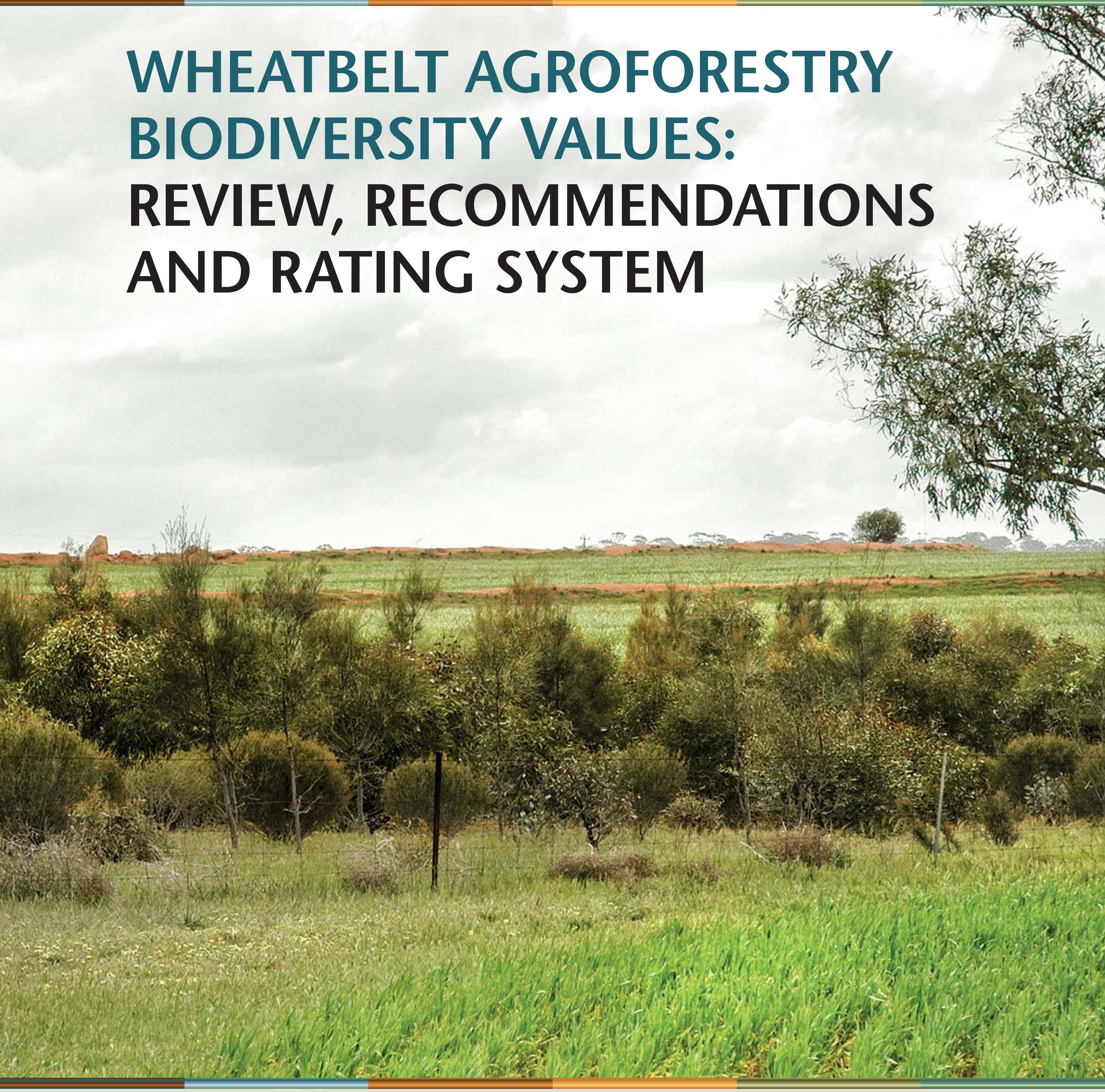


WHEATBELT AGROFORESTRY BIODIVERSITY VALUES: REVIEW, RECOMMENDATIONS AND RATING SYSTEM



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WHEATBELT AGROFORESTRY BIODIVERSITY VALUES: REVIEW, RECOMMENDATIONS AND RATING SYSTEM

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DISCLAIMER



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EXECUTIVE SUMMARY

Agroforestry systems are likely to play an important role in supporting biodiversity in highly modified landscapes, such as the Western Australian Wheatbelt, where the reserve system is unlikely to be expanded and landowners are looking to diversify their land-use options and incomes. This report is intended to help land managers, community groups and industry bodies incorporate biodiversity outcomes when planning agroforestry plantations.

This report reviews the current literature to provide a summary of the biodiversity benefits of agroforestry systems which could be utilised in the Western Australian Wheatbelt. Recommendations are then made for potential habitat manipulations appropriate for agroforestry systems to increase biodiversity outcomes. A new scoring system is also proposed, in which various agroforestry systems can be compared in terms of their potential benefits to biodiversity.

The literature review demonstrated that all agroforestry systems tend to increase the habitat value of the site beyond that of normal farmland, but that the systems do not directly mimic natural vegetation, with particular taxa usually missing from the system. This increase in habitat value is most clearly evident for birds for which most research on agroforestry systems has been carried out.

The range of habitat manipulations possible can broadly be described as augmentation of habitat strata, both in terms of structural diversity and species diversity. At a broader scale, conservation benefits in agroforestry may be increased if plots are large, and adjacent to remnants of indigenous vegetation.

The proposed habitat scoring system provides a tool to assess the biodiversity value of an agroforestry system at the planning stage. However, the system is a prototype that will require field testing, and consequent adjustment as new knowledge is developed. It is similar to several other schemes, in that it is an additive scoring system, based on the presence and diversity of various habitat strata. It is a standalone system, in that it does not require comparison with an analogue. This is particularly appropriate as a habitat-rich agroforestry system will be a novel habitat, not directly comparable to local native vegetation.

‘The literature review demonstrated that all agroforestry systems tend to increase the habitat value of the site beyond that of normal farmland, but that the systems do not directly mimic natural vegetation…’

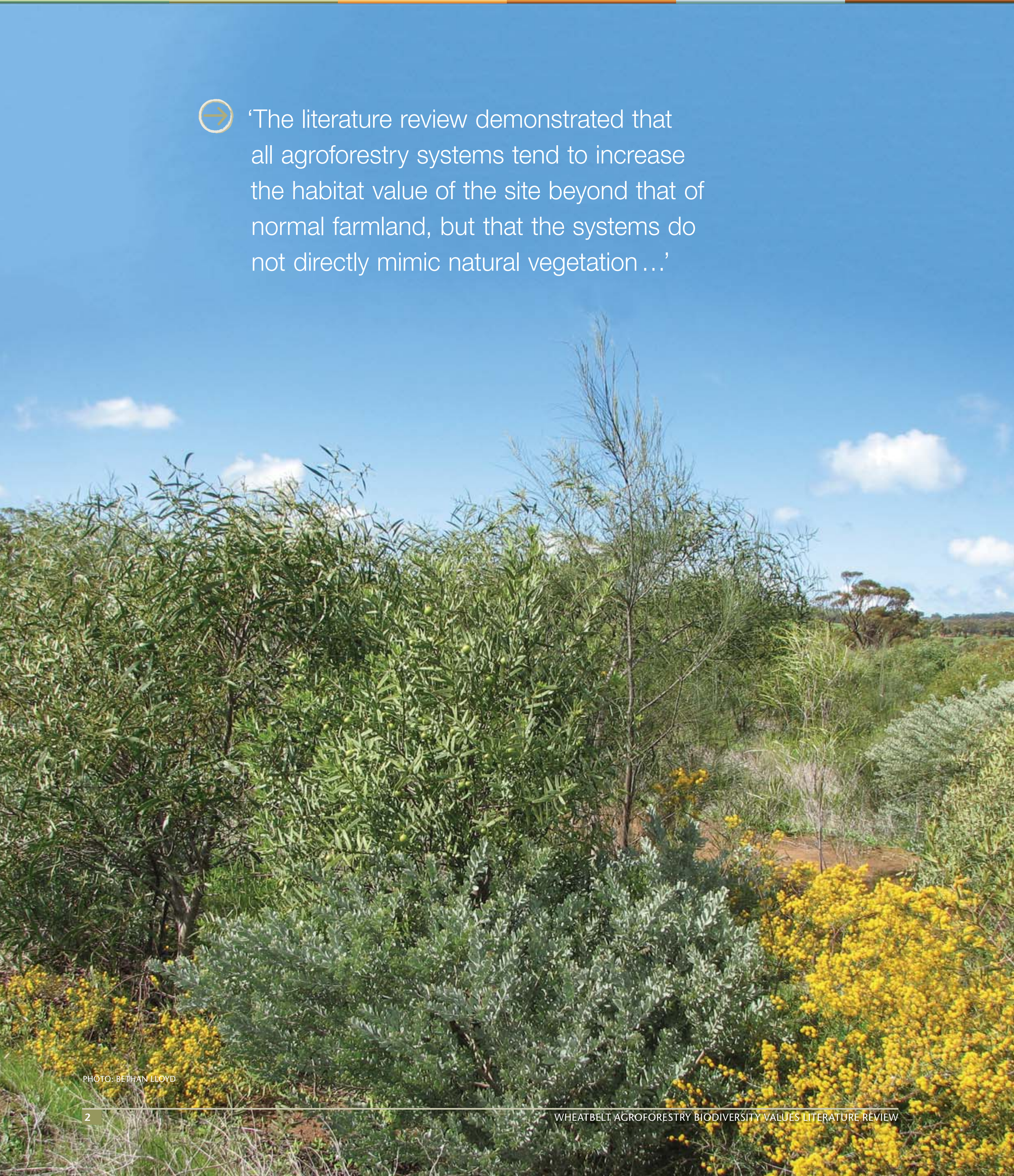


PHOTO: BETHAN LLOYD



INTRODUCTION

There is general consensus that Australian agricultural landscapes could be better managed for a balance in production and protection of biodiversity values. As part of a scheme to accomplish this goal, Smith et al. (2013) have suggested that around 40% of each of these landscapes should be maintained as "modified intensive agriculture", while 20% is "utilised natural vegetation", with the balance comprised of core conservation areas and intensive agriculture.

