Notes and Abstracts

Full titles of proceedings abstracted in this issue

Hobbs R. J. and Yates C. J., eds (2000) **Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Management and Restoration.** Surrey Beatty & Sons, Chipping Norton. (Published in book form and available from Surrey Beatty & Sons, 43 Ricard Road, Chipping Norton, NSW 2170, Australia)

McDonnell M. J. and Williams N. S. G., eds (2000) **Directions in Revegetation and Regeneration in Victoria.** Proceedings of a forum held at Greening Australia, Heidelberg, Victoria, 5–6 March 1999. Australian Research Centre for Urban Ecology Occasional Publication Number 2, Royal Botanic Gardens, Melbourne. (Available from the Royal Botanic Gardens Shop, Birdwood Avenue, South Yarra, Vic. 3141, Australia and the Department of Natural Resources and Environment Book Store, 8 Nicholson St., East Melbourne, Vic. 3002, Australia)

Robertson A. and Watts R. eds, (1999) **Preserving Rural Australia: Issues and Solutions.** CSIRO Publishing, Collingwood. (Published in book form and available from P.O. Box 1139 Collingwood Vic. 3066, Australia. Email: sales@publis.csiro.au)

Waters C. M., ed. (2000) **Proceedings of the First STIPA Native Grasses Association Conference.** Mudgee, New South Wales, 16–17 March 2000. NSW Agriculture, Trangie. (Available from Darryl Cluff, STIPA Native Grasses Association, P.O. Box 18, Coolah NSW 2843, Australia. Email: stipa@coolahddg.com.au)

RAINFOREST

1.6

Rainforest restoration on a larger scale?

David Lamb (CRC Tropical Rainforest Ecology and Management, Botany Department, University of Queensland, Brisbane, Qld 4072, Australia. Email: d.lamb@botany.uq.edu.au)

Key words: *direct seeding, landscape reintegration, rainforest.*

A number of trials are currently underway attempting to restore rainforests at cleared sites in Queensland and New South Wales. The early results at many sites are promising but most of these efforts have been costly and have covered relatively small areas. A key question therefore is how to restore biodiversity over much larger areas. The question is relevant in Australia but is even more relevant in other parts of the world's humid tropical forest regions where biodiversity levels are plummeting because of land degradation.

Direct sowing. One method might be to use direct seeding rather than planting seedlings. Most rainforest restoration efforts to date have relied on planting seedlings of as large a range of native species as possible. By contrast, direct seeding has been commonly used for restoration in temperate regions and in many minesite rehabilitation programs. It is a useful way of reducing costs and, hence, allowing larger areas to be treated. It bypasses the need to raise large numbers of seedlings in nurseries and reduces field planting expenses. One of the drawbacks of direct seeding, however, is that only a small proportion of the seed applied actually yields seedlings so that large numbers of seeds of each species are usually required. As well, it requires a high standard of weed control which, on minesites is most easily carried out immediately after mining ceases when sites are weed free. Both factors are serious limitations in most wet tropical forest situations away from minesites. Seed is usually difficult to acquire for all but a handful of species and seed of most of the later successional species is only viable for a short time. Further, most degraded tropical forest sites have a large population of weeds, including grasses. Direct seeding will fail unless these are controlled.

Notwithstanding these problems, there are signs that direct seeding may have a role to play in some rainforest rehabilitation programs. Preliminary trials by the Queensland Forest Research Institute and the Queensland Parks and Wildlife Service on the Atherton Tablelands have yielded promising early results and further studies are now being undertaken by the

CRC for Tropical Rainforest Ecology and Management. Current approaches involve herbicide application to remove weeds followed by the sowing of pioneer rainforest species such as White Ash (*Alphitonia petriei*), Bleeding Heart (*Omalanthus novoguineensis*) or Brown Salwood (*Acacia aulacocarpa*). Being pioneers, the seed of these species is relatively easily collected and remains viable for some time. The density of seeds applied is adjusted to ensure that canopy closure is rapid and occurs within the first year. This increases the likelihood that weeds, especially grasses, can be competitively excluded from the site. Site biodiversity can then be enhanced by undersowing seeds or planting seedlings of species from later successional stages beneath this short-lived canopy, enhancing those that may be dispersed beneath the canopy by birds. Since competition on the forest floor is substantially reduced the site is also now more easily colonized by animal-dispersed seed from any nearby intact forest. Together, these several processes can contribute to an accelerated recovery of much plant biodiversity. Current research is examining the reasons for differences in the success rate of direct seeding using pioneer species at different sites. Research is also underway to assess which species from later successional stages might be established using direct seeding beneath these initial canopies or other simple canopies created by planted pioneer seedlings.

Plantation forestry. A second approach to restoring biodiversity over larger areas is to combine reforestation to enhance biodiversity restoration with land-use practices that might be financially attractive to landowners. That is, to find ways of making rehabilitation an attractive investment rather than simply a socially beneficial cost for individual landowners. One of these ways might be timber plantations. Until now, most tropical forest plantation programs have involved monocultures and these have almost invariably utilized fast-growing exotic species. Within Australia most tropical and subtropical plantations have used *Pinus caribaea* or *P. elliottii*. Outside Australia *Pinus*, *Eucalyptus* and *Acacia* have dominated many tropical plantation programs. There is no doubt that these species have a valuable role to play, especially in some of the more severely degraded sites overseas. But there has been a tendency to regard these species as being the only option for landscape rehabilitation. What is needed is a more diverse range of alternatives including plantation systems that yield a financial return plus a degree of biodiversity.

An unusual situation is evolving in parts of Queensland and New South Wales formerly occupied by rainforest. Many landowners in these areas have become interested in reforesting part of their land but have not been enthusiastic about using pines to do so. Many have expressed interest in planting eucalypts but, now that rainforest logging has ceased, some have also seen a market opportunity to grow several of the former high-value cabinet timbers in plantations. The problem is that the time periods involved are long and the financial returns are uncertain. At the same time, however, a significant number of these farmers are also interested in reforestation for 'conservation' purposes. A system that combined long-term financial returns and short-term conservation benefits would be attractive to many landowners and could significantly increase the rate of reforestation of cleared land.

There are many ways this might be achieved including the use of species mosaics (i.e. a patchwork of monocultures), enhanced understorey development beneath monocultures and mixed species plantings (Keenan *et al*. 1997; Lamb 1998). Mixed species plantings are an intuitively attractive possibility because they automatically yield an increase in plant diversity as well as producing timber. There is also the theoretical possibility of increasing the overall productivity by improved nutrition (e.g. by including nitrogen fixers in the mixture), reduced insect or disease damage (e.g. because target species are dispersed among other species and hidden in space) and improved utilization of light or soil resources (e.g. because species with spatially complementary root or crown architectures are mixed). The combination of biodiversity benefits and these potential biological gains prompted the adoption of mixed species plantations when the Community Rainforest Reforestation Program was established in north Queensland to create a new timber resource after the cessation of rainforest logging in 1988.

This decision to use mixtures was not without risks. Only small areas of mixed species plantations have been established in the past despite the theoretical benefits. The reason is that random assemblages of timber species do not necessarily yield productive mixtures. For this reason the CRC for Tropical Rainforest Ecology and Management has sought, first, to identify complementary species and desirable attributes; and, second, to establish how much commercial timber production is lost as increasing levels of plant species richness are incorporated into plantings.

Current field trials established by the CRC are exploring these several problems using plantings in which rows of a particular species are interplanted with rows of another and the growth of each species in the mixture is compared with its growth in monoculture controls. The four species being used have a variety of canopy architectures, leaf morphologies and growth rates. The trials are replicated in three locations including the tropical lowlands and the Atherton Tablelands. Another trial involving 16 species involves more intimate mixtures and is examining growth responses when more than one other species is involved in the mixture. Other studies have examined staggered planting times so that certain species (e.g. Red Cedar; *Toona ciliata*) grow up under the canopy of 'nurse' trees. Results suggest this approach avoids much of the insect damage that normally occurs when Red Cedar is planted in the open (e.g. Keenan *et al*. 1995).

Both direct seeding and species-rich timber plantations involve some disadvantages. It is likely that direct sowing will only be appropriate for a limited number of species so that significant biodiversity increases will depend on seedling enrichment or colonization from nearby intact forest and so may take some time to develop. On the other hand, the technique, if successful, offers the prospect of initiating restoration over much larger areas than are currently being tackled. Timber plantation systems that seek to achieve production and biodiversity on the one landscape also involve trade-offs. The more tree species involved, the more likely it is that commercial timber productivity will decline. On the other hand, higher tree diversity is likely to increase the structural complexity of a plantation and, hence, make a site more attractive to wildlife. Where the trade-off is and how much of a sacrifice needs to be made by 'timber' interests or 'conservation' interests is yet to be resolved. What is important, however, is that we develop reforestation options that fit somewhere between 'pure' ecological restoration involving the planting of seedlings of many species and industrial plantations involving only a single species. This is necessary if we are to widen the alternatives available to those wanting to enhance tropical landscape biodiversity on a larger scale than has occurred to date. Both direct sowing and new forms of plantation forestry are likely to have a role in this process.

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1.7

Regenerating degraded rainforest with the help of abandoned Brush Turkey nests.

Hans Westermann (Ford Street, Elands, NSW 2429, Australia. Email: hwesterm@tsn.cc)

Key words: *megapodes, rainforest, soil seed banks.*

At the Wingham Brush, a restored remnant of lowland subtropical rainforest near Taree, New South Wales, we undertook a trial to compare the potential use of seed banks from two spent Brush Turkey mounds (turkey compost) as a method to rehabilitate degraded rainforest. A successful pilot with turkey compost (during which 12 native species were established in a 2 m × 1 m area 17 months after spreading the material) prompted the experiments.

The Australian Brush Turkey (*Alectura lathami* J.E. Gray 1831) is a large mound-building megapode. It is common in the humid rainforests of eastern Australia and in places has adapted to human-dominated environments (Jones & Everding 1991). During the breeding season which starts in May or June, the male birds construct large mounds of litter. This takes from 3 to 6 weeks (Jones *et al.* 1995). Mounds measured in the Wingham Brush average 6.1 m long × 5.2 m wide × 1.1 m high. Temperature within the heap may first rise to well over 40°C before eventually stabilizing at around 33°C (Jones 1988a). The mounds which may contain between 0 and 58 eggs (Jones & Everding 1991) are tended for up to 8 months (Jones *et al.* 1995).

A census of mounds in the Wingham Brush in August 1997 showed that out of 28 nest sites, seven had fresh mounds which were actively tended. By early January, the Brush Turkeys usually abandon their mounds and only rarely are there any active mounds between early January and May. This accords with observations by Jones (1988a). Mound site selection is thought to be largely based on the suitability of site conditions including availability of suitable leaf litter, protection from drying, and protection from pirating by a neighbouring male (Jones 1988b). Brush Turkeys may use the same nest sites on and off for several years (Jones 1988b; Jones *et al.* 1995). The role of residual humus in the Brush Turkey's return process has not yet been studied experimentally.

