced height, although it is uncertain whether this reduced survival continues as the plants age (Barbour *et al.*, 2006a).

Barbour *et al.* (2008) have created a framework for managing the risk of gene flow from *E. globulus* plantations, and this would be equally applicable to *E. nitens* plantations. The first step of this framework requires a pre-planting risk assessment to determine the spatial proximity of plantations to native species, assess the crossability with indigenous eucalypt species and assess the conservation status of adjacent forest (Potts *et al.*, 2003). This may identify major risks in which case planting should not proceed or hybridisation levels need to be closely monitored (Barbour *et al.*, 2010). For situations where there is a lower risk, several monitoring steps and management actions are suggested. During plantation establishment, planting non-flowering edge rows could be one option (Barbour *et al.*, 2008). After establishment of the plantation, the risks can be assessed through monitoring plantation flower abundance (Barbour *et al.*, 2008), plantation and native flowering synchrony (Barbour *et al.*, 2006b), levels of hybridisation in native seed (Barbour *et al.*, 2008), and fire events which could lead to seedling establishment (Barbour *et al.*, 2006a). Even if hybrids occur in open-pollinated seed, they may not survive in competition with natives. High-risk sites may be monitored several years after a recruitment event (Barbour *et al.*, 2006a, 2007; Barbour *et al.*, 2008) and weeded if necessary. This research has now been included into risk assessment guidelines in Tasmania (Forest Practices Authority, 2009b).

4B. Do plantation trees invade adjacent land?

There are numerous examples of tree species used in forestry plantations that have colonised areas outside plantation boundaries (Richardson, 1998). These 'escapees' (often called wildlings) can have considerable ecological impacts on the land they invade especially if the adjacent land use is native vegetation of high conservation value. Many of the traits that make tree species attractive to forestry (e.g. fast growth) are the same traits that make them potential weeds (Richardson, 1998).

On the whole, eucalypts are not very successful invaders despite their ability to produce large amounts of seed (Richardson, 1998). Forsyth *et al.* (2004) found that despite 149 different eucalypt species having been introduced to South Africa for a variety of reasons and uses, including plantations, only two species (*E. camaldulensis* and *E. grandis*) have become problematic invaders. However, there are examples of *E. globulus* escapees from plantations in Chile (Becerra, 2006), California (Kirkpatrick, 1977), Hawaii (Wagner *et al.*, 1999) and New Zealand (Webb *et al.*, 1988). *Eucalyptus globulus* has also naturalised in Portugal and Spain (and probably other countries in Europe and South America) since plants were first introduced during the 1800s. The overall distance of dispersal and establishment for *E. globulus* is low (Kirkpatrick, 1977), and consequently an assessment of the overall weed risk for this species in California was only moderate (Warner, 2004).

In Australia, cases of wildlings of *E. globulus* or *E. nitens* colonising land adjacent to eucalypt plantations are probably more common than the literature suggests, although it is has been argued that wildlings are less of a problem than pollen-mediated gene flow (Barbour *et al.*, 2003, 2006a; Barbour *et al.*, 2008). For example, one study in Tasmania found *E. nitens* wildlings up to 30 m from plantations whereas hybrids were found up to 310 m from plantations (Barbour *et al.*, 2003). However, the movement of forestry machinery may be able to move seed much greater distances (Barbour *et al.*, 2006a) but this would not be into native forest.

Few formal surveys have quantified how common wildings are in Australian eucalypt plantation landscapes. Preliminary surveys of *E. globulus* plantations in Tasmania and Victoria found wildlings present in 16 out of 47 plantations (Matthew Larcombe, pers. comm.). Across four sites where the frequency was quantified, 191 *E. globulus* wildling seedlings were counted along 3.8 km of plantation edge and these were within 20 m of the plantation edge. It appears that two key conditions are needed for wildling establishment: 1) plantations of sufficient age so that trees are reproductively mature and can produce seed, and 2) a recruitment trigger (e.g. disturbance events such as fire, road construction, or rainfall).

There are several reasons why wildlings from eucalypt plantations may not be a significant problem in Australia. First, and most importantly, eucalypt wildlings from plantations may have poor survival in adjacent native forest (Barbour *et al.*, 2006a) even when favourable disturbance conditions (e.g. fire) are present (Barbour *et al.*, 2008). Second, short rotation times mean that plantation trees have relatively few years in which they can produce significant amounts of flowers and seed before they are harvested. Third, seeds do not travel far. Fourth, relatively few problems of wildlings have been reported in landscapes in which *E. globulus* has been established for many years (see above). By contrast, wildlings from pine plantations are much more of a problem in Australia (Williams & Wardle, 2007) and elsewhere in the Southern Hemisphere (Richardson, 1998).

4C. Do plantations harbour pest species?