The Western Australian (WA) Wheatbelt has significant capacity to support modified intensive agriculture in the form of perennial agroforestry systems, which have the potential to represent both the modified intensive agriculture and utilised natural vegetation systems. Perennial agroforestry systems have been recognised as having the capacity to deliver key biodiversity outcomes, as well as a number of other environmental benefits including addressing land degradation and water quality issues. As such, they are gaining recognition as a key component of a sustainable agricultural landscape in Australia (George et. al., 2012). However, in order to optimise the value that can be derived from these systems, tools need to be developed that allow the planning of agroforestry systems to consider both their biodiversity and financial outcomes.

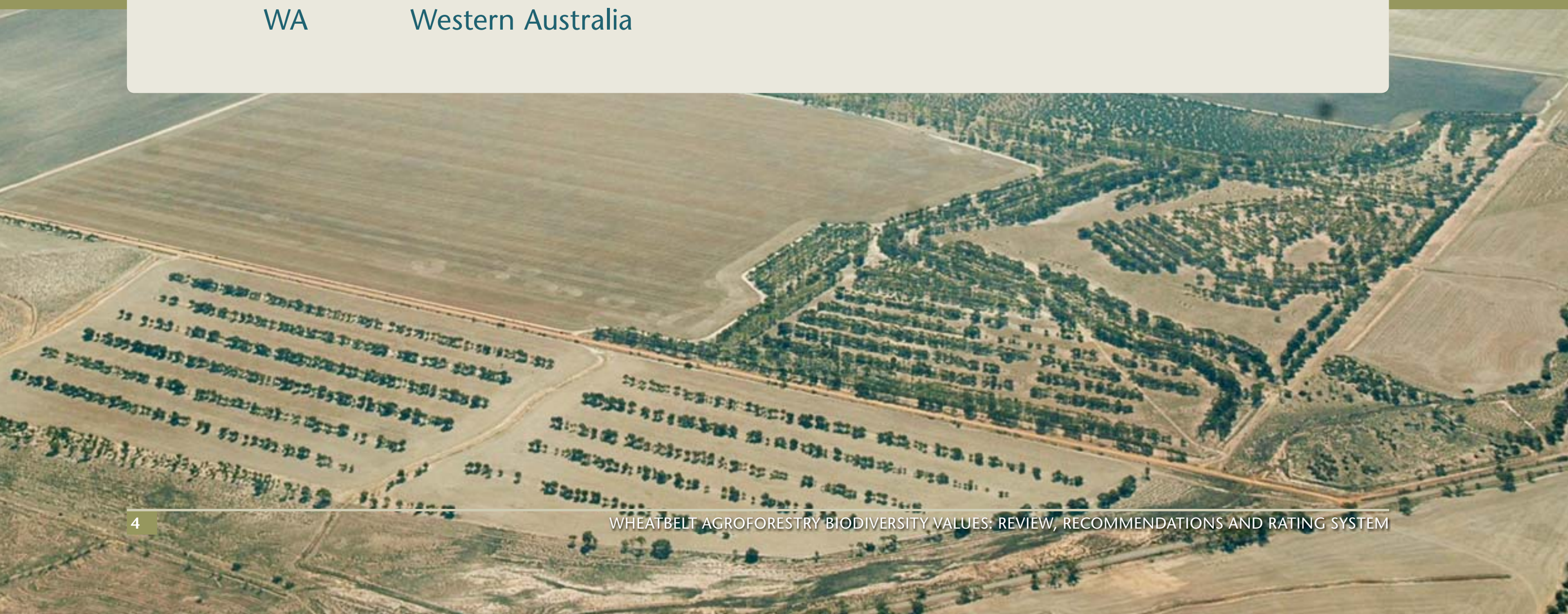
This report has three components that collectively explore biodiversity values in WA Wheatbelt agroforestry, where agroforestry is defined as "the intentional integration of trees and shrubs into crop and animal farming systems to create environmental, economic, and social benefits." (USDA National Agroforestry Center, 2015). In the first section, current knowledge on the biodiversity supported within agroforestry systems is reviewed. This mainly focuses on native perennial systems that are already established within the WA Wheatbelt, but draws on research from around Australia. The second section identifies possible habitat enhancements that could improve the habitat quality of such agroforestry systems. As there are very few studies of enhancement of agroforestry systems, this section draws on literature for plantation forestry, revegetation and natural remnants. In the final section, a preliminary habitat value scoring system is developed. This scoring system will allow the comparison of various agroforestry systems, and be able to consider possible enhancement within the context of a financial trade-off.

With the aid of this document, Land Managers will be able to:

- plan for the development of agroforestry systems that incorporate habitat benefits,
- improve the habitat value of any given agroforestry system, and
- compare trade-offs between particular agroforestry systems, their enhancements and financial outlay when considering their investment.

ABBREVIATIONS

CRC	Cooperative Research Centre
CSIRO	Commonwealth Scientific and Industrial Research Organisation
ha	hectare(s)
km	kilometre(s)
m	metre(s)
mm	millimetre(s)
NRM	Natural Resource Management
NSW	New South Wales
WA	Western Australia



AGROFORESTRY SYSTEMS AND THEIR KNOWN BIODIVERSITY OUTCOMES

This section outlines the biodiversity benefits and possible enhancements to improve biodiversity within existing agroforestry systems established in the WA Wheatbelt. The systems described are largely designed to be integrated within existing Wheatbelt farming systems, as the economic viability of each as a stand alone production system can be limited given the low rainfall of the region (Harper et. al. 2012). It should be noted that significant research exists on the production aspects of these systems, but is not specifically reviewed here unless it relates to the biodiversity value of the system. Further information on implementation and management of these systems can be found in the Wheatbelt NRM Agroforestry Guide (Wheatbelt NRM, 2013).

OIL MALLEE

Oil mallees have been planted with a range of purposes including the provision of biomass (eucalyptus oil, bioenergy) and carbon sequestration. They are generally grown in narrow, unfenced belts. Wheatbelt NRM (2013) lists six eucalypt species recommended as Oil Mallee and six Mallee Eucalypts for biomass. Oil mallee systems have probably been the subject of more biodiversity related-research than any other agroforestry system. This has included work carried out by the Commonwealth Scientific and Industrial Research Organisation (CSIRO), Future Farms Industries Cooperative Research Centre (CRC), and associated research students.

While oil mallees enhance habitat structure by adding a tree canopy layer, as do most agroforestry systems, the belts do not possess a shrub layer, or a complex layer of litter and woody debris, and hence habitat structure is simpler than natural eucalypt woodlands (Smith 2009b). Oil mallees are, however, likely to provide good quality food resources for birds, including nectar and invertebrates. In Wheatbelt oil mallee systems, Smith (2009a) recorded 22 foraging bird species which were known to be declining in the Wheatbelt. However, bird species diversity in oil mallees is lower than that in remnant woodland or mixed revegetation plantings (Smith, unpublished data).

A study also shows that oil mallees provide sufficient nectar to support the foraging of Western Pygmy Possums (Short *et al.* 2009). The possums nested in nearby tree hollows, but foraged amongst the oil mallee belts. However, in general, mammal capture rates are significantly lower in oil mallees than they are in remnant woodland (e.g. Smith, unpublished data).

Reptile sampling often only provides low numbers of samples, and only one unpublished study has examined reptile use of oil mallees. Reptile diversity is significantly lower in oil mallees than remnant woodland or mixed revegetation plantings (Smith, unpublished data).

In an honours thesis, Leng (2006) demonstrated that oil mallees may support significant numbers of ground-dwelling insects, but the terrestrial beetle assemblage was significantly different than that of nearby *Eucalyptus* woodland. Leng examined terrestrial beetle fauna using pitfall traps, while Lyons (2009) examined arboreal insect species (focussing on beetles) in oil mallees using chemical fogging. Lyons (2009) found that the diversity of insect orders was similar between oil mallees and nearby woodland but the actual composition of beetle species was quite different between the two systems. These studies suggest that oil mallees support a good range of insects, which can then be utilised by birds and mammals as food, but that the mallees are not necessarily providing habitat for those insects usually dependent upon the nearby natural woodlands.

PHOTO: BETHAN LLOYD

SANDALWOOD

Sandalwood (*Santalum spicatum*) is a small tree indigenous to much of semi-arid Australia, including Western Australia, and is harvested principally for its oil, and has supported a successful export industry for over 150 years (Shea et al. 1997). It is a hemiparasite, parasitising the vascular tissues of other plants, making it essential that sandalwood be grown with an appropriate host plant species. Within part of its native range, Woodall and Robinson (2003) recorded 68 different host species amongst a range of plant families. Sandalwood is most often grown with a single host species—*Acacia acuminata*, which supports higher growth rates and has been found to be an important host species for Sandalwood in Wheatbelt vegetation (Fox, 1997). But up to 40 host species have been grown in one plantation (Geoff Woodall, pers. comm.).