Method. Two spent Brush Turkey nests were sampled. Both were in very shady positions. When collecting turkey compost for the trials care was taken to sample material from all sides of the nest. This maximized the area from which nest material, and therefore potential seed, was gathered. In order to obtain current season compost as well as to avoid material that might have been heated up excessively, only the top 50 cm was collected for the trials. This compost was 6 months old or younger. The trial plots were set out in an area which, until 1993, had been open wasteland serving as an access and storage facility for the Wingham sewerage works. The area now grows scattered young rainforest trees as well as Blackwood (*Acacia melanoxylon*) as nurse trees.

A total of six 2 m × 1 m plots were established on two trial sites (A and B) in February 1999. Site A comprises four plots including two controls. Site B has two treated plots, B1 and B2. Details of treatments for all the plots, including preliminary weed control by slashing or the removal of the top 10 cm of soil (scalping), are provided in Table 1. Site A received sunlight for most of the day and Site B received about 2 h of morning sunlight. Site B plots were overshadowed by several 5 m tall Blackwoods. During the trial, rainfall was plentiful and evenly distributed. There were no extreme temperatures recorded; however, some seedling loss occurred during a few hot days.

Table 1. Plots and treatments for Sites A and B, Wingham Brush Turkey compost trial

	Plot	Treatment
SITE A	A1	Scalped and covered with 7–8 cm of compost from Mound 1
	A2	Scalped and covered with 7–8 cm of compost from Mound 2
	Cs	Slashed only (control)
	Cr	Scalped and left bare (control)
SITE B	B1	Scalped and covered with 7–8 cm of compost from Mound 1
	B2	Scalped and covered with 7–8 cm of compost from Mound 2

The plots were inspected at least once a week and seedlings were counted after 8, 21, 30, 60 and 90 days. After 21 days, plot A2 carried a dense cover of Tobacco Bush (*Solanum mauritianum*) seedlings, swamping other plants. Consequently, the trial was modified after 1 month and Tobacco Bush was removed from half of plot A2, with no disturbance to natives. Some disturbance occurred early in the experiment and probably had negligible effects. Native seedlings were identified to at least genus level and, in most cases, species level. In the counting process, each individual seedling of a species was flagged with a bamboo skewer. When all seedlings were marked the skewers were removed and counted. This improved accuracy and double counting was avoided.

Results. The species and densities of native rainforest seedlings germinating in all plots are listed in Table 2. The first native seedlings appeared in all compost-covered plots after 3 weeks. White Cedar (*Melia azedarach* var. *australasica*) and Rough-leaved Elm (*Aphananthe philippinensis*) appeared simultaneously and in large numbers. Ground cover plants including Aneilema (*Aneilema biflorum*) and Basket Grass (*Oplismenus aemulus*) quickly covered any bare ground. No native seedlings were found in the unweeded half of Plot A2 or in the two control plots. The slashed control plot (Cs) continued to have 100% cover of weeds, while the scalped control plot (Cr) had 80% weed cover and 20% bare ground. Some weeds also appeared in the compost-covered sites. Weeds most commonly occurring in the sites were Nut Grass (*Cyperus rotundus*), Paddy's Lucerne (*Sida rhombifolia*) and Tobacco Bush.

Many seedlings died in the early stages due to heat stress. Rough-leaved Elm and White Cedar were the most susceptible. Many Round-leaf Vine (*Legnephora moorei*) seedlings were eaten off as soon as the true leaves had appeared. Most natives in the shady trial plots (B1 and B2) had disappeared by the end of the trial.

Discussion. Both composts clearly contained numerous viable seeds of rainforest plants as well as weeds. The emerged native seedlings encompass a range of successional phases comprising fast-growing pioneers including Tobacco Bush, Poison Peach *(Trema aspera)*; nomads Giant Stinger (*Dendrocnide excelsa*), Creek Sandpaper Fig (*Ficus coronata*) and Koda (*Ehretia acuminata*); and longer lived trees (White Cedar and Rough-leaved Elm) as well as shrubs, ground cover and vines. Growth rates in general seemed to compare favourably with natural regeneration.

The two composts appeared to contain quite different species. Compost 1 has large numbers of White Cedar and Round-leaf Vine, both of which are poorly represented in Compost 2. However, Compost 2 contains many Giant Stingers and Creek Sandpaper Fig. The former is represented with only two specimens and the latter is absent in Compost 1. Koda, Poison Peach and *Nyssanthes diffusa* are represented by a few specimens in Compost 1 but are absent in Compost 2.

The differences of species and number in the plots appear to reflect the origin of the two compost samples. Compost from Mound 1 was taken from a rather species-poor area where flying-foxes have not roosted for many years, potentially reducing seed inputs. Compost from Mound 2, however, originates from a site which is a habitual roosting spot for Grey-headed Flying-foxes (*Pteropus poliocephalus*). It had been severely disturbed in 1995 by an influx of thousands of Little Red Flying-foxes (*Pteropus scapulatus*). The large number of Tobacco Bush seeds in the soil was most likely brought in by the flying-foxes which have Tobacco Bush fruits in their diet (Eby 1995).

Table 2. Densities of all native rainforest species found in all **six** plots, 3 months after treatment at Wingham Brush

Species	Site A				Site B	
	A1 (Compost 1)	A2 (Compost 2)	Cs (Control slashed)	Cr (Control scalped)	B1 (Compost 1)	B2 (Compost 2)
Trees						
Aphananthe philippinensis	13	10	0	0	1	1
Dendrocnide excelsa	2	19	0	0	0	7
Ehretia acuminata	2	0	0	0	2	0
Melia azedarach var. *australasica*	93	5	0	0	0	0
Ficus coronata	0	9	0	0	2	0
Shrubs						
Nyssanthes diffusa	1	0	0	0	1	0
Trema aspera	7	0	0	0	0	0
Herbs						
Aneilema biflorum	17	11	0	0	1	0
Oplismenus aemulus	0	2	0	0	0	0
Vines						
Legnephora moorei	28	1	0	0	9	0
Maclura cochinchinensis	0	0	0	0	1	0
Pandorea pandorana	0	0	0	0	1	0

Note: Seedlings were found only in half the area of plot A2, the weeded half.

The plots in the shaded area had considerably fewer germinations and a high mortality rate. This could be attributed to increased shade as summer came to an end, or due to the surrounding Blackwoods whose copious leaf drop may have had an adverse impact. Hopkins (1990) noted that Blackwood may, at least temporarily, prevent later phase species from establishing.

Scalping the soil to a depth of 10 cm was very useful in reducing weed germination but did not completely remove the need for follow-up weed control. This may have been due to either deeply buried weed propagules or the introduction of weeds within the compost. Weed seed imported in the compost may explain the lack of natives in the unweeded half of plot A2, which may have produced higher germination levels had the entire plot been weeded.

Since compost was taken from only a small proportion of the mound, we assume that our trial caused minimal disturbance to the Brush Turkey breeding behaviour. Damage to the vegetation community is also probably minimal, given that the mounds are located in very shady spots where seed germination is unlikely to occur.

Conclusion. The trials have shown that sufficient viable seed was available in the samples of spent Brush Turkey nests to allow them to be used to advantage in our project. Less than 20% of the mound was removed in case future research shows residual humus to be important to Brush Turkey nest site selection. We found, however, that compost must be distributed when growing conditions are optimal or where irrigation can be provided. Also, care must be taken to not spread serious environmental weeds such as Trad (*Tradescantia fluminensis*) and Madeira Vine (*Anredera cordifolia*). While the range of species that germinated in our compost trials was limited, it included a number of growth forms (trees, shrubs, vines and ground cover) belonging to several successional stages and from a seed source whose provenance is compatible with the site. Apart from introducing seeds, soil introduction may have the additional advantage of importing fungal spores and other microorganisms.

Jones and Everding (1991) noted that Brush Turkeys have adapted to human-dominated environments and in the Brisbane area between 40 and 100 mounds are reported each year. This suggests that the above technique may be useful elsewhere.

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GRASSLAND AND GRASSY UNDERSTOREY

2.6

FROM: Waters C. M., ed. (2000) **Proceedings of the First STIPA Native Grasses Association Conference.** 16–17 March 2000, Mudgee, New South Wales. NSW Agriculture, Trangie.

2.6.1

The response of native pastures to planned cell grazing on 'Lana'. Pp 96–100, Tim and Karen Wright ('Lana', Uralla, NSW 2645, Australia) *Key words: grassland, native pasture, nature conservation, socio-economic issues and solutions, sustainable agriculture.*

The Wrights provide some hard data to demonstrate the economic and land management benefits of an integrated 'planned cell grazing' program applied to the management of native pasture on 'Lana' in the Northern Tablelands, New South Wales. In the early 1990s, the 34 000 ha property was subdivided from 36 paddocks to 215, and a vegetation and soils monitoring program was commenced by the University of New England. Four main cell areas were formed, each containing about 48–50 small paddocks of an average size of about 15–18 ha. Mobs of sheep were amalgamated to larger mobs, as were cattle herds — enabling more intensive, even grazing of smaller areas; with longer rest areas between grazing events. Stock is moved every 2–3 days and each paddock is spelled for 3–4 months depending on the season. (A whistle is used to move stock to an adjacent paddock so minimal labour is required.) Grain and hay have not been fed for the last 9 years; carrying capacity has increased by an average 30% depending on season; and costs have reduced 25–30% due to reduced labour, machinery and animal health costs. Monitoring has found that higher ground cover has been maintained during extremely dry conditions and that infiltration capacity has increased. Available phosphorus and calcium have doubled after 2 years, possibly due to increased soil biological activity — and soil strength has decreased, enabling greater root penetration and recovery. In terms of native grasses, the desirable natives have increased relative to the conventionally grazed controls while the relatively undesirable native pasture species have declined markedly. *Glycine* and *Desmodium* have been enhanced — and Purple Donkey Orchids (*Diuris* spp.), Chocolate and Vanilla Lilies (*Dichopogon* spp.) and Copper-wire daisies (*Podolepis* spp.) have been observed in the cell grazed paddocks, at times in great abundance. The grazing sensitive perennial Scented-top Grass (*Capillipedium* sp.) has increased markedly.

2.6.2

Harvesting native grasses: innovation and development in the Central West. Pp 113–121, Andrew Briggs,[1] Ian Cole[2] and Alan Nicholson.[1] ([1]Department of Land and Water Conservation, Wellington, NSW 2820, Australia; [2]Department of Land and Water Conservation, Cowra, NSW 2794, Australia) *Key words: grassland, native pasture, nature conservation, seed harvesting, sustainable agriculture.*

While there is a huge interest in what native grasses have to offer in terms of sustainable grazing, habitat restoration or aesthetics; harvesting sufficient high-quality native grass seed has, until recently, been a slow and laborious task. The Native Grasses Innovation and Development Program reports that, in New South Wales' Central West, it has improved the accessibility of native grass harvesting equipment and has improved the expertise necessary to harvest high-quality seed regularly and reliably from natural stands. The paper evaluates two harvesters extensively modified over the project period and provides operational specifications, adaptability and suitability details, along with some price and manufacturer details. Advice on improved methodologies learned from the project is also provided, including methods for reducing weed contamination in the stand in the seasons leading up to seed harvest — as well as some useful data on local seasons of harvest for individual species. A number of practical pointers for site monitoring prior to harvest are given, plus recommendations for post-harvest drying, labelling and storage. The paper concludes with the challenge 'it is now up to the community, the seed industry and various other potential users of native grass seeds to take up and make use of this technology, information and support base'.