Plantations can provide habitat for a significant number of species (see Section 2B). Some of these species will be associated with forested habitats and some with more open habitats characteristic of pastures. Some of the species found in plantations may also be considered weeds, pests or ferals (Lindenmayer & Hobbs, 2004; Kanowski *et al.*, 2005; Brockerhoff *et al.*, 2008). Species may be defined as ‘pests’ because they have negative effects on biodiversity values, are exotic, elicit a negative functional effect on plantation trees (e.g. through herbivory), or replace other species and thereby simplify communities. Defining species as ‘pests’ is rather subjective and context-dependant, and consequently species may be viewed as a pest in one habitat type but not another.

Numerous exotic and native vertebrate ‘pests’ have been documented within eucalypt plantations in southern Australia. The exotic vertebrates include rats, the house mouse, rabbits, cats and foxes (Borsboom *et al.*, 2002; Bulinski & McArthur, 2003; Hobbs *et al.*, 2003; Loyn *et al.*, 2009). There is little evidence of exotic birds in eucalypt plantations. Loyn *et al.* (2007; 2009) found that introduced birds were scarce in eucalypt plantations and native forest, but were more common in pasture. A number of herbivorous native vertebrates causing tree damage in eucalypt plantations have also been recorded, and they include several species of macropods, brushtail possums, cockatoos and parrots (Marks & Moore, 1998; Bulinski & McArthur, 2003; Dep. Ag. and CALM WA, 2005b), although it appears that many of these species reside in adjacent native forest and simply visit plantations to forage (Bulinski & McArthur, 2003). Overall densities of ‘pest’ vertebrates in eucalypt plantations appear to be relatively low, although their effects on tree health can be severe but localised (Bulinski & McArthur, 2003).

Few exotic invertebrates have been recorded in eucalypt plantations with the exception of an introduced snail (Bonham *et al.*, 2002). But there are several examples of insect pests being moved within Australia and introduced to states where they are not native. Western Australia in particular has seen the introduction of the eucalyptus weevil (*Gonipterus*) and the leaf blister sawfly from eastern states (Mayo *et al.*, 1997). There are 85 pest insect species that have been recorded in eucalypt plantations in Australia (Strauss, 2001), although no doubt since the time of that study more pest species have emerged but have not necessarily been formally documented. The high number of insect species present in plantations probably reflects the fact that native eucalypt vegetation already contains a diverse insect fauna that feeds on eucalypts (Majer *et al.*, 2000), and with phylogenetically similar plants used in plantations, the ability for insects to shift hosts is high (Bertheau *et al.*, 2010). For example, Radho-Toly *et al.* (2001) found that in Western Australia, rates of herbivory on two exotic species of eucalypt species originating from the east coast of Australia was comparable to two locally indigenous eucalypt species. Consequently, some have suggested that the increasing size of the eucalypt plantation estate in Australia will lead to increasing pest problems (Loch & Floyd, 2001; Strauss, 2001). However, results from a study by Grimbacher and colleagues has shown that although the number of insect taxa in *E. globulus* and *E. nitens* plantations across southern Australia is indeed increasing, the magnitude or severity of insect pest outbreaks is not (Grimbacher *et al.*, in press).

Generally, the intensive management and use of herbicides to suppress competition results in few plant pests or weeds in eucalypt plantations in southern Australia (Jenkin & Tomkins, 2006). The dense canopy provided by eucalypt trees is also not conducive to understorey plant growth. In spite of this, Hobbs *et al.* (2003) documented that plantations of blue gums in Western Australia had more species of weeds and a greater abundance of weeds than remnant eucalypt forest, but that this was much less than in agricultural land (pasture). On average, plantations supported 6.4 weed species per 20 × 20 m quadrat. Munro *et al.* (2009) surveyed revegetated sites (including woodlots but not industrial plantations) in Gippsland, Victoria and found that in woodlots (the most similar site type to industrial plantations) weed species richness and cover increased with site age. However, it is possible that this result is specific to woodlots with their relatively small size and high proportion of edge and may not apply to industrial plantations that are larger than the plots surveyed and probably have a denser canopy.

Despite suggestions that eucalypt plantations may act as reservoirs or sources of pest species (Strauss, 2001), there is only one alleged example of eucalypt plantations being the source of an insect pest and creating an ecological bridge to colonise native vegetation. It has been suggested that in Western Australia the eucalyptus weevil *Gonipterus platensis* was accidently introduced to the state from the east coast with *E. globulus* seedlings. It is thought that after colonising plantations, the weevils colonised adjacent native vegetation (Loch & Floyd, 2001; Cunningham *et al.*, 2005). However, even this example has recently been questioned by forest entomologists because no evidence of egg-laying or larval feeding by *G. platensis* has been observed on native eucalypt species (M. Matsuki; pers. comm.). There is no evidence of vertebrate pests moving out of plantations into agricultural areas or native forests.