This diversity of host plant species clearly offers potential for increased habitat structure and biological diversity, which could support a range of taxa. It is important to note that the host species selected for the plantation can have a direct impact on the growth rates and potential economic values of the sandalwood plant (Brand 2002).

Sandalwood tends to be grown in blocks, with the row configuration and spacing varying greatly between plantations. Sandalwood systems are often divided into two types: "single host" (*Acacia acuminata*) and "biodiverse" (a range of host species).

Some research on the habitat value of sandalwood has been carried out, including a report commissioned by Avongro (Gove 2012), and an unpublished PhD thesis (Leng, pers. comm.). All research has been carried out in Western Australia.

Due to the mixture of species, habitat structural complexity in sandalwood plantations is generally high for a plantation, but lower than natural woodland due to lack of a taller canopy species, and lack of some

of the coarser woody debris (Gove 2012). Sandalwood plantations quite often include an occasional *Eucalyptus* tree which adds to the habitat complexity (Gove 2012).

Gove (2012) found that sandalwood systems in the Western Australia Wheatbelt supported a high number of bird species, many of which are woodland-dependent, or declining in the wheatbelt. The species diversity was similar to that of nearby natural woodlands. While it may be expected that the single-host plantations would support less diversity than biodiverse host plantations, Gove (2012) did not demonstrate a difference between the two systems. This may have been due to the fact that replication was reasonably low, and that the overriding factor in determining biodiversity value was plantation age (and hence complexity).

In the study by Gove (2012) the diversity of bee and wasp species in sandalwood was similar to that of nearby woodlands. While this may suggest that sandalwood is particularly good insect habitat, in this study, lupin crops also supported similar numbers of bee and wasp species. Ground-dwelling insect abundance was greater in sandalwood than paddocks and slightly higher in remnant than sandalwood (Leng, pers. comm.). Insect diversity was also intermediate in sandalwood.

No studies of mammal diversity in sandalwood plantations are known, however, Leng (pers. comm.) studied the use of sandalwood plantations by reptiles. Reptile abundance was similar across paddocks, sandalwood and remnant vegetation. However, sandalwood plantation species richness for reptiles was intermediate between remnants and paddocks, suggesting there was improvement in habitat value as compared to paddocks but that there are some limitations when compared to remnant woodland.

'This diversity of host plant species clearly offers potential for increased habitat structure and biological diversity, which could support a range of taxa…'

TIMBER

Another tree-based agroforestry system is timber production which is an option in the higher rainfall regions of the WA Wheatbelt. Wheatbelt NRM (2013) lists *Pinus pinaster*, *Casuarina obesa* and four species of *Eucalyptus* as appropriate for the WA Wheatbelt. No biodiversity research has taken place on these species, in either a plantation or agroforestry setting. However, relevant information can be gleaned by consideration of biodiversity studies in other potentially similar eucalypt and pine systems.

In Western Australia, studies of biodiversity benefits associated with timber production have focussed on *Eucalyptus globulus* (Blue gum) which grows in the higher rainfall regions of Western Australia. Hobbs *et al.* (2003) estimated that four- to six-year old eucalyptus plantations in Western Australia were half as structurally diverse as native forest, due primarily to the lack of a shrub layer, and the presence of large areas of bare ground. As a consequence, Hobbs *et al.* (2003) found more bird species in forest than *E. globulus* plantation, particularly ground foraging, shrub and canopy insectivores.

In New South Wales (NSW), Law *et al.* (2014) demonstrated that, after 11 years, indigenous eucalyptus plantations still supported fewer bird species than nearby forest plots in NSW. Birds that were lacking were categorised as "forest birds".

Loyn *et al.* (2007) found far more forest and woodland bird species in *E. globulus* plantations than farmland in eastern Australia, and species diversity was only marginally less than that of native forest. The detailed examination of the various feeding guilds in this study provides insight into the specific responses to habitat provision: insectivores that foraged in canopy and tall shrub layers were equally as common in the two systems, while nectarivores, carnivores, species that foraged among low shrubs and those that forage among eucalyptus bark were relatively less common in the plantations.

Likewise, Munro *et al.* (2011) found that eucalyptus woodlots had the same bird species richness as ecological plantings and native vegetation remnants, but woodlots possessed more generalist species while the ecological plantings contained more shrub-related bird species. Like the study of Loyn *et al.* (2007), both the woodlots and ecological plantings failed to support species dependent upon mature trees (Tree Creepers and Sittellas), which were found in the remnant plots. In a study in south Western Australia, fewer bird species were found in Blue gum plantations than the embedded remnants (Archibald *et al.* 2010).

Due to their low numbers of occurrences in samples, reptiles and amphibians are often overlooked in favour of the more ubiquitous birds. However, Hobbs *et al.* (2003) recorded twice as many reptile and amphibian species in forest compared to *E. globulus* plantations.

Hobbs *et al.* (2003) also found that the Southern Brown Bandicoot and Bush Rat were common in remnant vegetation but absent from *E. globulus* plantations. Similar trends, although not statistically significant, were revealed for plantations when compared to small remnants embedded in the plantations in southern Western Australia (Archibald *et al.* 2011).

The same study also demonstrated that native forest had the highest use by bats, and isolated plantations were used far less than those adjacent to native forest. Law *et al.* (2011) found eucalyptus plantations provided little habitat value for bats, which was attributed to a lack of old remnant trees and hollows. Loyn *et al.* (2008) recorded higher levels of bat activity in eucalyptus plantations than in farmland, although again, these values were lower than those of native forest. Loyn *et al.* (2008) also recorded mammals such as *Antechinus* and wallaby using the plantations.

Very few studies have examined the habitat value of timber to invertebrates. Several WA studies focus on invertebrates as plantation pests (e.g. Abbott *et al.* 1999), however Cunningham *et al.* (2005) examined the insect habitat value of Blue gum plantations in southwest Australia, and found that the plantations supported a different insect assemblage, with fewer species than native *Eucalyptus marginata* (Jarrah) forest. A Tasmanian study (Bonham *et al.* 2002) found that native snails and millipedes were less diverse in pine and eucalyptus plantations than in adjacent native forest. However, many taxa were equally as common in the plantations as the forest. Curiously, in terms of community composition, pine plantations were more similar to native forests than the eucalyptus (*E. nitens)* plantations. Given the contrast in rainfall it is difficult to extrapolate these results to the WA Wheatbelt, where native snails and millipedes are less common.

Loyn *et al.* (2009) found birds more abundant in native forest than plantations but somewhat more abundant in *E. globulus* than *Pinus radiata* (Radiata pine). Canopy foraging insectivores and nectarivores were common in *E. globulus* but virtually absent from *P. radiata*. Bark foraging insectivores were only abundant in native forest. As Radiata pine is the most common plantation species in Australia, significant research has focussed on biodiversity associated within this species. *Pinus pinaster* (Maritime pine) is recommended for the WA Wheatbelt (Wheatbelt NRM 2013), and there appears to be no biodiversity research associated with *P. pinaster*. While very similar in structure, contrasts between ecological outcomes in *P. radiata* and those expected for *P. pinaster* could occur due to the contrast in rainfall. However some consideration of the habitat value of *P. radiata* is warranted in light of the potential use of *P. pinaster.*

Lindenmayer and Hobbs (1994) summarised research into faunal use of radiata pine and concluded, like that of Loyn *et al.* (2009b), that animal assemblages in plantations are general less diverse than those of native forest.

Lindenmayer *et al.* (2008) examined the development of *P. radiata* plantations surrounding native forest, and observed that reptiles and native mammals were "virtually absent" from the six- to eight-year-old plantations (and hence Lindenmayer *et al.* (2008) focus on the analysis of data-rich birds). While bird species diversity increased over the first five years of plantation growth, diversity tended to then flatten out and was significantly lower than that of surrounding native forests.

ABOVE: *Eucalyptus maculata.*
BELOW: Wandoo.
PHOTOS: BOB HINGSTON

BRUSHWOOD

Brushwood is a shrub or small tree indigenous to much of Australia, including Western Australia. It is grown for the production of brush which is used in screening and fencing. Brushwood species appropriate for brushwood include *Melaleuca uncinata*, *M. atroviridis* and *M. hamata* (McKinnel 2008). All three species are native to the Western Australian Wheatbelt with *M. hamata* being more widespread throughout south-western Australia and slower growing. Brushwood is generally grown in blocks, formed by rows, with open laneways to aid in harvesting.

Little research has been done on the biodiversity supported within brushwood systems. Gove (2012) compared the value of brushwood systems with other agroforestry systems, woodlands and lupin crops.

The habitat structure of brushwood systems is relatively simple, with dense foliage at shrub level (i.e. up to 2 m), but a very homogenous structure at a plot-scale. Brushwood systems are usually maintained to a high level of simplicity and, therefore, few large trees are maintained, plant diversity is low and ground cover is limited (Gove 2012).