2.7

FROM: Hobbs R. J. and Yates C. J., eds. (2000) **Temperate Eucalypt Woodlands in Australia: Biology, Conservation, Management and Restoration.** Surrey Beatty & Sons, Chipping Norton.

2.7.1

Recreating eucalypt woodland with a grassy understorey on a gold mine in the Central Tablelands of New South Wales. Pp 298–317, Donna M. Windsor,[1] AnneMarie Clements,[2] Mark B. Nolan[3] and Heath Sandercock.[4] ([1]Environmental Studies Unit, Charles Sturt University, Bathurst, NSW 2795, Australia; [2]Anne Clements and Associates, Environmental and Botanical Consultants to Climax Mining Limited, P.O. Box 1623, North Sydney, NSW 2059, Australia; [3]Climax Gold Mine, Mandurama, NSW 2797, Australia; [4]General Manager Operations, Climax Mining Limited, Goldfields House, Sydney, NSW 2000, Australia) *Key words: direct seeding, grassy understorey, minesite rehabilitation, woodland.*

Post-mining rehabilitation at the Junction Reefs mining lease near Blayney in the New South Wales Central Tablelands involves the rehabilitation of 42 ha of mining-affected areas as well as the enhancement of the adjoining 50 ha of remnant woodlands. This chapter focuses on methods and results involved in constructing woodland with grassy understorey on the overburden material, contoured to resemble natural undulating topography. In particular, it provides detail on the progress of re-establishing a sward of native grasses characteristic of the original vegetation of the Tablelands. Methods and results of trials of a number of different 'rates' of seed-bearing mulch of Kangaroo Grass (*Themeda australis*) are summarised, showing that intermediate intensities were more resource efficient at this site. A roller brush harvester was also used to collect a range of a further eight species of grasses (collected from a 20 km radius), allowing the grass seed to be mixed with tree and shrub seed for direct sowing by tractor-drawn sowing implement. Vegetation establishment has been satisfactory to date, with higher establishment on weed-free overburden compared to sites where topsoils were used, showing the importance of weed control. After 4 years since revegetation began, sustainable eucalypt woodland with a grassy understorey appears to be developing. Lists of weed and native species (including their densities at reference sites and treated sites) are provided in the paper.

2.7.2

Description, distribution, conservation status and weed management of box and buloke grassy woodlands in South Australia. Pp 167–189, Richard J.-P. Davies (Black Hill Flora Centre, Botanic Gardens of Adelaide, Maryvale Road, Athelstone, SA 5076, Australia. Current address: School of Biological Sciences, Flinders University of South Australia, GPO Box 2100, Adelaide, SA 5001, Australia) *Key words: disturbance manipulations, grassy understorey, weed control, woodland.*

This chapter discusses early descriptive surveys of the woodlands formerly widespread on the heavy soil plains of South Australia; and also examines more recent quantitative studies which provide more detail of the understorey component. Most woodland types are poorly conserved, with threats including ongoing (permitted) vegetation clearance, weed invasions and ecologically inappropriate revegetation activities. While commencing with an overview of vegetation communities, the second half of this chapter contains substantial, practical information about the competitive attributes of a range of individual exotic grasses and forbs (information needed if an ecologically informed approach to reducing weeds and increasing natives is to be successful). Corresponding information is provided for native species, providing a basis for the development of site-specific strategies for weed control based on applied disturbance regimes to reduce or increase competitive advantages of particular species.

2.8

FROM: McDonnell M. J. and Williams N. S. G., eds (2000) **Directions in Revegetation and Regeneration in Victoria.** Proceedings of a forum held at Greening Australia, 5–6 March 1999, Heidelberg, Victoria. Australian Research Centre for Urban Ecology, Occasional Publication Number 2, Royal Botanic Gardens, Melbourne.

2.8.1

Riparian revegetation in the urban environment: the Merri Creek experience. Pp 15–20, Brian Bainbridge and Judy Bush (Merri Creek Management Committee Inc., 2 Lee Street, Brunswick East, Vic. 3057, Australia) *Key words: community involvement, fire, grassland and grassy understoreys, native forbs, planting, weed control.*

This chapter describes the evolving approaches to revegetation by the Merri Creek Parkland Management Team over the past 10 years; where the original high density tree and shrub revegetation approach has gradually shifted towards restoration and management of the remaining remnant vegetation, particularly native grasslands. Revegetation works also shifted to the planting of understorey grasses and forbs in patches of maturing overstorey plantings in sites originally planted with trees and shrubs, in recognition of the original nature of the Basalt Plains landscape among other reasons. Grassy weed control has become an increasingly important part of the maintenance regimes, especially in remnant sites. Emphasis on plant identification skills of staff and recognition of the threat posed by exotic *Nassella* species in particular has led to this becoming a prime concern. Simple techniques for ensuring methodical weed control spraying have been developed over the years. Projects combining regeneration burns and woody and grassy weed control are now planned and carefully timed, with successful burns requiring a high level of communication in advance with urban neighbours.

2.8.2

Success in hindsight: managing La Trobe University Wildlife Reserves. Pp 21–25, Adrian Daniell (Wildlife Reserve, La Trobe University, Bundoora, Vic. 3083, Australia) *Key words: grassy understorey, native forbs, weed control.*

The La Trobe University Wildlife Reserves (containing 100 ha of remnant and restored River Red Gum Woodland and wetlands) have been recognized as having State-level conservation significance. The early approach to revegetation was based on trial-and-error planting of overstorey species. Initial plantings were made into dense exotic pastures with a history of grazing and fertilizer application. The lack of suitable herbicides and other weed control technologies meant that this 'plant and see' approach was the best option available at the time. In the 30 years since, this technique has proved very successful: exotic understorey species are declining and native grasses, in particular *Microlaena stipoides* and *Danthonia* spp., herbs and orchids are re-establishing. *Atriplex semibaccata, Acaena agnipila, Brachyscome diversifolia, Chrysocephalum semipapposum, Coprosma quadrifida, Einadia nutans, Glycine clandestina, Hypericum gramineum, Leucochrysum albicans, Pultenea daphnoides, Vittadinia muelleri* and *Wahlenbergia gracilis* are some of the more obvious understorey species which are successfully recruiting from revegetation sites. The Slender Sun Orchid (*Thelymitra pauciflora*) has spread from remnant sites into the older revegetation areas, apparently in association with this improved ground cover. While the density of the original plantings was probably too high, this has allowed the removal of some trees and, as a by-product, increased the amount of deadwood on the ground. Associated with development of the canopy was the development of a soil crust of mosses and lichens and a diverse leaf litter. Indigenous litter also appears effective in the suppression of exotic species and, concurrent with litter development, an increase in the diversity of soil invertebrates seems to have developed over 10 to 20 years.

OTHER COMMUNITIES

4.7

The return of the unwanted shrubs. Manda J. Page (School of Natural and Rural Systems Management, University of Queensland, Gatton, Qld 4343, Australia. Email: mpage@uqg.uq.edu.au)

Key words: *herbage, rangelands, regeneration, woody weeds.*

'Woody weeds' are indigenous woody shrubs that increase in density in rangelands due to a combination of climate, competition, increased grazing pressure, lack of fire and a reduction in browsing of postfire regeneration (Noble 1997). Such increases in woody plant populations have been reported in arid and semi-arid rangelands worldwide (Archer 1995). Two-thirds of south-west Queensland's mulga lands reportedly exhibits such increases in woody weeds (Queensland Department of Lands 1993). Impacts include the suppression of pasture growth leading to reduced available feed for both native and domestic animals, increased difficulty of livestock management, and increased soil erosion (Booth 1985; Burrows 1986).

Many have claimed that the control of woody weeds is vital if rangelands are to remain stable, productive and economically viable (Booth 1985). In fact, it is speculated that restoration of Australia's rangelands may not be possible without the control of woody weeds (Pressland 1984; Hodgkinson 1991).

While much research has been conducted on woody weed control methods (Burrows 1973; Scanlan & Pressland 1984; Booth 1985; Harrington 1986), there are few studies that report vegetation dynamics after the removal of woody weeds (Noble 1994). This is despite suggestions that more research is required on the response to clearing woody plant populations (Burrows 1973; Pressland & Graham 1989; Scanlan *et al.* 1991). Noble (1994) and Page *et al.* (in press) have reported that herbage production following woody weed reduction can be limited by the subsequent grazing pressure. Increased densities of woody weeds may thus result due to reduced competition from regenerating grasses and other herbage. This research note reports the regrowth of woody weed species 5 years after they were removed from two vegetation communities in south-west Queensland's mulga lands. Results are reported for three different grazing pressure regimes.

Methods and Materials. Woody weed populations in two semiarid mulga land vegetation communities were measured on Currawinya National Park (28°52′S, 144°30′E) in south-west Queensland. Botanical nomenclature follows Neldner (1992).

The vegetation communities examined were Dunefields and Mulga Sandplain (Dawson 1974). In these two communities woody weeds have increased in density, particularly since 1970 (Witt & Beeton 1995). The Dunefield site was dominated by dense stands of Hopbush (*Dodonaea viscosa* ssp. *angustissima*) and Turpentine (*Eremophila sturtii*). Low shrub species are common and most are chenopodiaceous species. The soils are predominantly red, earthy sands and the topography is low dunes intermixed with claypans (Dawson 1974). In the Dunefield system, Hopbush and Turpentine are classed as woody weeds. The Mulga Sandplain site consists of a Mulga (*Acacia aneura*) overstorey with a distinct low shrub layer of Turkey Bush (*Eremophila gilesii*). Ground cover occurs in isolated patches and Woolly Butt (*Eragrostis eriopoda*) is the most abundant species. The soils are moderately deep, sandy red earths and the relief is low with interspersed clay alluvia and small claypans (Dawson 1974). In this community, Turkey Bush is classed as a woody weed.

The experiment consisted of three grazing regimes: (a) off-park with traditional grazing pressure from domestic (mostly sheep), native and feral herbivores (b) on-park where domestic stock was removed, and (c) exclosures where no mammal grazing pressure was present. The on-park sites were still subjected to grazing pressure by native (e.g. kangaroos) and feral (e.g. goats) herbivores. Within each grazing regime, all woody weeds were removed from two randomly located 20 m × 20 m plots in January 1994 (Table 1). This included all Turkey Bush plants in the Mulga Sandplain and all Turpentine and Hopbush plants in the Dunefields. The cut and swab method was used and Garlon® (600 g/L triclopyr: Dow Elanco Australia Ltd) was the chemical applied to the cut at a rate of 1:60 using diesel distillate as the carrier. All shrub parts cut were removed from the site. Experimental and control sites were measured biannually

Table 1. The experimental design used in each of the two vegetation communities, showing the number of replicates in each site

	Woody weeds present	Woody weeds absent
Off-Park	2 (20 × 20 m)	2 (20 × 20 m)
On-Park	2 (20 × 20 m)	2 (20 × 20 m)
Exclosure	2 (20 × 20 m)	2 (20 × 20 m)

from 1994 to the present to detect herbage response (Page *et al*. in press). Due to the regrowth of woody weeds in the experimental sites, the process of woody weed removal was repeated in September 1999. All individuals removed initially (1994) and in 1999 were recorded.

Results. Table 2 presents the number of individuals removed from each site initially and in 1999. Figure 1 represents the average change in woody weed abundance between 1994 and 1999.