Several pest species have been documented moving into plantations from adjacent habitats and land uses (see Section 2E). The adjacency effects of vertebrate browsers (especially possums) at the edges between plantation and native forest are well documented (see Munks & McArthur, 2000; Bulinski & McArthur, 2003). Several insect pests are known to move from agricultural areas into plantations. These include the wingless grasshopper which is actively targeted by pesticide spraying at the edges between plantations and agricultural land (Abbott *et al.*, 1999). Another example is a species of spring beetles (*Heteronyx*) whose larvae feed on the roots of grasses found in agricultural areas or degraded native vegetation remnants (Abbott *et al.*, 1999). Overwhelmingly more evidence exists of agricultural and native forest areas being sources of pests in plantations rather than plantations acting as reservoirs of pests.

It is also perceived that eucalypt plantations have higher levels of predation than native eucalypt forest and thus may act as population sinks for fauna moving out of remnant eucalypt vegetation. However, this perception is not supported by the literature. Working in Victoria and South Australia, Loyn *et al.* (2007; 2008; 2009) found that carnivorous birds were less common in eucalypt plantations compared to remnant eucalypt vegetation. Hobbs *et al.* (2003) found that foxes and cats were more abundant in remnant eucalypt vegetation than eucalypt plantations in south-west Western Australia. In pine plantations of south-eastern Australia, the biomass of both native and exotic vertebrate predators was estimated to be similar to that of native eucalypt forest or agricultural land (Gill & Williams, 1996). Further evidence at odds with the perception that predation is higher in eucalypt plantations comes from invertebrates, where natural predators of insect pests are normally more abundant in remnant vegetation than in plantations (see Section 2E).

4D. How do herbicides and pesticides affect biodiversity in plantations?

Contrary to public perception, the plantation forest industry is a very minor consumer of herbicides and pesticides, when compared with other agricultural crops, and is estimated to use only 0.7% of all herbicides and pesticides consumed in Australia (Jenkin & Tomkins, 2006). Within the Australian plantation forest industry, the proportion of herbicides used relative to pesticides (mostly insecticides) is about 99 to one (Jenkin & Tomkins, 2006). At the level of individual plantations, the application of herbicides and pesticides can vary from 100% of the plantation down to only 30% (Jenkin & Tomkins, 2006). On the whole, most plantations will not be sprayed with insecticides during a complete rotation. For example, on average, only about 1% of Forestry Tasmania's eucalypt plantation estate is aerially sprayed with insecticide each year (Forestry Tasmania, 2010).

Herbicides are primarily used to kill and suppress plants other than the desired tree species, otherwise it is argued that they will compete with plantation trees for water, nutrients and space (Guynn *et al.*, 2004; Jenkin & Tomkins, 2006). Indeed, the gains in plantation productivity proffered by weed control can be very significant (Guynn *et al.*, 2004). Poor weed control can lead to the complete failure of a plantation. A typical management strategy in plantations is to apply herbicides (knockdown and residual) immediately before planting. A follow-up application of knockdown herbicide may be warranted in the first two years of plantation establishment. Once canopy closure is achieved, weeds are suppressed by the trees. Animals, especially insects and some browsing mammals (see Section 4C) may also need to be controlled to protect plantation trees from defoliation. Pesticides are applied only in response to pest outbreaks (Elliott & Hodgson, 2004). The most economical application of herbicides and pesticides to eucalypt plantations is aerially with fixed-wing aircraft or helicopters (Jenkin & Tomkins, 2006). Although spray drift from aerial spraying onto adjacent land uses including remnant native vegetation is possible, tight regulations and guidelines limit the likely incidence of such events (Elliott & Hodgson, 2004; Jenkin & Tomkins, 2006). Currently there are no estimates of how often spray drift may occur and whether it is a significant biodiversity issue for terrestrial systems. However, it is recognised that spray drift poses a greater risk to aquatic ecosystems downstream and assessments have been made of the frequency of events and impacts on biodiversity (see Section 4E).

Despite the relatively low use of herbicides and pesticides in eucalypt plantations by the forestry industry, these chemicals do have negative impacts on biodiversity. Suppressing plant growth other than that of the desired tree species through the application of herbicides prevents the development of an understorey or shrub layer (Guynn *et al.*, 2004). This not only reduces habitat complexity, making conditions less favourable for animal and microbial biodiversity (see Section 2C; Fig. 6), but it can also have negative consequences for natural pest control. This is because multi-species plantings, or those that contain flowering plants other than plantation trees, provide important resources for natural predators that suppress pest populations (Steinbauer *et al.*, 2006; Gamez-Virues *et al.*, 2009). It is unlikely that the use of herbicides in the eucalypt plantation industry will reduce significantly in the foreseeable future, hence plant diversity and structural complexity within eucalypt plantations is also likely to remain low.