Brushwood supports fewer woodland bird species than sandalwood or natural woodland, however it does support some woodland-dependent bird species, including red-capped robins, and blue breasted fairy wrens (Gove 2012). These bird species probably benefit from the dense shrub layer of shelter, juxtaposed with the large open corridors usually maintained in brushwood systems. Brushwood systems contained several native bee and wasp species, and contained no less diverse assemblage of these species than the more complex sandalwood and natural woodland systems (Gove 2012).

FORAGE SYSTEMS

Perennial forage shrubs consist of a range of palatable native shrub species, particularly Chenopods. Wheatbelt NRM (2013) lists six forage species: five species of *Atriplex* and one *Rhagodia* species suitable for planting. It is grown both in belt-alley systems and contiguous blocks. The Enrich program (Future Farm Industries CRC, no date) also identified a range of high quality forage species which, in many cases, provided substantial habitat (Norman *et al.* 2008, Collard *et al.* 2011, Lancaster *et al.* 2012; detailed on next page). Given that the shrub layer is such an important habitat, particularly for a large number of declining bird species, it is worth considering the habitat value of shrub-level foraging systems along with brushwood systems in this review.

Seddon *et al.* (2009) found that the habitat structure of Old man saltbush (*Atriplex nummularia*) grown in belts with crop alleys (NSW) was intermediate in comparison to native woodland and pasture. Saltbush belts lacked the native overstorey, ground covering plants, old trees with hollows and fallen timber. However, saltbush belts possessed more native plant species and native mid-storey cover than conventional farmland. Seddon *et al.* (2009) described native grasses and herbs establishing under and around the saltbushes. Habitat complexity improved in the saltbush throughout the three-year study. Despite this habitat complexity, Seddon *et al.* (2009) did not demonstrate any increased bird diversity in the saltbush, as compared to the conventional agriculture. In contrast, Collard *et al.* (2011) studying contiguous blocks of Old man saltbush in South Australia found higher bird species diversity and abundance in saltbush than pasture, although values were still lower than that in native woodland. Saltbush treatments were also intermediate in terms of native plant diversity. Several native bird species utilise saltbush shrubland as it occurs naturally, and Collard *et al.* (2011) found several threatened species utilising saltbush, and included observations of nesting chats within saltbush. Richards (2013) also found that 10- to 20-year old rows of Old man saltbush created feeding resources and some nesting opportunities for more bird species than open pasture, but less than remnant woodland, with most birds being generalists or shrub-dependent species associated with the vegetation's simplified structure.

Norman *et al.* (2008) sampled invertebrate families in saltbush alley systems but found no discernible difference between unimproved pasture, saltbush planted pasture and native remnants.

Also studying blocks of Old man saltbush in South Australia, Lancaster *et al.* (2012) found that bobtail lizards (*Tiliqua rugosa*) not only occupied and utilised revegetated saltbush but the appearance of juveniles indicated successful breeding within the habitat.

PHOTOS: DEAN REVELL

SPECIES MOVEMENT THROUGH AGROFORESTRY SYSTEMS

Unhindered movement and dispersal for both plants and animals is considered important for the maintenance of populations particularly under a changing climate where survival may depend on the ability to migrate. An alternative to the creation of dispersal corridors across landscapes is to make the overall landscape more permeable. This is achieved by making the landscape "matrix", the land uses surrounding remnant habitats, more hospitable (Ricketts 2001). It is generally thought that converting open agricultural habitat to semi-wooded habitats promotes dispersal across the landscape (Salt *et al.* 2004). While there is little published evidence for increased movement through Australian agroforestry systems, other woodlot systems have acted as dispersal stepping stones (Uezu *et al.* 2008), while remnant trees in Australian farmland have aided in bird dispersal (Fischer and Lindenmayer 2002a). On some occasions, novel wooded systems may potentially discourage movement. For instance, *Pinus* woodlots, which contrast significantly with natural habitats, may act as barriers to movement, as shown by Villard and Hache (2012) in Canada.

CONCLUSION

All agroforestry systems appear to provide habitat beyond that provided by other conventional agricultural systems. However, agroforestry systems do not directly resemble natural habitats such as native woodlands or forests, with numbers of woodland-dependent species usually fewer than that found in woodland plots.

This species impoverishment can be broadly explained by the simplified habitat structure of agroforestry systems, with the lack of particular structural features readily related to observed species assemblage and absence of particular taxa (McElhinny *et al.* 2005). For instance, oil mallee provides overstorey forage for nectivorous birds and mammals but generally lacks a complex shrub layer, leading to a lack of smaller insectivorous ground and shrub dwelling species. Likewise, small insectivores are often supported in saltbush and brushwood systems, principally due to the provision of a dense shrub layer. Contrasts in invertebrate assemblages are less well understood but are probably due to lack of microhabitats, including bark and litter layers and particular food sources.

Interestingly, almost all the studies cited investigate the ecological value of one type of agroforestry system and very few comparisons are made between different systems.

No studies looked at the complementary effects of multiple agroforestry systems within the same farm or landscape.

Birds are clearly the most well understood group in terms of their use of agroforestry systems, which is assisted by their conspicuousness, moderate diversity and reasonably well studied ecologies. Insects are far more abundant, require specialised sampling and taxonomic expertise, and the individual habitat requirements of individual species are far from understood. Reptiles are infrequently a focus, and may be excluded from statistical analysis due to low abundance (e.g. Lindenmayer *et al.* 2008). They are probably perceived as not responding as favourably to habitat manipulations as layers that are likely to be important to reptiles, such as rocks, dead wood and thick litter, are not given as much emphasis as nectar-rich shrubs and diverse canopy layers. Dispersal is also likely to be limited in reptiles and hence colonisation of isolated habitat patches is unlikely (Archibald *et al.* 2011). In this regard, management that applies a range of manipulations beyond those of vegetation may assist a range of fauna other than birds.

PHOTO: BOB HINGSTON

ENHANCEMENT OF AGROFORESTRY SYSTEMS FOR BIODIVERSITY OUTCOMES

> **There has been little investigation of the potential modification of agroforestry systems and consequent increases to habitat quality, although several studies have examined the variables associated with habitat value of plantation forestry. Given this lack of research into the habitat elements contributing to agroforestry systems, the approach that has been applied here is to look at the elements which contribute to the habitat value of revegetation projects, and the habitat value of remnant woodlands. As manipulation of agroforestry systems for habitat is in essence an attempt to mimic natural woodland (or to reproduce some components), examining the elements of revegetation and remnant woodlands that contribute to biodiversity maintenance is the approach taken here.**

Some exceptions are the tests of habitat quality of eucalypt plantations as a product of proximity to remnant vegetation (Law *et al.* 2011, Hobbs *et al.* 2003, Cunningham *et al.* 2005). Furthermore, Gove (2012) examined the role of agroforestry species diversity but could not demonstrate any increase in habitat quality (as measured by birds, bees and wasps) in biodiverse sandalwood versus single host sandalwood. However, we would expect that increasing plant diversity would lead to an increase in animal diversity. The lack of difference found between the two different sandalwood systems may be due to limitations in the number and variation of sites visited and the similarities in complexity of habitat.

The majority of habitat enhancement studies focus on the response of birds because they are abundant, reasonably diverse and ecologically well understood. They are also particularly easy to survey compared to other fauna groups. In contrast, reptile numbers are often considered negligible (Lindenmayer *et al.*

2008), and unsuitable for statistical analysis (Cunningham *et al.* 2007). Birds also respond favourably to revegetation, and therefore provide encouraging results and a useful focus for rehabilitation efforts. Little is known of the response of insects to habitat manipulation, with the significant "taxonomic impediment" limiting knowledge of individual species and their habitat requirements and dispersal abilities. This taxonomic bias tends to be reflected more generally in restoration ecology where the emphasis tends to be on revegetation, with the expectation that fauna will naturally recolonise a site (Palmer *et al.* 1997). Amongst fauna, emphasis is typically placed on conspicuous vertebrate species, despite the abundance and important functional roles of invertebrates (Majer 2009).

Habitat variables which may potentially be enhanced can be divided into two categories:

- Within-site variables which are those of most interest to individual landholders and managers who wish to manipulate habitat quality.
- Landscape scale variables which may be of more interest to rural planners, policy makers and managers of larger scale processes, although farmers and managers may be able to make some choices as to where their agroforestry plots are located.

Management of within-site variables are principally focussed on increasing the habitat complexity of the site. As species have specialised niches, increasing habitat complexity provides more ecological space and allows the "packing" of more species (Yahner 1982). The approach taken here is to break habitat complexity into three strata (Lindenmayer *et al.* 2011) and examine the manipulation of each of these separately. These are:

- The ground layer;
- The understorey layer; and
- The overstorey layer.

Disturbance regimes (for example, fire history, local and exotic grazing) are important ecological factors but it is expected that their influence will be reflected in the structure associated with the various strata. Other variables considered here are larger scale, and are not conducive to direct manipulation, but may be considered at the planning stage. These are:

- Patch size and shape; and
- Landscape context.

These five elements are addressed individually in the following sections.

THE GROUND LAYER

Organic and non-organic material (litter, rocks, timber) which covers the ground layer and limits the proportion of open ground is an important habitat for smaller animals such as reptiles, amphibians and invertebrates. Many of these animal groups then form an important food source for birds and mammals. These materials also form important moisture and nutrient rich microhabitats, most suited for plant germination and survival. Each of these ground layer materials, including plants, is examined individually.

Litter

Litter increases moisture and nutrient retention and is an important microhabitat for invertebrates and small vertebrates. It is an important foraging stratum for many declining bird species, both worldwide and specific to the WA Wheatbelt.