There was little variation in the abundance of woody weeds between the three grazing regimes prior to the initial removal (1994) and between the two plots (28–48 in Mulga Sandplain, 29–48 in Dunefields). During the following 5 years no resprouting from the cut stumps was observed, so all woody weeds removed in 1999 were assumed to be new individuals germinating. Before removing these regrowth shrubs in 1999, a noticeable increase in the shrub abundance was observed in the three grazing regimes, for both vegetation communities. Similar increases were not observed in the ungrazed control plots. In fact, from quadrat data, there were no noticeable increases in woody weed population in any of the three grazing regimes. While only two samples were available for each treatment, on average the number of woody weeds in the Sandplain Mulga off-park site increased by 13.8-fold (from 813 stems/ha to 11 075 stems/ha), on-park by 6.4-fold (750–4800 stems/ha), while in the exclosures the shrub populations increased by 2.3-fold (1075–2450 stems/ha). Similarly in the Dunefields, the shrub population off-park increased by 5.4-fold (988–5350 stems/ha), 2.6-fold on-park (788–2050 stems/ha) but decreased dramatically to a population of almost zero in the exclosure (1100–12.5 stems/ha).

Discussion. As would be expected, the removal of woody weeds makes way for new plants to germinate and establish in all grazing regimes except the ungrazed Dunefields. Interestingly, however, lower increases in woody weed recruitment were associated with lower rates of grazing. This was consistent in both vegetation types and all three species of woody weed removed. While sample size may be too small to statistically verify the trend, the off-park site had the highest grazing pressure and exhibited the densest woody weed population. This was followed by the on-park site, then the exclosure site. Importantly, the establishment of perennial grasses followed a converse pattern in that, where woody weeds were removed and grazing pressure reduced, a greater perennial grass cover was observed and less new woody weed plants established (Page *et al*. in press). This adds weight to the suggestion that

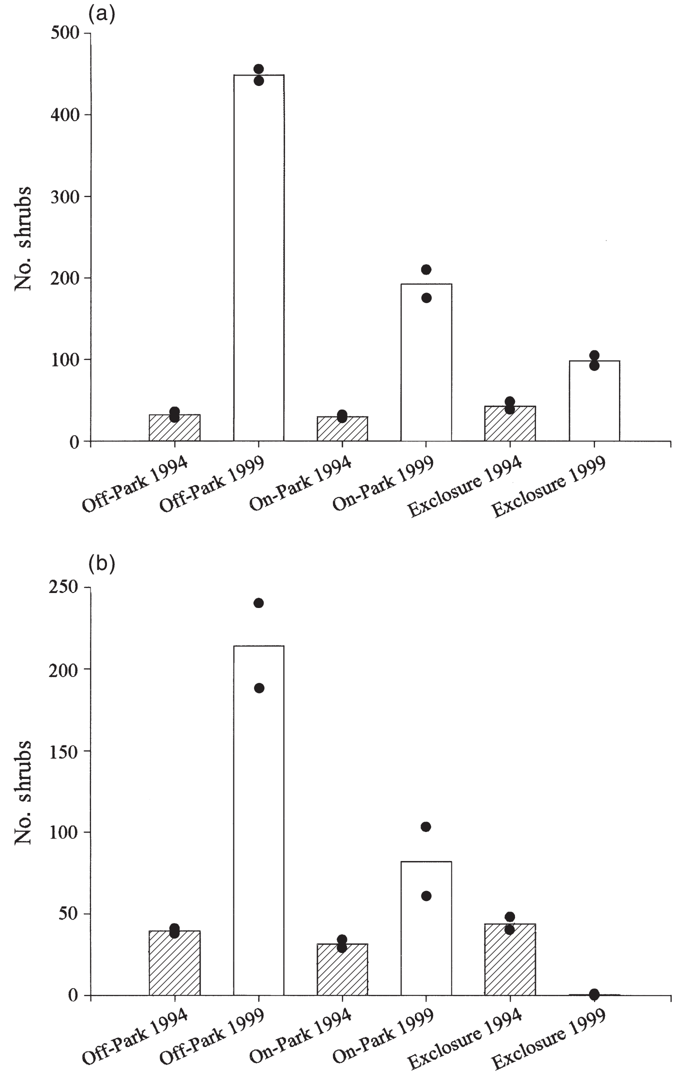

Figure 1. Average number of shrubs in 1994 and 1999 in (a) the Mulga Sandplain vegetation community and (b) the Dunefield vegetation community. Black dots represent the actual scores.

Table 2. Number of individuals removed from each grazing regime in 1994 and in 1999

	Mulga Sandplain				Dunefields			
	1994 Plot 1	1994 Plot 2	1999 Plot 1	1999 Plot 2	1994 Plot 1	1994 Plot 2	1999 Plot 1	1999 Plot 2
Off-Park	36	29	455	441	41	38	188	240
On-Park	28	32	209	175	34	29	61	103
Exclosure	38	48	92	104	48	40	0	1

woody weed recruitment may be lower in less heavily grazed sites due to competition from grass. There is obviously a sound store of woody weed seed in the seed bank that are only given the opportunity to establish when adult plants are removed and a high grazing pressure is maintained. Removing or reducing competition seems to be the key mechanism for woody weed species to germinate and establish.

Should these results be borne out at other sites in the future, they suggest that grazing pressure must also be reduced to ensure the best results. Managers must be aware that follow-up control methods will most likely be required. If grazing pressure is not also reduced or removed, an even denser population of woody weeds may be the result, thus wasting limited woody weed control resources.

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RIPARIAN AND STREAM ECOLOGY

8.1

Habitat assessment using the River Styles™ methodology.

M.P. Taylor, J.R. Thomson, K. Fryirs, G.J. Brierley (Macquarie University, Department of Physical Geography, North Ryde, NSW 2109, Australia. Email: mataylor@laurel.ocs.mq.edu.au)

Key words: *fauna and habitat, planning assessment and monitoring, riparian.*

Assessments of available habitats along river courses, and how these are affected by variability in river discharge, are required for setting environmental flow allocations. If abstraction or storage of water is planned or ongoing within a catchment it is essential to understand the sustainable thresholds for maintaining functional habitat diversity so that the effects of management practices on ecological processes can be predicted. Traditional methods of predicting changes in habitat availability as a result of changes in flow regime assume fixed-bed morphology, whereas, in practice, geomorphic processes of sediment deposition and transport are tightly linked to discharge fluctuations (Carling 1995). Therefore, habitat availability is unlikely to be a simple function of wetted area. A system of habitat assessment that is more closely linked to geomorphological processes, while maintaining ecological relevance, may help to overcome some of these shortcomings.

Numerous studies have investigated the morpho-dynamics of modern and historical river systems but rarely has geomorphological research successfully linked the temporal and spatial variations in stream power and sediment flux to ecological condition. As McEwen *et al.* (1997) suggest '. . . there is a need for an integrated and well-structured assessment scheme which both accurately records and interprets morphological characteristics, and reinforces the interrelationships between geomorphology and ecology in river system evaluation.' Geomorphological processes operating at a range of spatio-temporal scales determine the physical structure of a river system, providing the physical template on which ecological processes occur (Frissell *et al.* 1986). Changes to the geomorphic structure and stability of rivers will inevitably affect their ecological functioning, for example by influencing the availability of instream and floodplain habitats, the structure and function of riparian vegetation, and the production and transport of nutrients and organic matter (Brierley 1999). The principle aim of the research presently being undertaken by the Rivers Group coordinated by Dr Gary Brierley at Macquarie University is to examine the relationship between River Styles™ and the ecological structure of fluvial systems within an Australian context. The project will initially focus on analysing intact or relatively undisturbed River Styles™ across coastal New South Wales.

Relevance of River Styles™ in habitat assessment

The significance of habitat surveys or assessments has become increasingly evident in river management agencies. Habitat surveys are conducted for four principal reasons: (i) as part of river health assessments (Harper *et al.* 1995; Maddock 1999); (ii) in the assessment of conservation value (Harper *et al.* 1992); (iii) in design of river rehabilitation programmes (Kemp *et al.* 1999); and (iv) for prediction of the effects of flow regime changes (Carling 1995). The underlying philosophy in each case is that physical habitats provide the template for biological processes, and that a greater diversity of habitats will support greater diversity of aquatic species.

Among the perceived benefits of habitat surveys are that they: (i) link the ecology and geomorphology of fluvial systems (Harper & Everard 1998); (ii) overcome the taxonomic bias inherent in purely biological monitoring programmes (Harper & Everard 1998); and (iii) can be rapidly and cheaply conducted, allowing many sites to be assessed for conservation potential (Harper *et al.* 1992). If habitat assessments are to link ecology and geomorphology, then the physical units measured must also be geomorphologically meaningful and relevant to the scale of assessment being undertaken. To overcome taxonomic bias, habitat surveys should be multi-scalar, as different taxa live and move over different scales. Ideally, habitats outside of the wetted channel should also be considered (floodplains, bars, riparian zones). Decisions about what physical structures (habitats) should be enhanced or recreated at a particular site need to be based on an understanding of wider temporal and spatial aspects that determine geomorphological processes at that site, as well as on an understanding of the habitat requirements of biota. Many attempts to restore habitat have failed because physical structures have been placed in inappropriate places within the catchment (Harper *et al.* 1999).

Irrespective of the management purpose, it is imperative that habitat assessments are based on ecologically meaningful features, and not simply on physically distinct features that are arbitrarily determined to be 'habitats' on the basis of appearance. A system classification or assessment must be based on habitat units that can be demonstrated to be important to the structure and function of ecological communities. Similarly, the geomorphological processes responsible for creating habitat units must be understood. Determining the relationships between the geomorphic structure of fluvial systems, their morpho-dynamics and the assemblage of aquatic biota is essential for understanding the spatial and temporal impacts of anthropogenic disturbance to these systems.

The River Styles™ methodology has been developed by Brierley and Fryirs (2000) as a means of determining the geomorphic character, behaviour and the physical condition of a river with respect to its location within a catchment. The geomorphic structure of a river system is determined by channel geometry, planform and by the assemblage of geomorphic units such as different bar types, pools, riffles, inset benches, backswamps, floodplains configurations etc. Through the assessment of geomorphology at individual reach-sites, fluvial geomorphologists such as Rosgen (1994, 1996) and Frissell *et al.* (1986), have been able to develop river classification schemes using nested hierarchical methodologies. However, while the River Styles™ methodology also uses a nested hierarchical technique, its principal advantage over other methodologies is that it is based on the assemblage of geomorphic units found within a reach. Genetic groupings of geomorphic units are used to identify each river style, from which interpretations of river behaviour and ultimately condition can be derived. Unlike Rosgen (1994, 1996) it does not simply predict river behaviour from the appearance of channel zone attributes in a prescriptive manner (Miller & Ritter 1996). The River Styles™ methodology can be used to establish the long-term trajectory of river behaviour only after the current form is set with a long-term evolutionary context (using interpretations of channel and floodplain geomorphic units). Once the evolutionary pattern is established, the relative current condition can then be assessed by looking at the behaviour of the reach within its catchment context (i.e. by evaluating the linkages with other adjacent reaches). Armed with this information, a reasonable prediction of the long-term trajectory of river evolution can then be attempted.