A study by Loch (2005) conducted in Western Australia showed that spraying plantations reduces populations of natural predators and other benign arthropods. It is suspected that those arthropods that are protected in microhabitat refugia (i.e. within plant tissues, under bark and under leaf litter) or through alternative life stages (e.g. eggs) are most likely to immediately survive spraying events (Loch, 2005). In any case, populations of insect that do not survive spraying are likely to rebound after 10 to 12 months (Loch, 2005). A Tasmanian study found much more ephemeral effects of spraying with broad-spectrum insecticides compared to the results of Loch (2005). Elek *et al.* (2004) found that plantations sprayed with alpha-cypermethrin had a dramatic reduction in populations of natural enemies and other non-target invertebrate species compared with unsprayed eucalypt plantations or plantations sprayed with the 'softer insecticide' spinosad. Invertebrate populations within the plantations sprayed with alpha-cypermethrin returned to non-sprayed levels within only a month or so.

Given that most eucalypt plantations will not be sprayed with insecticides during a complete rotation (Forestry Tasmania, 2010), and the potentially rapid recovery of arthropod populations (Elek *et al.*, 2004; Loch, 2005), the overall negative consequences of insecticide spraying on the biodiversity in eucalypt plantations is likely to be relatively small. The need for FSC certification is likely to reduce the use of insecticides even further. Effective monitoring of pest and disease outbreaks and application of biosecurity measures will limit the spread of pests and pathogens and ultimately reduce the amount of pesticides used. Management strategies that incorporate better pest-resistant genetics into plantation trees, provide structures for overwintering of natural enemies, replace broad-spectrum insecticides with 'softer' pesticides and deploy 'trap trees' (Elek & Wardlaw, 2010) are also likely to generate outcomes that are more positive for biodiversity.

4E. Water quality and eucalypt plantations

Generally, the quality of water draining from forests is much higher than that draining from other land uses, especially agriculture (Binkley *et al.*, 1999; van Dijk & Keenan, 2007). Nevertheless, it has been suggested that the quality of water originating from, or travelling through, eucalypt plantations may adversely impact on aquatic biodiversity because surface runoff from eucalypt plantations may contain insecticides, herbicides, fertilisers and sediments (Stewart, 2011).

The mobilisation of sediments leads to increased stream turbidity which reduces water quality and negatively affects aquatic biodiversity (Campbell & Doeg, 1989). It is thought that forestry plantations are generally beneficial in reducing the volume of sediment and nutrients transported into river systems compared to agricultural land uses (van Dijk & Keenan, 2007). A study conducted in Western Australia comparing grass and *E. globulus* riparian buffers found that both vegetation types were capable of intercepting sediments and nutrients although grass was more efficient (McKergow *et al.*, 2006a; 2006b). It was thought that the absence of an understorey in the eucalypt plantation combined with water repellency of the soil led to reduced water infiltration and higher surface flow, reducing the capacity for sediment and nutrient trapping (McKergow *et al.*, 2006a; 2006b).

It has been shown that roads, rather than the actual harvesting of trees, are the main source of sediments mobilised by forestry operations (Croke *et al.*, 1999). Neary *et al.* (2010) compared turbidity along a stream in Tasmania and found that *E. nitens* plantations were a minor source of sediment relative to roads, a dam accessible to cattle and a cultivated paddock. Unsealed forest roads can facilitate the movement of significant amounts of sediments into streams which can reduce water quality (Lane & Sheridan, 2002; Sheridan & Noske, 2007b). However, at a whole-catchment scale, total amounts of mobilised sediments originating from forest roads may be small (Sheridan & Noske, 2007a).

Possibly the greatest threat to aquatic biodiversity from eucalypt plantation management comes from the use of insecticides (this also applies to other forms of forestry; Neary *et al.*, 1993). This is because insecticides used in plantations may find their way into streams either through spray drift or surface runoff associated with storm events, although riparian buffer strips are very effective in reducing the mobility of pesticides (Neary *et al.*, 1993). One insecticide commonly used in eucalypt plantations is alpha-cypermethrin (Jenkin & Tomkins, 2006) and this pyrethroid is more toxic to fish and aquatic invertebrates than to terrestrial invertebrates (Solomon *et al.*, 2001; Hartnik *et al.*, 2008). However, if this insecticide is used responsibly in eucalypt plantations, it is likely to have minimal impact on aquatic biodiversity. For example, Elliot and Hodgson (2004) reviewed all water testing results conducted by Forestry Tasmania for the period 1993–2003 and found that none of the insecticides used by Forestry Tasmania during that time (including alpha-cypermethrin) were ever detected in streams. This is a noteworthy result given that it is a requirement that all applications of herbicides, insecticides and fertilisers on state plantations in Tasmania require the testing of water before a spray event, immediately after spraying, and after the first significant rains (Elliott & Hodgson, 2004; Forestry Tasmania, 2009). This means that hundreds of water testing events take place annually. Although no insecticides were detected in water samples between 1993 and 2003, residues from some herbicide and fertiliser applications to eucalypt plantations have been detectable in waterways (Elliott & Hodgson, 2004). Of 4396 tests conducted between 1993 and 2003, three samples were in excess of NHMRC values and 87 exceeded Forestry Tasmania guidelines with 49 of these attributed to herbicides (atrazine and simazine), not used by Forestry Tasmania since the mid 1990s. During the 2008–09 spray seasons, Forestry Tasmania conducted 196 tests, with zero positive results (Forestry Tasmania, 2009).