Most studies directly related to habitat enhancement examine the response of birds to a range of litter cover levels. For instance, Stagoll *et al.* (2010) sampled a range of sites including woodland and treeless paddocks and found higher woodland bird species richness where leaf litter was present. Montague-Drake *et al.* (2009) found an increase in site occupancy of bird species of conservation significance with increased litter cover in wheatbelt vegetation remnants of New South Wales. In the 'Birds on Farms' study, which examined farms in both eastern and western Australia, bird diversity increased with increased litter cover (Barrett 2000).

Few other taxonomic groups have been studied in terms of litter as a potential variable of influence, however, Jellinek *et al.* (2014) found that litter cover was positively related to reptile abundance in revegetation and remnants. Given that the litter layer is such an important layer for invertebrate species, we would expect a relationship between litter density and invertebrate diversity and/or abundance (Ober and DeGroote 2011).

Rocks

Rocks form an important shelter for invertebrates and small vertebrates, many of which form prey for other larger animals. With the main focus of remnant habitats and revegetation being the response of birds, rock cover is not often considered a variable of interest. An experimental addition of artificial rocks (Croak *et al.* 2011) found that these rocks were colonised by more than 40 invertebrate species and six reptile species. In a study of revegetation and remnant vegetation, Jellinek *et al.* (2014) found that rock cover was related to reptile abundance. These studies indicate the importance of rocks for taxa not often considered in habitat manipulation studies.

Fallen timber

Lindenmayer *et al.* (2002) provide a thorough review of the ecological contributions of fallen logs and tree branches ("coarse woody debris"). Such material is not only habitat and foraging strata for animals but is also a germination site for plants and substrate for fungi.

Manipulations of coarse woody debris loads are relatively straightforward experiments, and hence there is quite clear evidence for the habitat value of increased coarse woody debris loads. Barrett (2000) quantified the value of fallen trees and estimated that: "For every 10 fallen trees present in a farm site, the diversity of ground-foraging birds increased by 30 per cent and bark-foraging birds by 70 per cent." Lindenmayer *et al.* (2010) also estimated that a doubling of log density per hectare in replantings lead to a 0.84 increase in bird species richness.

In several other correlative studies, log density has been positively associated with terrestrial mammal activity. In plots of revegetation Cunningham *et al.* (2007)

found that the density of logs was related to abundance of Ringtail possums and Lindenmayer *et al.* (1999b) found that small mammals were absent from radiata pine stands except where extensive areas of windrowed eucalypt logs were maintained.

Several studies in which log densities are manipulated have demonstrated a positive response by reptiles. Shoo *et al.* (2014) added coarse woody debris to revegetation plantings in Queensland wet tropics and found increased abundance and species richness of reptiles. Addition of logs in temperate woodland restoration led to increased reptile abundance (Manning *et al.* 2013). Brown *et al.* (2011) also found that log density in temperate woodland remnants was associated with the abundance and diversity of reptile species.

Logs are generally considered to be good habitat for a range of invertebrate species. In the same study system as that of Manning *et al.* (2013, above), Barton *et al.* (2011) demonstrated a positive response of beetle diversity from log additions. This study also demonstrated that logs could buffer the effects of heavy grazing by protecting small-scale beetle habitats. Logs and harvest material also provide safe sites for natural regeneration (Lindenmayer *et al.* 2002).

Ground layer vegetation

Ground layer vegetation represents an important microhabitat for small animals, and also represents an important food source which includes fruits and seeds, foraged by both reptiles and birds.

Barrett (2000) recorded an increase in bird diversity with an increase in native pasture and reduction in grazing. Montague-Drake *et al.* (2009) also found that several woodland-dependent species responded favourably to ground cover of moss and lichens. In woodlots and ecological plantings, Munro *et al.* (2011) found that weed cover was positively related to the number of generalist species. These studies indicate that ground layer management, which promotes indigenous ground cover vegetation, is likely to favour targeted bird species.

Reptiles forage amongst ground layer vegetation, and increased vegetation probably leads to increased food resources and shelter. Increased grass and tussock cover lead to increased reptile species in revegetation projects of NSW (Cunningham *et al.* 2007), while Brown *et al.* (2011) also found that reptile abundance was related to ground layer plant diversity.

Invertebrates are likely to shelter in ground layer vegetation but also forage on many of the plants, and these herbivores are likely to be prey of predatory and parasitic insects. Invertebrate diversity may be related to ground cover vegetation (Oxbrough *et al.* 2010), however, at a higher trophic level, Gove (2012) found no contrast in wasp diversity between lupin monocultures and more diverse native woodlands.

THE UNDERSTOREY LAYER

Many animals forage within shrub layers, and may even specialise on this habitat. Understorey insectivorous birds are often considered to be those experiencing the most rapid declines.

Arnold (2003) has suggested that when there is a perch within 1 m of the ground, the abundance of ground-foraging insectivorous birds increases. In a study of wandoo woodland Arnold (2003) found that insectivorous bird abundance was highest in remnants that possessed a shrub layer of *Banksia sessilis*. Lack of this understorey species also lead to a decline in honeyeater abundance. Within plots of revegetation, Arnold found a significant relationship between the abundance of ground and low-shrub insectivores and total leaf cover and number of branchlets to 2 m height.

Stagoll *et al.* (2010) sampled a range of sites including woodland and treeless paddocks and found that several sensitive woodland bird species responded favourably to the presence of a shrub layer. Montague-Drake *et al.* (2009) also found an association of woodland-dependent species on shrub cover or the number of strata in woodland remnants. Loyn *et al.* (2009a) found that in well structured ecological plantings, shrub cover was associated with bird abundance. Shrub presence in the 'Birds on Farms' study (Barrett 2000) led to an increase in woodland-dependent species, and small foliage gleaners and ground nesters. Particular to eastern Australia, shrub presence also led to a decrease in the presence of the Noisy Miner—a dominant honeyeater which excludes other bird species (Hastings & Beattie 2006, Lindenmayer *et al.* 2010). Munro *et al.* (2011) also found that species composition (shifts between generalist and woodland species) was related to shrub cover in tree plantings of both woodlots and ecological restoration.

The above studies are all examples of tests of correlation. In what is perhaps the only active experimental manipulation of shrub layer, Loyn *et al.* (2008) initiated an experiment in which shrub layers were planted within a *Eucalyptus* plantation. After four years, growth was limited, and results not particularly strong. There was, however, a marginal increase in forest species, including specific feeding groups in plantations which were planted with a shrub layer. In young eucalypt plantations Loyn *et al.* (2008) found that shrub-foraging bird species were not those that were missing from plantations, suggesting that young eucalypts were acting as a shrub layer strata.

In a study of windbreaks, Kinross (2004) found that they could support significant numbers of woodland bird species due to complexity and diversity of the shrub layer.

While it has been suggested that shrub layers may be detrimental to reptile assemblages (Driscoll 2004), Jellinek *et al.* (2014) demonstrated a positive effect of shrub layer cover and reptile abundance.

Shrubs are likely to be shorter-lived than an overstorey tree crop, so it would be ideal if the established shrub layer in agroforestry plots was self-sustaining and able to successfully reproduce and re-establish over the entire life of the tree crop. Selective weed control may contribute to this management goal.

'… a significant relationship was found between the abundance of ground and low-shrub insectivores and total leaf cover and number of branchlets to 2 m height …'

PHOTO: BOB HINGSTON

THE OVERSTOREY LAYER

The overstorey layer represents two important habitats: the first of which is the canopy, which is an important source of nectar, an important feeding and shelter habitat for insects and, consequently, an important foraging habitat for a range of birds which specialise on this stratum. The other habitat in this stratum is the trunks of trees, which also represent a habitat for insects, reptiles, and specialised foraging birds. An important conclusion which emerges from this section of the review is that the habitat value of trees improves greatly with age. Within this structural layer we also include older, remnant trees, which often include unique structural features, most notably tree hollows.

Some agroforestry systems are primarily formed from an overstorey layer (timber and oil mallees), and best support biota which depends upon this layer (for example, nectivorous and foliage gleaning birds) and enhancement could best be achieved by increasing the diversity of canopy species, while other systems which lack a canopy layer (for example, saltbush, or brushwood) tend to support a different biota (for example, ground-foraging birds) and could be best enhanced by the initial introduction of canopy species.

Barrett (2000) demonstrated that bird species richness was related to the level of tree cover, tree species, and the level of natural tree regeneration. Bird diversity, including the diversity of hollow-nesting species, increased with the density of old trees. Kavanagh *et al.* (2007) found high species richness and high similarity with natural woodland bird assemblages, and suggested that this was due to the fact that the reafforested patches were made up of a high diversity of trees and shrubs.

Many canopy species (e.g. *Eucalyptus* spp.) provide particular types of bark microhabitats, which may be missing in homogenous plantations of species such as *E. globulus*. As a consequence, Loyn *et al.* (2008) found that bark-foraging species, such as tree creepers, were missing from eucalypt plantations and would require the addition of a more diverse range of tree species.

In terms of tree health, Montague-Drake *et al.* (2009) found a decline in site occupancy of bird species of conservation significance with increased dieback in wheatbelt vegetation remnants of New South Wales and, according to Shaun Molloy (pers. comm.), brushtail possums in southern Western Australia avoid habitat patches containing dieback.