The River Styles™ is an open-ended, generic framework that has, to date, been extensively applied along coastal rivers in New South Wales. River behaviour and character are analysed at four interconnected scales: catchments, landscape units, river styles and geomorphic units (table 1 in Brierley & Fryirs 2000). A catchment is subdivided into distinct landscape units (e.g. uplands, escarpment, rounded foothills and lowland plain). These provide the background geomorphological setting and the catchment boundary conditions under which distinct river styles operate. Inset within these landscape units are an open-ended number of river styles, reflecting distinct patterns of geomorphic units and river behaviour. Assemblages of geomorphic units have been shown to be associated with particular river styles (table 2, Brierley & Fryirs 2000).

Because individual geomorphic units are physically distinctive, they are likely to support different biota. Furthermore, because biota have different habitat requirements at different life stages (e.g. invertebrates may live in fast-flowing water as juveniles, require emergent rocks for hatching or egg laying, and riparian vegetation as adults), or under different environmental conditions (refugia), the associations between geomorphic units may be important determinants of species composition and distribution within a reach. As particular river styles have characteristic assemblages of geomorphic units, it is contended that they may support characteristic biotic assemblages. This will be particularly so if, as seems likely, geomorphic units of a given type (e.g. riffles) are physically more similar within river styles than between styles (e.g. riffles in headwater streams are different to riffles in gorge streams).

The biotic assemblages of a geomorphic unit will be partly determined by the degree of physical heterogeneity of that unit.

Small-scale patches of particular substrate or hydraulic character may form discrete habitat units, or microhabitats, supporting relatively distinct assemblages. Organisms may exploit a variety of such patches at different life stages or for different resource requirements, so that the associations between microhabitat patches will be important determinants of species composition both at the microhabitat and geomorphic unit scale (and therefore at higher spatial scales).

Thus, the biotic assemblages at each spatial scale will be influenced by the physical heterogeneity at higher (e.g. associations of geomorphic units) and lower (e.g. microhabitat diversity) spatial scales. The links between spatial scales will be reinforced by trophic webs, because organisms are affected by food and presence of predators that may move over different spatial scales. Thus a hierarchical classification scheme may be the most appropriate framework in which to describe and assess physical habitat, especially if it is linked to the geomorphic processes that create the physical heterogeneity at those scales.

This research will explore the relationships between river styles, geomorphic units, microhabitats and biotic assemblages at each scale. Individual river reaches will be determined using the River Styles™ methodology outlined above (see also Brierley & Fryirs 2000). At selected river reaches, detailed mapping will be undertaken to determine the type, size and frequency of geomorphic units present, and to describe the range of physical and hydraulic heterogeneities of microhabitats within them. Figure 1 shows an example of how the procedure will use a nested hierarchical approach for each site. Following completion of the field mapping, the biotic assemblage of each geomorphic unit and associated microhabitats will be sampled. Biotic sampling will initially focus on instream geomorphic units, although the dominant taxa and nature of the riparian vegetation will be described. Aquatic biota to be sampled will include macroinvertebrates, fish, and macrophytes. Microbial samples may also be collected.

Outcomes. The biophysical data collected during this research project will be used to determine the ecological significance of the geomorphic features identified using the River Styles™ methodology. This will reveal whether the physical units identified for different River Styles™, at various spatial scales, support distinct biotic assemblages. If they do, then the River Styles™ approach will provide an important research and management tool, enabling habitat quantity and quality to be rapidly assessed within an ecologically and geomorphologically significant context. This method may potentially enhance river health assessment methods (Harper *et al.* 1992; Maddock 1999), facilitate setting of environmental flow allocations (Carling 1995; Harper & Everard 1998), and aid the design and evaluation of river restoration programmes (Harper *et al.* 1995; Kemp *et al.* 1999). A geo-ecologically meaningful classification scheme would also be useful for selecting appropriate control sites for environmental impact assessments (e.g. determining reference conditions) and other ecological studies, where natural between-reach variation in species composition and abundance needs to be minimized.

In summary, the ultimate aim of this project is to establish meaningful geo-ecological units, such that contingency management planning for future effects of land use changes or flow manipulations on habitat availability can be predicted more accurately.

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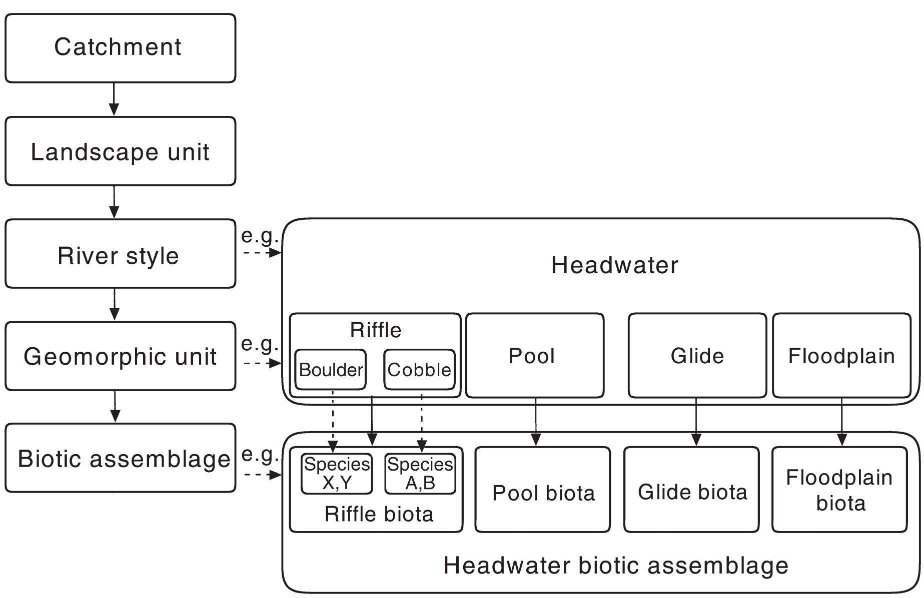

Figure 1. A graphical representation of the nested hierarchical approach that will be used to establish the links between river style, geomorphic units and biotic assemblage.

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8.2

FROM: Robertson A. and Watts R., eds (1999) **Preserving Rural Australia: Issues and Solutions.** CSIRO Publishing, Collingwood.

8.2.1

Challenges for conserving biodiversity in Australian freshwater ecosystems. Pp 33–42, Robyn J. Watts (The Johnstone Centre, School of Science and Technology, Charles Sturt University, P.O. Box 588, Wagga Wagga, NSW 2678, Australia) *Key words: education and communication, integrated ecosystem management, landscape pattern and design, research, riparian and stream ecology.*

Biodiversity in Australian riverine systems is especially vulnerable because human populations and most of our agricultural activities depend upon (and are usually located near) freshwater systems. Decline can result from a range of impacts including the fact that waste materials usually flow from the land to aquatic systems and the fact that rivers, streams and adjacent wetlands are subject to changes to river-flow regimes; introduced species; commercial and recreational harvesting; and vegetation modification. Some hard data are available on the decline of fish species in Australia (e.g. more than 99% of the fish in the Murrumbidgee catchment are the introduced noxious carp and about one-third of Australian freshwater fish species are threatened or rare, with declines occurring for many species). For species of amphibians, crustaceans, gastropods, small freshwater invertebrates and microorganisms, however, much more research is needed to quantify the extent to which the decline is occurring. Challenges for conserving biodiversity are discussed in depth in this paper and include: developing greater societal awareness of freshwater biodiversity and a collective social responsibility for its conservation (matching that given to terrestrial systems); continuing support for dialogue within established catchment groups to balance environmental benefits and sustainable development; involving private landholders more in freshwater conservation management; focusing and coordinating new research to efficiently fill knowledge gaps; and designing models specifically for freshwater systems to ensure links between fragmented freshwater habitats are given similar attention as fragmented terrestrial landscapes. A range of specific challenges and potential solutions for managers are detailed, not least the need to direct more funding towards meeting the challenges of freshwater biodiversity conservation.

8.2.2

Water and landscapes: perceptions and expectations. Pp 43–50, Kathleen H. Bowmer (Deputy Vice-Chancellor, Academic, The Executive Centre, Charles Sturt University, P.O. Box 588, Wagga Wagga, NSW 2678, Australia) *Key words: landscape arts and aesthetics, public attitudes, riparian and stream ecology.*

The community cannot be expected to recognize gradual, insidious, hidden and cascading problems such as rising saline water tables, salinization of streams, nutrient enrichment and loss of aquatic ecosystem diversity. Yet the values and expectations of the community are becoming increasingly important in developing policy in natural resource management before the changes reach critical thresholds beyond which river health may be degraded beyond repair. This chapter illustrates how human perceptions of the same landscape can be diametrically opposed. One person may see a cleared riverbank as a step backwards from its more pristine former 'naturalness', whereas another may see it as 'improved' pasture, a symbol of success and security. Rivers, however, offer powerful images and symbols which can be used to affirm perceptions of their value in a sustainable landscape. The author maintains that it is now time to plan for the future and to agree on expectations about water and landscapes: whether to accept further deleterious changes, maintain the current status, or to look for improvements in river health. In this context, myth, memory, art, advertising and the media are seen as powerful both in shaping our expectations and in providing a historical perspective on the rate and direction of change. We must, however, be careful to look to the future, to engage with ecological concepts including biodiversity, and to guard against the hidden and insidious changes which might be difficult to reverse.

PLANNING, MONITORING AND ASSESSMENT

10.1

The need for strategic revegetation planning in Australia's agricultural regions.

Brett Bryan (National Key Centre for Social Applications of GIS, Department of Geographical and Environmental Studies, University of Adelaide, Adelaide, SA 5005, Australia. Email: bbryan@gisca.adelaide.edu.au)

Key words: *integrated ecosystems management, landscape pattern and design, monitoring and assessment, planning, remnant vegetation.*

Social sustainability in agricultural regions is dependent upon the sustainability of economic processes such as agricultural productivity. Economic sustainability is, in turn, dependent upon the sustainability of ecological processes such as the cycling of energy and matter (Hobbs & Saunders 1993). One of the key factors in maintaining ecological processes is the maintenance of biological diversity at the genetic, species, ecosystem and regional levels (Noss & Cooperrider 1994). Many agricultural regions in Australia have been subject to the over-clearing of native vegetation communities, which has resulted in a significant loss of biological diversity and the reduced functioning of regional ecological processes (Hobbs & Saunders 1994). This may eventually threaten regional economic and social sustainability.

Common approaches to the conservation of biological diversity (such as reserve selection and threatened species management) cannot adequately address the continual fragmentation and degradation of remnant natural ecosystems in these agricultural regions (Saunders & Hobbs 1995). Rather, approaches to conservation must actively reverse the effects of land clearance through the revegetation and restoration of viable native ecosystems within these regions (Jordan *et al*. 1987, 1988; Hobbs & Norton 1996).

Ecological restoration involves the 'return of a degraded system to some desired state by accelerating biotic change or reintegrating successional processes' (Hobbs & Norton 1996) and revegetation is one mechanism by which ecosystems may be restored or constructed. Revegetation is expensive in terms of both economic costs and physical effort. In many regions, revegetation occurs on an ad hoc basis (Pastorok *et al*. 1997). There is an urgent requirement for strategic revegetation planning, which uses ecological science to prioritize areas and environments for revegetation in agricultural regions. In this way, the biological value of the precious revegetation effort in a given region can be maximized.