A large retrospective study of aquatic macroinvertebrate health and catchment land use in Tasmania was recently completed as part of a CRC for Forestry supported project (unpublished data, Peter Davies, pers. comm.; Smith *et al.* 2009) and the Landscape Logic project involving Regina Magierowski.[4] A key preliminary finding was that the proportion of native forests managed for wood production resulted in substantive macroinvertebrate community compositional changes above a threshold of around 45% in the catchment area when under a clearfell, burn and sow management regime (combined with associated roading). By contrast, catchments dominated by protected forests and plantations within catchments could not be well differentiated by macroinvertebrate assemblage composition. Catchments with large amounts of agricultural land

[4] Results are not yet published, but see http://www.landscapelogic.org.au/projects/project4.html for further information on the project.

(>40% of area under grazing management) had substantially altered assemblages. The drivers of macroinvertebrate assemblage change were the impact of fine sediment deposition into the streambed for both agricultural and forestry land uses. This was accompanied by a prevalence of algae in agricultural catchments arising from higher nutrient levels and lighter environments and a reduction in riparian forest vegetation. This study did not observe major deleterious impacts from eucalypt plantations on aquatic biodiversity when compared to more intensive forest and agricultural management regimes at catchment scale.

Stewart (2011) recently tested the quality of water and compared macro-invertebrate assemblages in a number of waterways in south-west Western Australia that differed in their surrounding land use. Five land uses within 200 m of streams were compared: *E. globulus* plantations, fenced pasture, unfenced pasture, remnant native vegetation, and a mix of pasture and remnant native vegetation. The quality of water and condition of riparian zones was poorest in unfenced pasture and best in remnant native vegetation. Despite the relatively recent conversion of pasture sites to *E. globulus* plantations, the water quality and condition of riparian zones in eucalypt plantations was intermediate. The aquatic macro-invertebrates did not show any significant patterns corresponding to surrounding land use. It is possible that the whole-catchment effects of land clearing and agriculture are long-lasting and override fine-scale differences in current land use (Harding *et al.*, 1998). The results from Stewart (2011) suggest that eucalypt plantations do not negatively impact on aquatic biodiversity. Rather, these results suggest that riparian zones and water quality may actually benefit from plantation establishment compared to pastures.

Currently there is little evidence that the fertilisation of forests causes negative impacts on water quality either globally (Binkley *et al.*, 1999) or locally with eucalypt plantations in southern Australia. However, this could change in the future as first-rotation plantation crops are harvested and second-rotation plantation crops become a larger proportion of the plantation estate. This is because second-rotation crops will need more fertiliser than first-rotation crops, which relied on the residual high fertility from agricultural crop and pasture land uses prior to plantation establishment (Mendham, 2007).

Some members of the general public have expressed concerns that the runoff from eucalypt plantations, specifically *E. nitens* plantations in the George River, Tasmania, contain toxic chemicals and are deleterious to human health, aquatic and marine organisms (George River Water Quality Panel, 2010). A panel comprising nationally and internationally recognised experts in the areas of human health epidemiology, ecotoxicology, water quality, oyster health, and eucalypt biochemistry was assembled to investigate these claims. A rigorous review of all available evidence found that there were toxins that were harmful to marine and aquatic life but that the natural concentrations were not high and that *E. nitens* plantations were unlikely to be the source of the compounds since water in sub-catchments without any *E. nitens* plantations contained these toxins.

4F. Water use by plantation trees

Afforestation of agricultural landscapes may alter hydrological processes in several ways (reviewed by van Dijk & Keenan, 2007). One of the major concerns is that plantation trees will reduce surface runoff, infiltration and ultimately streamflow and groundwater recharge. This can occur because deep-rooted perennial vegetation such as trees use more water than short-lived annual crops and pasture grasses (Zhang *et al.*, 2003). At a catchment scale, the replacement of pasture with plantations can alter groundwater levels and streamflows, and affect aquatic and riparian biodiversity. The consequential changes in hydrological flows caused by plantation establishment will vary from catchment to catchment depending on climate, topography, soils, past vegetation and land uses, and size and location of plantations in the catchment (Townsend, 2003; Vertessy *et al.*, 2003; Zhang *et al.*, 2003).