Tree age is an important habitat variable, and something difficult for agroforestry plots to overcome unless remnant old trees are incorporated into the design. The Loyn *et al.* (2008) study of 58 eucalyptus plantation sites demonstrated that forest and woodland birds responded favourably to the number of retained trees and number of hollow-bearing dead trees. Munro *et al.* (2011) found that species composition (shifts between generalist and woodland species) was related to the number of dead trees, plant species and the largest tree size. Loyn *et al.* (2008) recorded fewer possums and gliders in young plantations, likely due to the lack of hollow-bearing trees. Large hollow-dependent bird species (e.g. cockatoos, galahs) were not as affected as they forage over large ranges and can use hollows found in other parts of the landscape (Loyn *et al.* 2008). Loyn *et al.* (2008) suggested that the absence of bird species, such as white throated tree creeper and spotted pardalote, from otherwise successful ecological plantings was due to the absence of old eucalyptus trees with hollows and textured bark. In the Western Australian Wheatbelt, Short *et al.* (2009) demonstrated that pygmy possums were able to use oil mallees as a foraging resource, provided that an older eucalypt tree containing a nesting hollow was in reasonable proximity. Law *et al.* (2011) found that bat activity was directly related to the density of remnant trees in the landscape and activity in eucalypt plantations was particularly low, due to the absence of older trees containing tree hollows.

Density of large trees has also been related to the abundance (Brown *et al.* 2011; temperate woodland remnants) and species diversity (Cunningham *et al.* 2007; revegetation) of reptiles.

Trees also provide substrata for mistletoe which is an important source of nectar and fruit (Watson 2001). Birds, insects and mammals all utilise mistletoe either for food or habitat. Montague-Drake *et al.* (2009) and Lindenmayer (2010) were able to relate bird species diversity to mistletoe density in remnants and revegetation respectively.

Provision of nesting boxes within agroforestry plots may be an alternative solution to the challenge of incorporating old hollow-dwelling trees into plots. Loyn *et al.* (2009a) estimated that plantation eucalypts (sugar gum) took between 60 and 80 years to form tree hollows.

PLOT SIZE AND SHAPE

The species–area relationship is one of the best known patterns or "rules" of conservation science (Lomolino 2000). Originally established for oceanic islands, it is a consistent observation that larger islands support more species (e.g. MacArthur and Wilson 1967). This was soon applied to the question of conservation reserves and habitat fragments, in which the general rule tends to apply, but with a wide range of caveats and potentially modifying pressures, such as the quality of the habitat matrix. This principle has more recently been applied to the development of revegetation and could be applied to agroforestry systems.

The literature relating remnant size to species richness is vast, and not necessarily as relevant as studies relating revegetation or agroforestry area to species diversity. However, a few studies which are mentioned elsewhere in this review, and also refer to within-site variables, are worth describing. Montague-Drake *et al.* (2009) studying woodland remnants found some regionally-declining bird species responded positively to plot size, while Barrett (2000) found that bird species richness was significantly lower in vegetation remnants smaller than 10 ha.

In terms of revegetation, Kavanagh *et al.* (2007) demonstrated an effect of patch size for diverse eucalypt revegetation with patches <5 ha having fewer bird individuals and species than patches 5 to 20 ha, while Munro *et al.* (2011) found that bird species richness was positively related to the area of revegetation in a range from 0.07 to 10.6 ha in area. Selwood *et al.* (2009) found no effect of revegetation area on bird breeding activity with the largest patch surveyed at 54 ha.

Patch or remnant shape is considered to be potentially important due to the fact that as a patch shape shifts from circular (or more likely square), to rectangular or linear, the edge to interior ratio increases. This is broadly considered to be unfavourable, with the edge being a region of altered microclimate, and the region of invasion and interference from external factors (e.g. predation from feral predators) (Riess *et al.* 2004). For these reasons, there are species that are considered to prefer "core" habitats, with a natural buffer between their favoured habitat and less favoured environments (e.g. an annual crop). Small or irregularly shaped patches will not contain sufficient area of such "core" habitat.

In terms of the possible effects of patch shape, in a study of windbreaks, Kinross (2004) found that they could support more woodland bird species if the windbreaks were wider than 15 m. Barrett (2000) also found that broader strips of vegetation had more species than strips that were less than 50 m in width. However, remnant shape, including comparisons with linear strips, did not affect reptile diversity and abundance (Jellinek *et al.* 2014). Loyn *et al.* (2008) also found no response of forest and woodland birds to eucalyptus plantation area, or shape. In terms of insects, Cunningham *et al.* (2005) also found no clear difference in species composition between edge and interior communities in Blue gum plantations, suggesting that shape (that is, edge to interior ratio) didn't strongly affect species composition. Overall, the evidence so far tends to suggest that narrow linear strips may be limited in habitat value, but patch size is likely to be more important when considering the habitat value of other patches. This would suggest that belt and alley farming may be limited in habitat due to its strip-like shape, with the width of only several trees. Likewise, small patches may individually be less favourable. However, it is important to consider landscape context as small patches of appropriate habitat can play significant roles in connecting fragmented landscapes.

ADJACENCY AND LANDSCAPE CONTEXT

An emerging area of conservation science is the consideration of landscape context. The concept that species conserved within a plot are not only dependent upon factors within that plot but are also a product of the nature of the surrounding landscape. This is most clearly understood in terms of an extremely isolated habitat patch, which is unlikely to be colonised by nearby sources of biota. Colonisation by forest-dependent fauna is far more likely if the surrounding landscape supports such species.

This is clearly near-impossible to manipulate retrospectively but could assist in guiding initial establishment of an agroforestry plot. These principles also suggest that maintenance of highly variegated landscape, including the maintenance of older, remnant trees wherever possible, is likely to be beneficial for species that may be targets of conservation within agroforestry plots.

Landscape context is often quantified using remote sensing, and includes measures such as the proportion of various land use classes (for example, forest cover, grassland) within various distance classes (e.g. 100 m to 5 km).

Kavanagh *et al.* (2007) found that bird species composition in mixed eucalypt plantings was significantly influenced by the amount of remnant vegetation within 500 m of the plot. Compositional change was principally due to the favouring of woodland-dependent species such as the Olive-backed Oriole and Fuscous Honeyeater. Montague-Drake *et al.* (2009) found similar effects for bird species of conservation concern in woodland remnants of NSW.

These effects, however, appear to be far from universal, although the approach to this question tends to vary amongst studies. Hobbs *et al.* (2003) recorded little difference in bird numbers between eucalypt plantation that was adjacent or distant (200 m to 600 m) from remnant forests. However, bat activity was higher in plantations adjacent to remnant forests.

Munro *et al.* (2011) found no effect of the proportion of native vegetation found within 2.5 km of the plot, however, Lindenmayer *et al.* (2010) found an effect of the amount of area planted with native vegetation within 500 m, and estimated with each doubling of area that two more bird species were present in the plot (the level of remnant vegetation increased bird species by 0.75 species when doubled). In terms of actual breeding activity, Selwood *et al.* (2009) found that bird breeding activity in revegetation wasn't clearly related to the proportion of native vegetation in the surrounding landscape or distance to the nearest remnant. Loyn *et al.* (2007, 2008) found no effect of landscape context on birds of eucalypt plantations.

In terms of insects, Cunningham *et al.* (2005) found no clear difference between insect assemblages in blue gum plantations surrounded by open agriculture and those adjacent to remnant vegetation.

Most studies found stronger effects of within-habitat variables when compared to the effect of landscape context. Variation in the reported influence of landscape context may highlight the difficulty of summarising the complex nature of landscapes with simple metrics, such as distance to the nearest remnant woodland, potentially masking quite complex patterns.

MANAGEMENT

Agroforestry systems incorporate complementary land uses that need to be managed in a manner compatible with forestry practices and the promotion of natural habitats.

Grazing is often likely to be incorporated into agroforestry systems. Heavy grazing is likely to reduce herb and shrub cover and litter cover (Yates *et al.* 2000, Barrett *et al.* 2008) and suppress regeneration of trees and shrubs (Barrett 2000). This reduction in ground cover and vegetation diversity then has carryover consequences for fauna such as birds (described above). Grazing in temperate remnant woodlands is also associated with higher weed densities (Prober and Thiele 1995, Yates *et al.* 2000). With respect to the management of woodland remnants, grazing exclusion through the establishment of fences is one of the main management tools (Prober *et al.* 2011). However complete exclusion through fencing is likely to be impractical and inappropriate in most agroforestry systems so moderate, well-timed grazing is recommended. For instance, Wheatbelt NRM (2013) state that “Forage shrubs can be grazed for short periods (4 to 6 weeks or until 80% defoliation)”. Sheep exclusion is then required to allow the plants to regenerate.

Weed cover can be associated with the abundance of less-desirable, generalist bird species (Munro *et al.* 2011) and is also likely to support a less-desirable insect assemblage (Tallamy 2004). Weed cover also leads to competition and suppression of native plant species. Although weed control is generally desirable, there is little evidence to suggest that a herb layer dominated by weeds would be less beneficial to biodiversity than open, bare soil left as a consequence of weed control.

Vertebrate animals such as kangaroos and rabbits can hinder the establishment of an agroforestry system through herbivory, and also later shelter in such systems, as can other vertebrate pests such as foxes and feral cats. Several studies record the presence of introduced vertebrates in plantation or agroforestry systems (e.g. Smith unpublished data, Hobbs *et al.* 2003), with numbers similar to other habitats. Agroforestry systems should be incorporated into any property vertebrate management plan.