Because the science is still young, the theoretical foundations within restoration ecology that are necessary for setting revegetation priorities for the conservation of biological diversity in fragmented agricultural regions have not been well developed (Hobbs & Norton 1996). However, disciplines such as landscape ecology and conservation biology have long taken a broader landscape view of the world (MacArthur & Wilson 1967; Forman & Godron 1986; Burgman & Lindenmayer 1998). Landscape ecology has traditionally had a particularly strong role in assessment and planning for conservation in fragmented agricultural regions, particularly in terms of the spatial arrangement of habitat (Forman & Godron 1986; Forman 1995).

While other landscape-scale conservation principles such as adequacy and representativeness (NBAC 1992) have been used extensively in selecting nature reserves in less-disturbed regions (Austin & Margules 1986), they have not been used to assess fragmented agricultural regions, let alone in strategic revegetation planning. There is a considerable gap in the knowledge of the ecology of remnant ecosystems in fragmented agricultural landscapes, which has inhibited ecological and geographical priority setting for revegetation in strategic revegetation planning.

There is some urgency about the adoption of elements of strategic revegetation planning, as in many areas of the agricultural matrix it may not be too late to return much of the former biological diversity, in plants at least. Soil seed banks (Strykstra *et al*. 1998; Stocklin & Fischer 1999) and remnant populations of native plant and animal species contain invaluable stores of genetic material and can provide a basic dispersal and recolonization infrastructure for ecosystem regeneration (Grubb & Hopkins 1986; McClanahan & Wolfe 1993). However, the opportunities to tap into this resource are dwindling as the viability of seed declines, relict individuals senesce without replacement and species populations drift toward extinction.

The major benefit of strategic revegetation planning is that it can be used to coordinate and gain maximum ecological benefit from all revegetation initiatives within a region from the local landholder to major regional-scale programmes. If adopted immediately, the principles of strategic revegetation planning can minimize the further loss of biological diversity and maximize the chances of regeneration of native biological communities.

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10.2

Selecting native vegetation remnants for nature conservation on private land in north-west NSW with the assistance of four decision trees.

Bert Jenkins, Shane Andrews and Richard Morsley (Greening Australia NSW, North-west Region, P.O. Box 1467, Armidale, NSW 2350, Australia. Email: greenau@northnet.com.au)

Key words: *fencing, landscape pattern and design, monitoring & assessment, planning, remnant vegetation.*

Introduction. In the north-west region of NSW, we have developed a system for selecting native vegetation remnants for a fencing project funded by the Natural Heritage Trust. We have adopted the decision-making procedure described here to help us select remnants for protection by fencing and hope that it will also be useful to people assessing bush remnants on private land in other regions. The project specifically targets remnants that are under threat from cattle and sheep grazing in the region and provides landholders with financial assistance for fencing, thereby encouraging them to set aside an area of native bush for conservation. In addition to approving the fencing of certain remnants, we also advise landholders on the management of remnants for nature conservation. To date we have signed agreements with landholders to protect more than 3000 ha of native vegetation on more than 50 properties in the region.

The process of selecting bush remnants for the project involves two steps. We first visit each site and assess the vege–tation condition, habitat quality, floristic information and land use history of the site. We then examine the whole set of field data to identify the most suitable bush remnants for the fencing project. During a site visit, we produce a species list from the plants identified on a transverse walk across the bush remnant, recording species information for the main trees in the canopy, shrubs in the understorey and ground-cover plants. We also identify threatening 'weed' species and assign an abundance class to most species on the list, both native and exotic. This method of field assessment is a rapid technique that gives us time during the visits, to discuss with landholders the benefits of fencing bush remnants for nature conservation.

For the second step of the process, the appraisal of the information collected during site assessment, we have developed a decision model consisting of four decision trees to assist us. Members of a selection panel sit in pairs with the field assessment data and work through the decision trees before finally assigning a funding priority to each remnant. Note that the decision trees were designed for the selection of suitable bush remnants in vegetation communities with trees and not native grasslands or heaths.

The four decision trees are described below along with corresponding diagrams.

Decision tree 1: The value of land for agricultural use. The first decision tree enables us to select remnants on the basis of the agricultural value of the land on which a remnant is located (Fig. 1). On the New England Tablelands, soil derived from basic volcanic basalt has always been in great demand for various agricultural uses. Soils on sedimentary and granite parent materials, being less fertile and associated more often with greater physical and chemical limitations, are used primarily for pasture production on the mid- to upper slopes. As a result of a heavy demand for basalt soils, areas of native vegetation on basalt are uncommon in the region today and thus the

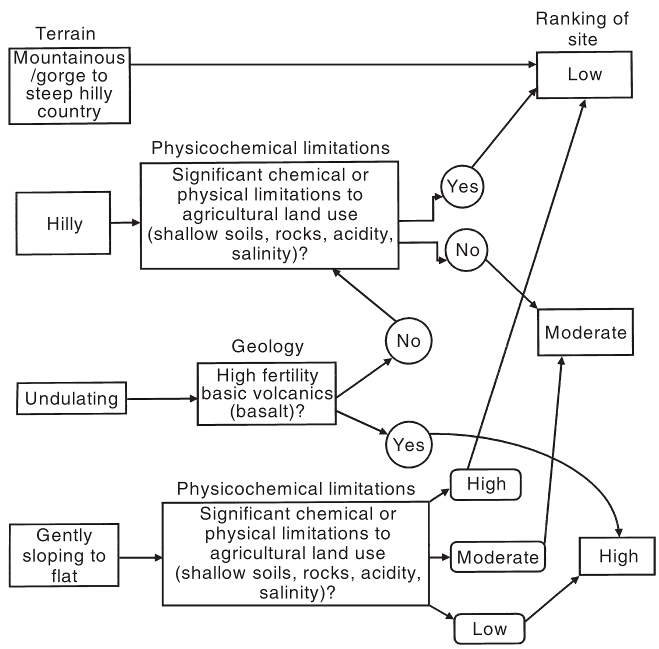

Figure 1. Decision tree 1: Value of land for agricultural use.

remaining remnants are valued highly for nature conservation.

Irrespective of soil type, flatter land is in greater demand for intensive agriculture than steeper land. Accordingly, we assign a low, moderate or high ranking to each site depending on the value of the land for agriculture. Note that we revisit these ranks again in the fourth decision tree before assigning a final funding priority.

Decision tree 2: Native vegetation condition. The second tree prompts us to examine native vegetation condition. We first scrutinize the species lists to explore threats from exotic shrubs and ground-layer species, mainly introduced herbs and grasses that have a tendency to spread or become environmental weeds (Fig. 2). Similarly, a threat from exotic trees may also be considered here if it is relevant.

We examine the health of trees in relation to symptoms of dieback and mistletoe infestation. Many unhealthy trees may suggest that the reproductive capacity of a certain species is low and potential for natural regeneration is reduced as a consequence. We examine the occurrence of natural regeneration within the remnant and inquire whether fencing would encourage new recruitment, given the current management of grazing in the area and overall tree health.

We use the data on native vegetation condition to rate the site as being either potentially satisfactory or unsatisfactory for protection by fencing. However, other factors, such as the willingness of landholders to actively manage the remnant and the presence of threatened species, a significant community or special habitat, can later alter the final funding priority for a remnant within the fourth decision tree.

Decision tree 3: Remnant viability. The third decision tree assists us in determining the long-term viability of a remnant. We use size, level of connectivity and degree of isolation as criteria to rank a remnant's long-term viability as being high, moderate or low. The decision process in the third tree continues on from that in the second tree, where scores of satisfactory or unsatisfactory are based on tree health, recruitment potential and overall vegetation condition. The condition rating carried over from the second decision tree can influence the viability rank assigned to a remnant in the third tree.

The information examined in the third decision tree may assist us in recommending whether planting a corridor with local indigenous species will enhance biodiversity or habitat quality in the remnant (Fig. 3). The viability characteristics of a remnant that are examined here will also indicate the extent to which it is under threat from adjacent land-use activities. The landholders may have to intervene by actively managing the identified threats if a small, isolated remnant is to survive in the long term. Intervention would be more important if the remnant contained a threatened species, significant community or special habitat. In the fourth decision tree, we calculate the ratio of land area to fence length, which may also modify the final funding priority by taking into consideration the shape of a remnant. Narrow remnants are possibly more vulnerable to external threats from adjacent areas and agricultural land use.

Decision tree 4: Assigning priorities for funding. The decision-making process follows on from the third decision tree through to the fourth tree, where we now examine special considerations that may be associated with a particular bush remnant. These include whether significant communities, threatened species, special habitats or riparian zones exist in the remnant (Fig. 4). The outcome of the first decision tree on agricultural land value is considered again before assigning a

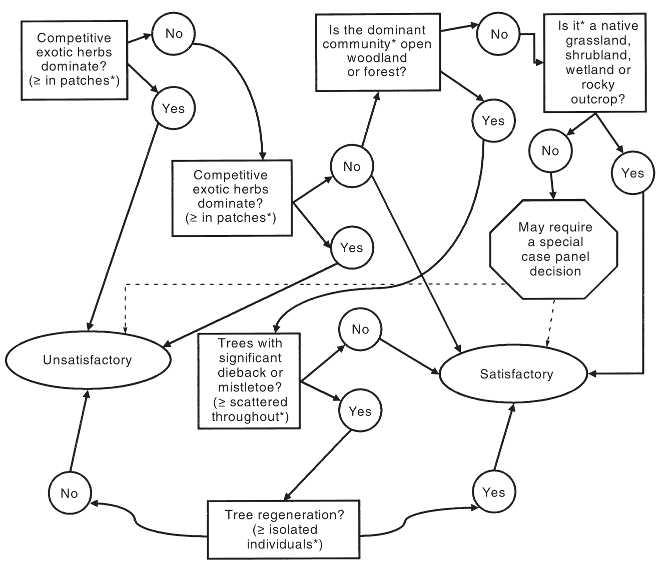

Figure 2. Decision tree 2: Native vegetation condition. *See site assessment sheets for details of site condition and habitat features.

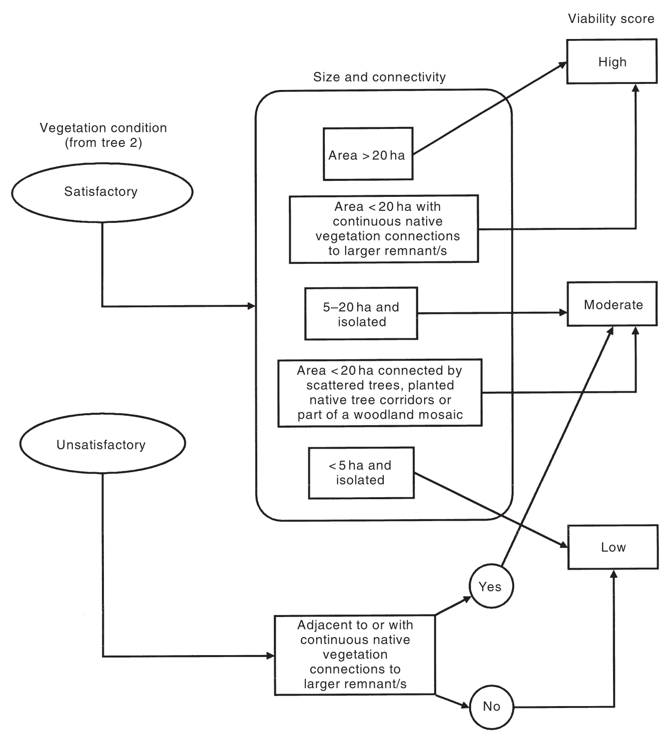

Figure 3. Decision tree 3: Remnant viability.

funding priority. For a remnant that is on a basic volcanic soil and is, therefore, valued highly for agriculture, this alone may result in the allocation of a higher funding priority in the fourth tree.