In Western Australia, clearing of native vegetation and replacement with crops and pasture grasses has increased water infiltration and led to widespread ecological problems of rising groundwater (Stirzaker *et al.*, 2002). These problems have been exacerbated because the groundwater is saline. Rising saline water tables are a major threat to biological diversity through waterlogging, inundation and the mortality of large areas of native vegetation (Cramer & Hobbs, 2002; Prober & Smith, 2009). Over 1.8 million hectares of agricultural land in the wheat belt of Western Australia is threatened by salinisation (Parsons *et al.*, 2007). However, revegetation, including eucalypt plantation establishment, may play an important role in lowering saline water tables (Stirzaker *et al.*, 2002; Parsons *et al.*, 2007; Prober & Smith, 2009), with occasional exceptions (see Sudmeyer & Simons, 2008), and reforestation has contributed to reducing salinity in at least four rivers in Western Australia (Parsons *et al.*, 2007).

The ability for eucalypt plantations to reduce groundwater tables can also have adverse effects on tree health. Considerable evidence from Western Australia shows that first-rotation *E. globulus* crops have performed well because the absence of deep-rooted perennials in previous pasture and cropping regimes created, over many years, a high water table, which first-rotation tree crops can utilise (White *et al.*, 2007). Over the first rotation, these trees gradually lower the water table (and nutrients) to the point where this first rotation can dry out the soil profile to depths of 10 m (White *et al.*, 2007). However, this now means that some second-rotation crops of trees are not performing as well as the first rotation and, in the absence of management intervention, this may lead to tree moisture stress (White *et al.*, 2007), which may affect overall plant and soil health, biomass, structural complexity and possibly biodiversity. Potential options to manage water stress are to leave the land to lie fallow between rotations to allow groundwater recharge (although this is not compatible with coppicing), pruning, fertilising, or lowering the tree density of the second rotation either through thinning or by planting fewer seedlings at establishment (White *et al.*, 2007).

In eastern Australia, rising saline groundwater is less of a problem but there are concerns that plantation trees intercept and utilise too much water, lowering the water table and reducing surface runoff and groundwater recharge that sustain streams and rivers (Vertessy *et al.*, 2003; Benyon *et al.*, 2006; but see Townsend, 2003). Lane *et al.* (2005) showed that, in small catchments, the number of zero-flow days can be significantly increased through afforestation. Plantation establishment can therefore lead to reduced and interrupted streamflows and consequently affect aquatic biodiversity. However, the amount of water that plantations intercept from entering streams can be minimised by limiting the total size of plantations in catchments, predominantly establishing plantations in climates that receive less than 800 mm precipitation per annum, staggering the age class of plantations, and by placing plantations on hillsides rather than valley floors near drainage lines (Vertessy *et al.*, 2003; Zhang *et al.*, 2003; Parsons *et al.*, 2007). Even in the major plantation regions of Australia, plantations are a relatively minor proportion of the total catchment and their contribution to reduced flows in streams and rivers has probably been overstated (Parsons *et al.*, 2007). Nevertheless, new plantations are now subject to assessments of their likely impact on catchment hydrology (Parsons *et al.*, 2007). It should be remembered that the overall impacts of plantation establishment on water use are relative to competing land uses and to the point in time used as the benchmark for comparison. If comparisons of water use of plantations are made with the native woodland vegetation of 1780, the conclusions will be very different from those made based on the cleared land of 1980.

5. Temporal and spatial scale for plantation effects to emerge

When considering the net contribution of eucalypt plantations to the persistence of biodiversity in landscapes of southern Australia, it needs to be remembered that eucalypt plantations are a relatively recent land-use change (Parsons *et al.*, 2006). Therefore, observing strong positive or negative effects on biodiversity is unlikely especially for long-lived organisms such as trees that may show time-lags in their response. Furthermore, the trees in eucalypt plantations in southern Australia are still relatively young and it may take some time for favourable habitat conditions or resources to appear before other species can colonise. In any case it is unlikely that eucalypt plantations will support much biodiversity in the first few years before canopy closure. The short life-cycle or rotation of *E. globulus* and *E. nitens* plantations (8–15 years) is insufficient for the development of specialised habitats which some forest specialists require (Vesk *et al.*, 2008). However, this may actually be beneficial when considering tree flowering and potential pollen-mediated gene flow (see Section 4A). The short rotation times and intense management of plantations mean that the biodiversity values of these plantations is likely to be lower than less-intensively managed plantations with longer rotation cycles (Brockerhoff *et al.*, 2008).