Fire is another risk to assets which should be appropriately managed through a property plan, which incorporates agroforestry, and the application of firebreaks, water points and appropriate access. Fire, if applied skilfully, may also have a role in promoting native plant regeneration and habitat heterogeneity.

RECOMMENDATIONS FOR SPECIFIC AGROFORESTRY SYSTEMS

This section relates the above findings to the most likely Western Australian agroforestry systems. Recommendations are made specific to each of these agroforestry systems.

All agroforestry systems

- Incorporate remnant trees wherever possible. This is probably the most difficult habitat element to re-introduce once lost.
- Maintain litter layers, and other ground level material, such as rocks and logs. Consider introducing these elements from areas of previous clearing or from plantations recently harvested.
- Maintain native herb and grass layer.
- Adjacency to remnants is favourable if possible.
- Large areas of plantings are probably of more benefit.
- Manage grazing appropriately.

Shrub based systems: saltbush and brushwood

- Introduce element of canopy (e.g. scattered eucalypts).
- Encourage diversity of herbs under shrubs.
- Consider incorporation of other native shrub species in order to improve diversity.

Tree based systems: oil mallee, timber plantation

- Incorporate a wider diversity of canopy species.
- Incorporate a diverse shrub layer.
- Belts should be as wide as possible (preferably > 15 m).

RATING THE BIODIVERSITY VALUES OF AGROFORESTRY SYSTEMS

CURRENT SCORING SYSTEMS

Several biodiversity-focussed scoring systems have been developed to score plots with foci ranging from plantation forestry (e.g. Cawsey and Freudenberger 2008) to woodland remnants (Gibbons and Freudenberger 2006).

Habitat Hectares was developed by the Victorian Department of Natural Resources and Environment (Parkes *et al.* 2003) in order to quantify the habitat quality and condition of woodland remnants. It was used by Munro *et al.* (2009) to quantify the condition of revegetation plots and Smith (2009b) used the system to compare habitat condition of oil mallee, restoration plantings and remnant woodland. In both cases the approach was able to discriminate between habitat types. Habitat Hectares works by assessing the condition of each stratum, and considering landscape context (both Munro *et al.* (2009) and Smith (2009b) only used the condition component of the score). A requirement of Habitat Hectares are that parameters are compared to a near-pristine baseline (usually woodland) with all parameters (for example, shrub cover) expressed as a proportion of that found in the baseline site(s), before being converted to a score.

A simpler habitat complexity score was developed by Coops and Catling (1997), which is an additive score based on the level of cover within each stratum. This score does not require comparison with a baseline data set and has proven useful when correlated with remotely sensed estimates of habitat complexity. Munro *et al.* (2009) and Smith (2009b) both employed the system of Coops and Catling (1997) as a complement to that of Parkes *et al.* (2003). Gove (2012) used most of the parameters described in Parkes *et al.* (2003), but instead of comparing with a benchmark, scored the parameters much like the system of Coops and Catling (1997) with a range of categorised scores. Additionally Gove (2012) used a 2 m pole to count the number of contacts with vegetation, and a measure of vegetation complexity. This was then converted to a score which fed into the total scoring system. This composite habitat score was correlated with the diversity of woodland birds (Gove 2012).

The Biodiversity Benefits Index score (Oliver and Parkes 2003) is a similar additive index that is somewhat more focussed on the consequence of land use change. The system includes vegetation condition and landscape context components, which are built on those developed in the Habitat Hectares approach. It also features a “conservation significance” component which takes into account the current representation of the assessed biodiversity unit in the current landscape. These three elements are combined to derive a “biodiversity significance score”. Biometric is a tool similar in design to that of Habitat Hectares in which condition scores are compared to established benchmarks (Gibbons *et al.* 2005).

Gondwana Link (2013) has developed a set of restoration standards which amount to a 100 point score which is then converted to a star system. The system incorporates several of the components described in the Enhancement of Agroforestry section of this report and featured in scoring systems described above (structural diversity and the enhancement of microhabitats such as litter and logs), but also includes broader strategic components (e.g. consideration of off-site effects, implementation of a monitoring plan). Scores are not compared to an established baseline, and much of the scoring can be established at the planning stage.

Cawsey and Freudenberger (2008) established a Plantation Biodiversity Benefits Score, which is an additive system incorporating many of the habitat elements described above, such as the maintenance of a diversity of species and the retention of habitat structures such as logs, rocks and remnant trees. It also includes several components specific to silviculture, such as thinning and pruning and rotation times.

A PROPOSED SCORING SYSTEM

The proposed habitat value scoring system needs to be applied at the planning stage, rather than scoring current condition as has previously been applied to revegetation and agroforestry (e.g. Munro *et al.* 2009, Smith 2009b, Gove 2012). This allows the system to be used when assessing the relative private versus public benefits of proposed projects for receiving public funding.

The proposed scoring system is similar to many other systems, such as Habitat Hectares, as it is an additive scoring system based on the level of representation of a range of habitat elements, using individual strata (Table 1) with a maximum of 100 points. Habitat Hectares, for instance, is scored against a benchmark ideal natural community, and the parameter values observed there. The proposed scoring system is also similar to the system applied by Gondwana Link in that it does not refer directly to a benchmark, but aims for what is perceived to be the ideal set of conditions. In this case, the ideal is somewhat similar to high quality revegetation, but accounts for the limitations in agroforestry systems (for example, absence of understorey) and aims for what could be considered to be the ideal novel habitat (Perring *et al.* 2013) with all appropriate habitat parameters in place, but not necessarily mimicking any particular natural habitat. This also eliminates the challenge of selecting an appropriate benchmark in a region such as the Western Australian Wheatbelt, which has a high diversity of differing native woodland systems (Harvey and Keighery 2012). Scores are based on what is known from the literature but there is often little information (e.g. how much better is three rocks than two?), so best approximations have been made to the degrees of improvement with each increment.

Scoring of a novel habitat for biodiversity value benefits from not needing to be compared with a near pristine benchmark site. However, comparisons to a native habitat are possible, simply by running the same scoring system for a sample of native habitat. The challenge in not comparing to a benchmark is where density of an element (e.g. logs) could be compared at an arbitrary spatial scale (e.g. per hectare).

An important caveat is that this is a desktop exercise leading directly from a review of relevant literature. It should therefore, undoubtedly, undergo peer review, field testing and refinement before being adopted as standard practice.

Table 1: Basis of the agroforestry habitat value scoring system.

	Component	Maximum value (%)
Site condition	Remnant large trees	10
	Tree (canopy) cover	5
	Tree diversity	5
	Understorey (shrub) cover	10
	Understorey diversity	10
	Herb and grass cover	10
	Litter layer	10
	Rocks	5
	Logs	10
Landscape context	Patch size	10
	Context	5
	Shape	5
Management		5
	Total	100

Note: Eighty per cent of the scoring is weighted towards site condition measures, including management, while landscape context including patch size represents 20% of the total score.

Remnant trees

The presence of remnant trees, particularly those with tree hollows, can ensure the presence of particular taxa which would otherwise not be present (see 'The overstorey layer'). Due to their unique habitat characteristics, and irreplaceability, the integration of remnant trees is weighted relatively highly. Tree densities are similar to those proposed by Cawsey and Freudenberger (2008). Three extra points are allocated if any of the trees have significant hollows, making a total of 10 points (Table 2).

Table 2: Remnant tree density within or immediately adjacent agroforestry site.

Remnant trees/ha	Score
None	0
1	3
2–5	5
>5	7
At least 1 tree with hollows	+3

Tree cover

Tree cover is directly related to the utility of a range of fauna. Canopy species such as eucalypts provide food sources in the form of nectar but also support many insects that are then foraged by birds and mammals. Two components of tree cover are included: actual cover values and tree species diversity.

Diversity is important in order to provide a range of microhabitats and food sources which have a temporal turnover in availability, ensuring food is available throughout the year. Species richness is quite straightforward to score in the planning stage, based on the number of species that are planted.

Tree cover is more difficult to predict and stipulate in any meaningful way at the planning stage. However, given the planned configuration in terms of belt width and inter-row spacing, an estimate of tree cover should be possible (Table 3). In native Western Australian wheatbelt woodlands the canopy cover is, according to Smith (2009), on average 17% and ranging from 1 to 32% in this particular study. Given the challenge in predicting absolute canopy cover, an alternative simplified set of rules is also provided, based on the connectedness of canopy cover.

The tree species diversity (Table 4) assessment does not currently consider whether a plant species is local or not, because this Review has shown that species that are not locally indigenous do provide habitat for a range of species. However, there is clearly some merit in the establishment of locally indigenous plant species, and it is possible to incorporate this component by reducing the number of points allocated for species number, and reallocating these points to a criterion stipulating locally or regionally indigenous species. Federal government guidelines for "biodiverse plantings" require two overstorey species which are indigenous to the area (Commonwealth of Australia 2011).

Table 3: Two metrics for scoring anticipated tree cover.

Tree cover	% cover	Canopy connectedness	Score
	None	None	0
	1–5%	Scattered, no canopy interconnectedness	2
	6–20%	Patchy, some interconnectedness of canopy	4
	>20%	Nearly complete or complete, interconnectedness of canopy (in at least one direction – e.g. row)	5

Table 4: Tree species diversity scoring.