From a different perspective, we may find that a certain remnant of moderate to high viability is not actually under threat from grazing because of its safe location and good vegetation condition. These remnants are often found in steep, inaccessible or rocky country and are likely to remain protected without fencing.

We may have to reflect further on a farmer's willingness to manage a remnant actively when exotic species and tree-health decline are potential threats. This may require a special-case decision before the panel can allocate a final funding priority to a remnant that is small, weed-infested or in poor condition but in which there may be a threatened species, a significant community or a special habitat. Protection of such a remnant may require more funds for active bush regeneration work if the landholders are either uninterested or do not have the resources to carry out such works.

For economic reasons alone, it is preferable to have more area protected for less fencing. Therefore, we determine the ratio of remnant area protected to fence length and adjust our funding priority accordingly for remnants with high to moderate viability ranking and no threatened species, significant community or special habitat present. In ecological terms, this ratio enables us to adjust our final funding priority based on the extent of core area compared with the length of perimeter or edge in a remnant. In our final evaluation, we generally fund any remnant to which the panel has assigned a greater than moderate funding priority after all else has been considered.

Discussion. Fencing remnants and then abandoning them is not a sound management practice. This approach would only work well for remnants that are unlikely to degrade further after fencing. In order to address this issue, we intend to initiate a monitoring programme for all fenced remnants. Monitoring would be particularly important in remnants that are under considerable threat from weeds or where the exclusion of grazing may lead to a decline in native species richness and reduction in tree regeneration. We intend to recruit farmers into a network of people interested in conservation and bush regeneration. We hope to encourage these farmers to monitor their own remnants by providing them with basic training. This form of community education is important because it could lead to the development of a monitoring ethic among interested landholders. In addition, we hope to monitor a selection of fenced remnants in the region more comprehensively, but may need additional funding support to carry this out extensively.

Apart from monitoring remnants, we also wish to carry out manipulative field experiments to determine 'best management practices' under different scenarios of management. Once again, however, we will need more funds before entering this next phase of remnant vegetation management and conservation on private land.

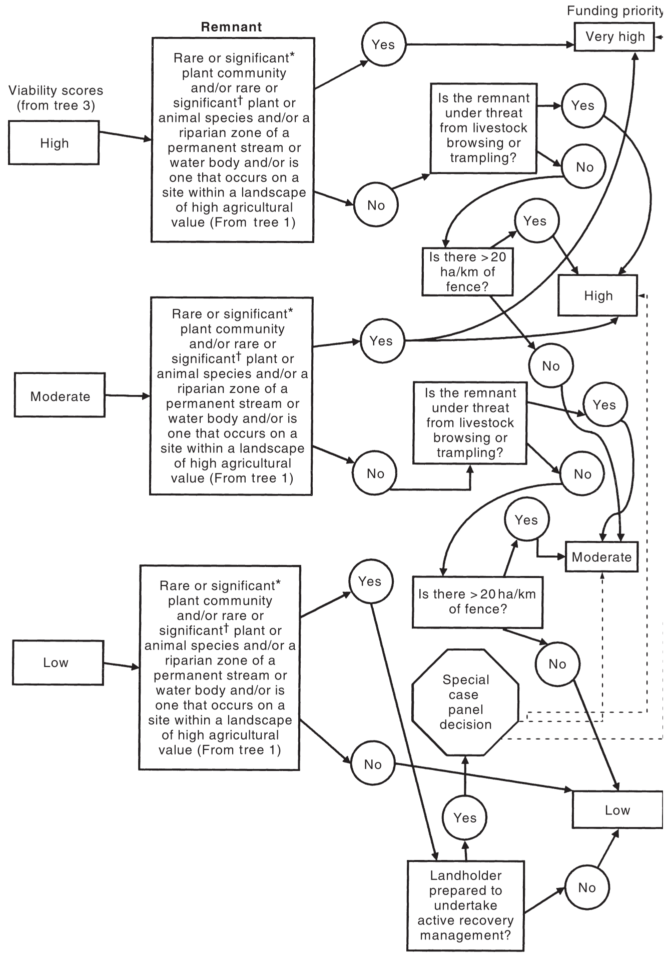

Figure 4. **Decision tree 4: Assigning properties for funding. *Threatened community under the *NSW Threatened Species Conservation Act, 1995* (TSCA Act) or recognized as regionally rare or poorly conserved. †Rare or Threatened Australian Plants (ROTAP) or those listed in schedule 1 or 2 of the TSC Act, or recognized as regionally rare, poorly conserved by scientific evidence (published scientific literature) or a species that is important as a component of habitat. The use of the word 'significant' in certain situations will require further discussion.**

In a fragmented landscape where grazing will continue to have an impact on at least some native vegetation remnants, fencing that excludes stock may help to prevent further loss of species and land degradation. There is a great deal we still need to learn, however, about managing native vegetation remnants for nature conservation on private land. In part, this will involve encouraging landholders to use an adaptive approach to monitor, record and manage changes in native vegetation remnants. Scientists can also contribute to the development of best management practices for native vegetation remnants by carrying out research in partnership with landholders and organizations such as Greening Australia in the already fenced management units of remnant bushland.

LANDSCAPE PATTERN AND DESIGN

12.1

The biodiversity integrity index: an example using ants in the Avon River Basin.

J. D. Majer (School of Environmental Biology, Curtin University of Technology, P.O. Box U1987, Perth, WA 6845, Australia)

Key words: *biological indicators, integrated ecosystems management, landscape pattern and design, monitoring and assessment, planning.*

A study was recently commissioned to review current knowledge of the status of biodiversity in the Avon River Basin, which extends from the edge of the Perth metropolitan area to the Goldfields (Environs Consulting 1999). The study covered all natural biodiversity, including vascular and non-vascular flora, vegetation communities and terrestrial and aquatic fauna. This was carried out to provide a baseline for the identification of trends for the purposes of planning, prioritizing and decision-making.

Biodiversity Integrity Index: Rationale. To synthesize the impact of disturbance on terrestrial invertebrate biodiversity, we applied a procedure developed by Majer and Beeston (1996), termed the Biodiversity Integrity Index (BII). What is missing from conventional measures of biodiversity is a consideration of the integrity, or departure from the unmodified state, of biological diversity across the entire landscape. Although there is clearly a need to assess biological diversity in selected areas (Noss 1990), the value of such a measure would be enhanced if this was combined with a consideration of landscape 'condition' or quality compared to the unmodified state. The BII attempts to do this.

Ants were used as the response taxon because their distribution is well known and their community response to disturbances has been the subject of a range of investigations. Other taxa, such as reptiles (e.g. Bracken 1996) or certain plant groups could also have been used, provided that the relevant data were available. The BII is the product of a measure of diversity or community composition for a particular landscape unit and the area that the unit occupies. The diversity measure could be almost any one of the conventionally used measures, such as species richness, Shannon's diversity index or even the number of threatened species within a particular area. Alternatively, the similarity between disturbed and control plots in terms of community composition could be used.

Consider a landscape that contains four major habitat units. The units are first standardized to the same area. Unit 1 is in a totally unmodified state, unit 2 contains a mine that occupies 10% of the area, while unit 3 contains a similar sized mine and also farm land covering 45% of its area. The remaining unit is all cleared farm land. Diversity or community similarity is set at 1 in the unmodified state and, in this hypothetical example, respectively, changes to 0.5 and 0.33 in the mined and farmed areas. The BII is then assessed within each habitat unit and for each type of disturbance. The BII in unit 1 is 100 × 1.0, or 100, which is the maximum attainable figure. In unit 2 it is 90 × 1.0 (unmodified) plus 10 × 0.5 (mine), which gives a value of 95. In unit 3, the BII is the sum of values for the unmodified area (i.e. 45 × 1.0) and the two types of disturbance [10 × 0.5 (mine) + 45 × 0.33 (farm)], which gives a value of 65. In unit 4 there are no unmodified habitats, so the BII is 100 × 0.33, or 33. In instances where diversity increases above unmodified state levels because of the presence of species favoured by disturbance, the value is still considered to be 1.

Land use data collection and analysis. The Avon River Basin Study was carried out by using the 40 individual shires within its boundary as the units of area. The major land-use categories present in the Avon River Basin can be broadly described as mining (generally with subsequent rehabilitation), agricultural clearing, range-land grazing, urbanization and roads. Salinization is considered as a subset of agricultural clearing. Aquatic ecosystems are not included in this analysis.

Information on the extent of uncleared vegetation, agricultural clearing and range-land grazing was obtained for each shire by superimposing digital data sets of vegetation boundaries over those for land use using the Intergraph® Microstation Geographic Information System Environment (Intergraph Corporation, North Ryde, Australia). The uncleared land comprised nature reserves, national parks, vacant Crown land and private land. The extent of urbanization was obtained from population figures derived from the national census (WAMA 1998). The areas occupied by a range of Western Australian town sites were obtained from maps and the populations of these town sites were regressed against area in order to elucidate the area requirement per capita of population. The resulting figure, 0.15 ha, was then multiplied by the population in each shire to estimate the area that has been urbanized.

The area under roadways was obtained from data on the length of roadways in each shire (WAMA 1998). The total length of road was multiplied by the mean width of the road plus drainway (10 m) to give an estimate of the area occupied by this category. The road verges were not included, because these can still support a considerable diversity of plants and animals (Keals & Majer 1991).

The area occupied by mining in each shire was obtained by counting the number of operating and closed mine sites using the Minedex data provided by the Department of Minerals and Energy. In view of the fact that 1500 km^2 of the Western Australian land surface has been disturbed by mining (CMEWA 1990), the area disturbed in each shire was obtained by multiplying this value by the proportion of mines that occurred in each shire. This provided only an approximation of the true figure, because mines differ significantly in size. However, the effect of differing size is likely to be averaged out by the large number of mines in the region.

Biodiversity data collection and analysis. Surveys of the impact of these uses of the land on ant communities have been carried out using broadly comparable sampling techniques at various localities throughout Western Australia. From these, a series of ant surveys, representing each of the major land-use categories were selected. Essentially, all surveys applied an insect-sampling protocol to a series of plots representing unmodified and disturbed environments. Each study yielded approximately 100 species of ants, with 20–40 species occurring in each plot. The reader is referred to Majer and Beeston (1996) for a full description of these studies. The ant data sets were first converted to presence/absence matrices and the total number of species per plot was summed. The mean value for species richness in both unmodified controls and disturbed plots was then calculated and disturbed plot means were converted to a proportion of the control plot value. The similarity between all plots within a data set was also calculated by the Systat® computer package (Evanstone, IL, USA) by using Sokal and Sneath's (1963) quotient of similarity. The plots were then hierarchically clustered by using the complete linkage procedure. The distance along the S3 axis of the resulting dendrogram was taken as a measure of the degree of similarity between the disturbed and control plots.