Although data are sparse, it is more than likely that the harvesting phase of the plantation cycle would have a detrimental effect on biodiversity. Harvesting would bring obvious changes to the physical environment within plantations, forcing mobile organisms to disperse to other plantations nearby, while more sedentary organisms may face local extinction if they cannot survive altered physical conditions (see Section 2C). Harvesting would also remove the positive edge effects on adjacent native vegetation provided by trees until the next crop is grown (see Section 3D). Spatially interspersing plots to be harvested at a similar time potentially allows organisms in plots being harvested the opportunity to disperse to neighbouring unharvested plots, thereby minimising the stand-level effects and hydrological effects across the landscape (Brockerhoff *et al.*, 2008). The effects of disturbance caused by harvesting have natural equivalents such as catastrophic wildfire or wind events (e.g. storms and cyclones). The net effect of these disturbances, plantation harvesting and short rotations is likely to select for mobile species that are opportunists and adapted to disturbance.

6. What don't we know about biodiversity in plantations?

Although a considerable number of studies have been published in recent years concerning biodiversity of eucalypt plantations in southern Australia, several topics remain poorly understood. This review has identified numerous limitations concerning data on biodiversity from eucalypt plantations. Most of the research tends to be short term and there are few long-term studies of faunal use in eucalypt plantations. Most studies assessing eucalypt plantation biodiversity have space-for-time based designs that sample eucalypt plantations, remnant vegetation and agricultural sites at one or several relatively short points in time. From this sampling it is inferred that the biodiversity detected among agricultural sites existed in plantation sites prior to trees being planted. Clearly, this approach has its limitations (Majer & Nichols, 1998). This review did not find any studies that have monitored biodiversity continuously at the same sites to document the biodiversity changes that take place through a full rotation from before trees are planted right through to harvesting. Such studies would be beneficial in determining whether harvesting in effect 'resets' the successional clock. In this context it would be useful to compare canopy, ground and below-ground components of biodiversity as they might respond differently.

No specific studies have determined whether eucalypt plantation establishment facilitates dispersal between native vegetation fragments in Australia, although this has been studied in pine plantations (Banks *et al.*, 2005; Taylor *et al.*, 2007). Likewise, little work has determined what effects plantation establishment (i.e. a change in matrix from pasture to trees) may have on the biodiversity within remnant vegetation, although this question will be addressed soon for birds by Mayumi Knight's PhD studies. Another question worthy of research attention is whether biodiversity losses from the demise of biological legacies (i.e. tree death) outweigh gains from succession and landscape-level management of harvesting and rotations.

This review has identified that little has been documented concerning the history of plantation establishment. What proportion of the remnant vegetation was cleared to establish eucalypt plantations and where? In Victoria, no remnant tree can be removed during the establishment of plantations. In Tasmania, until recently native forest could be converted to eucalypt plantation (Forest Practices Authority, 2009a). Further research is urgently needed to trial techniques that may lead to the cost-effective restoration of degraded patches of remnant vegetation, and this is being addressed in part by researchers in Tasmania and Western Australia.

Few studies considered multiple taxonomic groups, with the notable exception of the combined efforts of Hobbs *et al.* (2003) and Cunningham *et al.* (2005) on the same set of sites in south-west Western Australia (see Box 1) and the vertebrate studies of Kavanah *et al.* (2005; 2007) and Law and Chidel (2006) in northern Victoria and southern NSW. Among the eucalypt plantation biodiversity literature, different taxonomic groups were measured using different measurements and different sets of sites, and this makes comparisons and generalisations challenging. For some of the issues covered in this review, strong positive or negative effects were not observed possibly because some of the effects on biodiversity caused by plantation establishment were weak in magnitude (e.g. buffering of edge effects), or not enough time had elapsed since plantation establishment, so effects were difficult to detect.

It is important to recognise the dangers in making generalisations based on a limited set of studies. Even apparently well-known systems can behave in unexpected ways. For example, Lindenmayer and colleagues found that vertebrate species in two nearby landscapes of eucalypt remnants within pine plantation matrices behaved differently (Lindenmayer *et al.*, 1999; 2002), and this raises serious doubts about the generality of results and applicability to other landscapes and forest-matrix types. By contrast, studies by Loyn *et al.* (2007; 2008; 2009) on different regions found similar faunal responses to eucalypt plantation establishment.

There may also be some regional differences in the biodiversity values of eucalypt plantations. For example, the studies by Loyn *et al.* (2007; 2008; 2009) in Victoria and South Australia suggested greater faunal use of eucalypt plantations than those conducted in Western Australia (Hobbs *et al.*, 2003). Such regional differences could arise from different patterns in the past clearing of native vegetation (DEWR, 2007) and hence differences in the degree of habitat loss, fragmentation, and declines in species. Alternatively, this may be explained by differences in the degree of similarity of plantation tree species relative to endemic eucalypts.