Tree diversity	No. of Species	Score
	0	0
	1	1
	2–3	2
	4–5	4
	>5	5

Note: Species diversity is summed for the entire plot

Shrub layer

The shrub layer is potentially the most important habitat stratum in terms of addressing known species declines (e.g. small insectivorous birds) in the Wheatbelt (see 'The understorey layer'). The proposed scoring system for the shrub layer is similar to that of the canopy layer, considering both total cover (Table 5) and species diversity (Table 6); however the scoring is weighted more highly. Federal government guidelines for "biodiverse plantings" require three mid-storey species which are indigenous to the area (Commonwealth of Australia 2011). Again, potential shrub cover will require estimation based on the plot design and the proposed density of planting.

Table 5: Understorey (shrub) cover scoring.

Shrub cover	% cover	Canopy connectedness	Score
	None	None	0
	1–5%	Scattered, no canopy interconnectedness	3
	6–20%	Patchy – some interconnectedness of canopy	5
	>20%	Nearly complete or complete interconnectedness of canopy (in at least one direction e.g. row)	10

Table 6: Understorey species diversity scoring, based on the entire agroforestry plot.

Shrub diversity	No. of Species	Score
	None	0
	1	3
	1–3	5
	4–10	7
	>10	10

Ground layer

Materials which accumulate at ground level form important microhabitats for both animals and plants, and form important refuge sites. Following the format of this chapter, ground layer habitat is divided into: a herb and grass layer (Table 7), a litter layer (Table 8), logs (Table 9) and rocks (Table 10).

Like previous vegetation layers, although stipulating specific levels of cover is favourable, this is difficult to predict at the planning stage. Hence an alternative "aspirational" set of criteria are also provided. Weed control is also concluded as a component of management (Table 14). A specific number of species is not stipulated although Federal Government guidelines for "biodiverse plantings" for shrubby woodlands require two understorey species which are indigenous to the area (Commonwealth of Australia 2011). Future litter layers are difficult to predict, and hence a score is based on anticipated management of litter. It is possible to select for particular shrub and tree species in order to encourage a litter layer (Sanders 2009). Litter is defined in this context as being plant residues and coarse woody debris.

Logs are an important habitat element which provides habitat for invertebrates and small vertebrates, in particular reptiles. Insectivorous bark foraging bird species are also encouraged by the presence of logs. Logs also form important microhabitats for plant germination. Log quantification is based on the method employed in Habitat Hectares in which total length of logs >100 millimetres (mm) diameter is quantified. Rocks provide microhabitats in some respects similar to logs, but may be less beneficial for birds. Given their importance to invertebrates and reptiles, they receive a similar weighting to logs. Rocks are arbitrarily described as any rock with at least two dimensions >200 mm in length.

Table 7: Herb and grass layer scoring system.

Herb and grass cover	% cover	Ground vegetation management	Score
	None	None	0
	0–10%	Indigenous herb and grass layer development encouraged	5
	10–30%	Indigenous herb and/or grass layer planted by seed	7
	>30%	Indigenous herb and/or grass layer planted by tubestock, including species difficult to propagate	10

Table 8: Litter layer scoring system.

Litter cover	% cover	Litter management	Score
	None	Litter not maintained	0
	0–10%	Litter maintained	5
	10–20%	Litter actively maintained	6
	20–50%	Litter producing plants or provided for (thinning etc.)	7
	>50%	Plants introduced in order to gain highly diverse litter layer	10

Table 9: Scoring scheme for logs density.

Log density	Metres logs per ha	Score
	<1.0	0
	1.0–5.00	1
	5.01–10.00	3
	10.01–20.00	6
	>20	10

Note: Logs are defined as >100 mm in diameter and 1 m length (all other smaller material is considered litter).

Table 10: Scoring scheme for rocks.

Rocks	Rocks per ha	Score
	0	0
	1–5	3
	6–15	6
	>15	10

Note: Rocks are defined as any non-organic material with at least two dimensions >200mm.

Landscape context

While within-site habitat variables may be most influential on biodiversity, the broader context of this habitat may have some influence on habitat usage. Patch size may influence colonisation rates and determine the amount of favourable core habitat available to desirable species. Patch area scores are based on the understanding that habitat quality improves exponentially, with largest increases occurring at lower patch areas and categories are similar to those employed by Parkes *et al.* (2003) for native remnants (Table 11). As there appears to be 'spillover' effects of adjacent woodland, three points are allocated to plots adjacent to native woodland. The measure of native vegetation within a 5 km radius is a simplified version of that employed in Parkes *et al.* (2003) in which the 5 km radius was also employed (Table 12). Shape is also of consequence, particularly the contrast between a long linear structure and a square or rectangular shape. There is also evidence that increasing the width of narrow linear habitats can be particularly effective. Therefore, narrow belts are penalised most dramatically (Table 13).

Table 11: Scoring scheme for patch size.

Patch size	Area (ha)	Score
	<1	0
	1–3	2
	3–5	4
	6–10	5
	11–20	6
	>20	7
	Adjacent remnant woodland	+3

Table 12: Scoring scheme for landscape context.

Native vegetation	% cover	Score
	0	0
	1–10	2
	10–20	3
	20–50	4
	>50	5

Note: % native vegetation within 5 km radius.

Table 13: Scoring scheme for patch shape.

Shape	Shape	Score
	Belt <20 m wide	0
	Belt >20 m wide	2
	Block	5

Management

To allow biodiversity to continue utilising the plantation, only staged harvesting regimes should be used. This would mean harvesting in a mosaic pattern leaving corridors of vegetation connecting unharvested areas. This will continue to allow the terrestrial and arboreal species to move around the plantation. If a clear fell harvest regime is used, the long term biodiversity values are largely negated and this scoring system should not be used.

For other management practices, a score is allocated. The management practices include grazing intensity, fire, weeds and pest animal management (Table 14). Several of these management practices should also manifest themselves in the form of improved vegetation structure and diversity, and increased ground cover. This topic could be expanded by specifying particular parameters for management intensity, such as specific levels of grazing, weed cover and frequency of control and specific burning or fire control practices.

Table 14: Scoring scheme for management practices.

Management	Management	Score
	Grazing intensity managed appropriately	2
	Grazing and one other pressure managed (weeds, fire, vertebrate pests)	3
	Grazing and two other pressures managed	4
	Grazing and three other pressures managed	5

CONCLUSION

Agroforestry systems appropriate for the WA Wheatbelt generally improve habitat value beyond that of normal pasture or annual crops. In some cases, species known to be declining in the WA Wheatbelt are supported within these systems. This is most evident for bird species for which regional declines are well documented. There is less evidence to support the assumption that remnant-dependent invertebrates and reptiles are supported by agroforestry systems. Data for mammals are also scant, although there is some evidence that mammals will forage within agroforestry systems provided appropriate resources are available. Agroforestry systems tend to support fewer plant species than remnant indigenous vegetation, and as a consequence they support fewer native animal species.

Species absences can often be associated with particular habitat elements which may also be absent in agroforestry systems. Contrasts in species assemblages and species absences may be addressed, to some degree, by appropriate habitat manipulations. Little has been documented in terms of habitat enhancement of agroforestry, however, evidence from studies of revegetation have demonstrated that appropriate manipulations, such as the integration of multiple strata and species diversity (e.g. Loyn *et al.* 2009a and Munro *et al.* 2009), can dramatically increase the habitat value of such systems. These manipulated production systems will be novel systems, not directly mimicking any particular natural system.

In order to encourage strategic decision-making which will ultimately require consideration of a trade-off between biodiversity benefits and financial cost, Land Managers require a scoring system, which quantifies the potential habitat value of each agroforestry system and associated enhancements. The scoring system proposed here is an additive system which considers the addition of each stratum as a means of increasing habitat complexity. Species diversity of each major stratum is also considered and, to a lesser extent, patch size and broader landscape context. As this scoring system was developed as a desktop exercise, stemming directly from a literature review, the system will benefit from field trials, and modifications following discussions with those likely to employ the tool.

This report has highlighted a range of approaches in which the biodiversity values of Wheatbelt agroforestry systems may be optimised, with a potentially limited impact on their productivity. However, there is further work required to quantify this trade-off.

‘Agroforestry systems appropriate for the WA Wheatbelt generally improve habitat value beyond that of normal pasture or annual crops…’

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Within the Western Australian Wheatbelt, agroforestry plantations are producing timber, firewood, sandalwood, brushwood, biomass, carbon and forage. These systems are currently being used to revegetate degraded land or poorly performing paddocks and improve the overall whole-of-farm profitability. Agroforestry is also likely to play an important role in maintaining the biodiversity values of the Wheatbelt, where some regions have lost more than 95% of the original vegetation. This guide highlights how we can maximise the biodiversity values from agroforestry systems to support the conservation of this global biodiversity hotspot.

This report looks at how land managers can incorporate biodiversity into their production based systems, by making small changes in the planning stages. This report reviews the current scientific literature available describing how animals use or benefit from agroforestry plantations and discusses how plantations may be manipulated to improve the biodiversity outcomes while maintaining the production value within the plantation.

A new scoring system is proposed that will allow land managers, community groups or industry bodies to quantify the biodiversity value provided by changes to the plantation design and implementation, including the use of remnant trees, overall tree, shrub and ground layer characteristics, landscape context and on-going site management practices. This can be used to consider the financial costs and potential biodiversity value trade-offs so that the best possible systems can be designed that support Wheatbelt agricultural and ecological values.

Australian Government
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wheatbelt natural resource management