7. Conclusions

This review has shown that eucalypt plantations are not 'biological deserts', and have considerable biodiversity values. In southern Australia, it is evident that eucalypt plantations do not provide biodiversity values as high as remnant eucalypt vegetation, although plantations do provide much higher biodiversity values than agricultural land uses. This key conclusion concurs with other international and national reviews of biodiversity and plantations (Lindenmayer & Hobbs, 2004; Carnus *et al.*, 2006; Munro *et al.*, 2007; Stephens & Wagner, 2007; Brockerhoff *et al.*, 2008; Felton *et al.*, 2010). But is the net outcome of eucalypt plantation establishment in southern Australia on biodiversity positive or negative? At the level of individual plantations, it is likely that the answer to this question will vary with each region and may even be plantation specific. This is because the factors (discussed in this review) that influence biodiversity vary considerably at the scale of each plantation.

Of particular importance is the previous use of the land prior to establishment or conversion to plantation, especially the amount of clearing of native vegetation during plantation establishment. Most plantations have been established on agricultural land where there are few remnant trees, whereas a number of plantations (in Tasmania) have been established through the conversion of native forest (Forest Practices Authority, 2009a). Although the data do not exist, it is reasonable to assume that the vast majority of future plantations will be established on agricultural land in which the remnant vegetation (which may consist of paddock trees or remnant stands of native forest) will be retained and protected (e.g. Fig. 3). In Victoria, individual paddock trees of significant size are protected, although not all single trees will be retained. This review has shown that remnant vegetation and biological legacies have a massive positive influence on the biodiversity values of eucalypt plantations. Thus, at the level of individual eucalypt plantations, plantations that retain the most native vegetation are likely to generate the most positive biodiversity outcomes.

At the other end of the scale, plantations established from the clearing of native vegetation are certain to have net negative impacts on biodiversity (Brockerhoff *et al.*, 2008). It is generally accepted that plantation establishment through the conversion of native forest in Tasmania has had an overall negative effect on biodiversity in some regions of the state. Nevertheless, some argue that compared to the harvesting of native forests, plantation establishment provides productivity benefits which could translate to a smaller area (converted from native forest) needed to support a continuous supply of wood products compared to native forest (Carle *et al.*, 2002). If this area saving was offset against the protection of native forest, this could reduce the overall negative impact of plantation establishment from the conversion of native forest, as has happened in Tasmania with the Regional Forestry Agreement (DAFF, 2005). However, this argument ignores the contribution to biodiversity conservation of native forest areas outside formally protected reserves (e.g. remnants on farm land) and the role eucalypt plantation establishment can play in enhancing positive biodiversity outcomes in these landscapes (Forest Practices Authority, 2009c; and see below).

At a larger landscape scale, the net outcome of eucalypt plantation establishment is likely to be positive. The adequate protection of biodiversity requires thinking beyond the immediate boundaries of conservation reserves, and considering biodiversity within whole landscapes and their multiple uses (Margules & Pressey, 2000; Bennett *et al.*, 2006). In this regard, eucalypt plantations in southern Australia can play an important role in conserving biodiversity at a whole-landscape scale. Furthermore, because eucalypt plantations are able to support at least some species and provide numerous other biodiversity benefits, they may well be able to take some pressure off the areas protected in national parks and conservation reserves in regions with limited formal reserves. If some species are able to be supported by plantations, this could even allow species not supported by plantations to receive more targeted management in remnants (e.g. via suitable fire regimes). Importantly, the major vegetation groups most heavily cleared (eucalypt open forests and especially eucalypt woodlands—DEWR, 2007) coincide with the regions where eucalypt plantations are being established in southern Australia. Thus, for the regions where clearing of native vegetation has been acute, the protection of small patches of remnant vegetation and reduction in livestock degradation by inclusion within plantation estates provides obvious biodiversity benefits if enhanced and buffered during each harvest rotation.

This review has demonstrated the large influence that biological legacies and adjacent remnant vegetation have on the biodiversity values of eucalypt plantations. Replacing a matrix of pasture with eucalypt trees can lead to significant biodiversity benefits for remnant vegetation through several mechanisms including reducing livestock grazing pressure in remnants, buffering of edge effects, and potentially the facilitation of dispersal between remnants. Given the disproportionate positive contribution of remnant vegetation to landscape-level biodiversity values, perhaps biodiversity-focused management efforts in plantation estates should be concentrated on protecting and restoring remnant vegetation within plantation estates. That is, we should forget about economic and biodiversity trade-offs within plantations that maximise structural complexity, as has been suggested by many authors (see Hartley, 2002; Lindenmayer & Hobbs, 2004; Kanowski *et al.*, 2005; Carnus *et al.*, 2006; Brockerhoff *et al.*, 2008). Instead, we should direct management efforts towards conserving and rehabilitating remnant vegetation. Maximising structural diversity within plantations for biodiversity gains is largely incompatible with short-term rotations for pulp. Focusing on managing remnant vegetation is likely to maximise the biodiversity values of plantation estates even if many remnants are in poor condition and in need of rehabilitation and restoration.

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