This procedure can also provide a statement on the relative impact of different land uses on the BII. Loss of BII values in all 40 shires because of the five broad categories of land use may be calculated by subtracting the shire BII values from the values that would have been produced if ant similarity quotients were 1.00 in each of the component land uses (maximum loss = 40 × 100 = 4000).

Results and implications. The results using quotient of similarity values are described here. Values for each of the major land units were: urbanization, – 0.20; roadways, – 0.00; mining (including rehabilitation), – 0.38; agricultural clearing, – 0.32; and range-land grazing, – 0.71. These were then multiplied by the area of land in each land-use group within the shires and the total for each of the five land-use groups was summed to give the shire BII values shown in Fig. 1.

The BII values range from 33.48 in the Shire of Cunderdin to 93.90 in the Shire of Yilgarn. The map has somewhat of a 'checker-board' appearance in view of the coarseness of the plotting units; shires such as Mount Marshall and Kulin traverse a wide variation in land use. Shires such as these therefore tend to produce BII values that are an average of more than one type of land use (pastoral grazing and agricultural clearing in the case of Mount Marshall). Despite this limitation, a clear trend in BII is evident within the Avon Basin. Biodiversity is most intact in the far-eastern one-third and to a lesser extent the northern part of the Avon Basin, where shires have not been completely opened up for agricultural clearing (e.g. Lake Grace, Kent and Kulin in the east, Dalwallinu in the north), or where the more benign range-land grazing is a predominant land use (e.g. Coolgardie, Dundas and Yilgarn). A second region of relatively high BII values exists within the shires in the extreme west of the Avon Basin. These are the shires that contain a proportion of State Forest (e.g. Wandering, Beverley and Toodyay). Aside from these patterns, the most noticeable trend is the region of low BII that centres on the shires of Cunderdin and Tammin. The region extends with decreasing intensity through approximately three shires to the north-northeast, the south-southwest and to the east.

A summary of the loss of BII values as a result of each of the five groups of land use is shown in Table 1. It indicates that agricultural clearing in the Avon River Basin has produced by far the greatest impact on ant BII, closely followed by range-land grazing. The lower losses of BII as a result of urbanization, roads and mining puts into context the lower impact of these, relatively restricted, land uses.

The findings presented here indicate the relative simplicity and utility of the BII procedure. The invertebrate section was only a small component of the Avon River Basin study, yet the synthesis map provides a visual summary of the status of biodiversity, which corresponds to the trends that were generated in other parts of the report (Environs Consulting 1999). It is suggested that future studies of the status of biodiversity in a regional context might profitably adopt this procedure for synthesizing the impacts of different land uses.

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Table 1. Biodiversity Integrity Index (BII) values in the 40 shires studied (maximum loss = 4000) as a result of different categories of land use

	Land use				
	Urban	Roads	Mining	Agricultural clearing	Range-land grazing
Biodiversity integrity based on similarity quotient	3.30	13.45	0.76	1914.93	35.22

Loss is calculated by subtracting the BII values for each shire from the values that would have been produced if ant similarity values were 1.00 in each of the land categories.

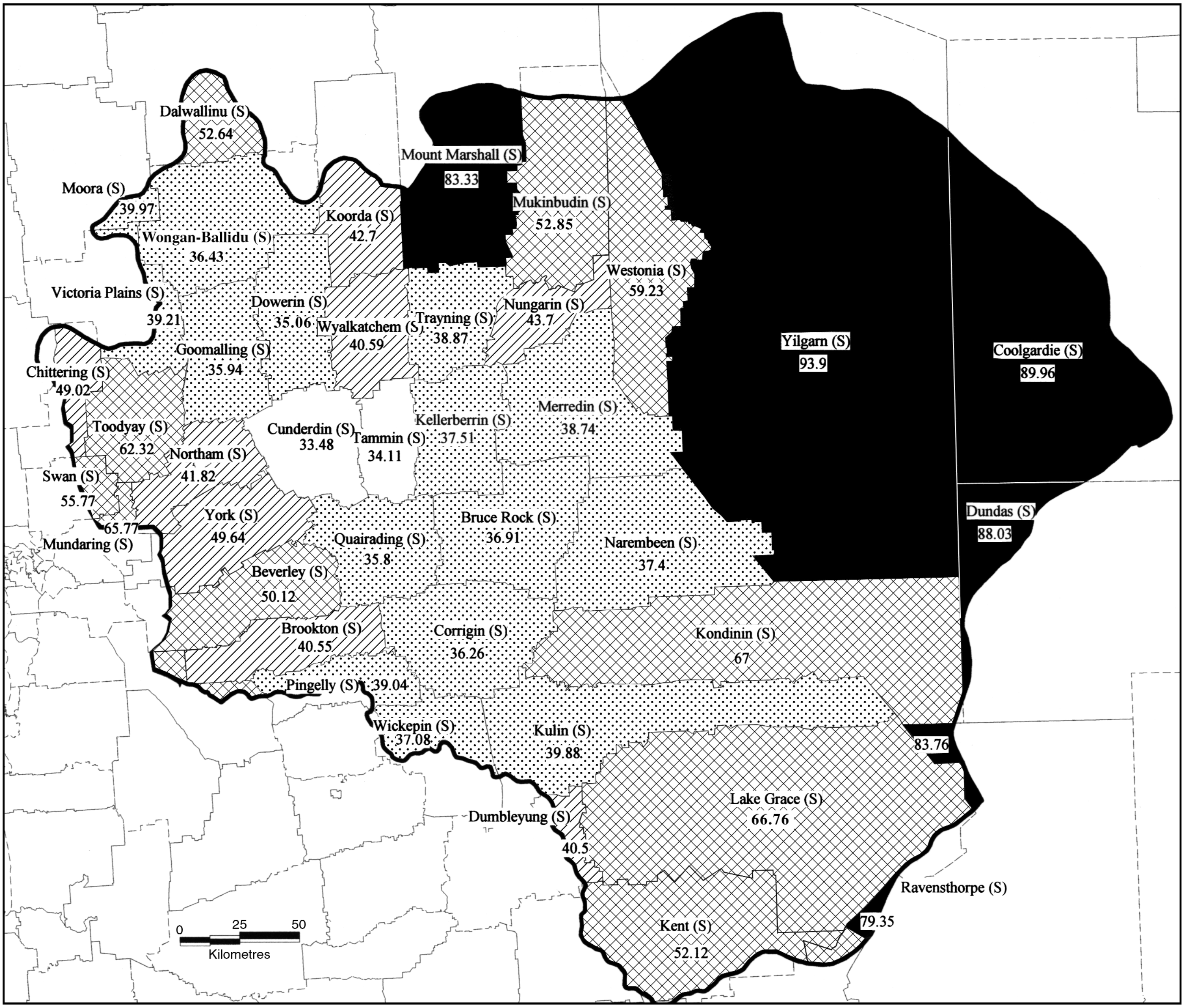

Figure 1. Map of the Avon River Basin (bounded by thick line), showing the Biodiversity Integrity Index (BII) scores for each of the component Shires. BII scores are based on quotients of similarity of ant faunas in disturbed and undisturbed areas. Biodiversity Integrity Index scores: (□), 0–35; (⊞), 35.1–40; (▨), 40.1–50; (⊠), 50.1–80; (■), all others.

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ORGANIZATIONS

24.3

The Queen Elizabeth II National Trust was established by an Act of Parliament in 1977 to commemorate the Queen's Jubilee year, 'to encourage and promote the provision, protection and enhancement of open space for the benefit and enjoyment of the people of New Zealand'. This broad statutory mandate means the Trust works with a wide range of people and organizations throughout the country, from individual landholders through to representatives of local and central government. The genesis of the National Trust was Federated Farmers' concern for growing threats to New Zealand's natural environment and their desire to see established some form of independent legal protection to help landholders protect valuable landscape features.

Functions. One of the Trust's principal functions is to protect privately owned areas of open space, without jeopardizing the rights of ownership. While private landowners may sell or gift land to the Crown or a local authority as a reserve, many wish to protect their land while retaining ownership, and open space covenants are a satisfactory answer. A Queen Elizabeth II National Trust open space covenant is a legal agreement between the Trust and a landholder to protect a special landscape feature for a specified time, more usually, in perpetuity. Covenants are almost always owner initiated, reflecting the landholders' goodwill toward this form of protection.

Coverage. Today, over 1300 open space covenants have been registered in New Zealand, covering approximately 49 000 ha. These covenants protect a variety of open space, including forest and forest remnants, wetlands, lakes, peat lakes, coastline, tussock grasslands, tracts of rural landscape, archaeological sites, and geological formations. Over 100 000 ha of land have been approved for protection.

Structure. The National Trust is administered by a Board of Directors, four of whom (including the Chair), are appointed by the Minister of Conservation. These appointments are made after a public invitation for nominations. In making the appointments the Minister must have regard to the interests of the rural community, the Maori community and environmental and conservation values. A subcommittee of the Board, Te Komiti Whenua Toitu, has been established with specific responsibility for matters relating to Maori land. Two Directors are elected by Trust members.

Financial basis. The Trust is financed partly by Government grant and partly from income from donations, interest on investments and membership fees. In all of its work the National Trust receives considerable voluntary assistance and advice from local interest groups, central and local government agencies, and from individuals. This is, in part, because of the Trust's independence from government, but possibly of more significance, its record of achievement in protecting private land in partnership with the landholders.

Further information: Queen Elizabeth II National Trust, P.O. Box 3341, Wellington, New Zealand. Email: qe2@qe2nattrust.org.nz

24.4

Wetland Care Australia is a national, independent, non-government organization which formed after a small group interested in wetlands attended a lecture in Adelaide in 1987, delivered by Dr Frank Baldwin from Ducks Unlimited, Canada. Dr Baldwin told how Ducks Unlimited Canada had invested $600 million on more than 10 000 projects covering 5 million hectares. Inspired by his talk, the group approached Ducks Unlimited in Canada, USA and New Zealand and started a similar organization in Australia.

In 1996, in order to reflect more fully the total focus on wetland ecosystems, the name Wetland Care Australia was adopted and the organization now has its head office in Berri, South Australia and branch offices in Canberra, Ballina and Darwin. Wetland Care Australia now has an established network of some of Australia's foremost experts in wetland conservation and management, and these experts are available for consultation by community groups and other interested people. Wetland Care is a not-for-profit company and trustee for the Wetland Care Australia trust fund. The company and the trust are dedicated to wetland conservation and donations to the trust are fully tax-deductable.

Mission. Wetland Care Australia realized early that most of Australia's wetlands are not protected in the public reserve system — thus their conservation depends on organizing many different groups and individuals to work together. The mission of Wetland Care Australia reflects this need: 'To harness community, business and government resources so that together we can work with landholders to enhance Australia's wetlands'.

Communications. The WetlandLink project aims to improve nationwide communication about wetland issues. It consists of a free quarterly, the *WetlandLink Bulletin*, the website and an email discussion forum. WetlandLink has access to a database of nearly 1000 people, including representatives of community groups, landholders and government agencies who can provide information, technical assistance and further contacts.

Further information: contact either Joanne Green and Heather Shearer (Wetland Care Australia, P.O. Box 154, Ballina, NSW 2478, Australia. Email: wca@linknet.com.au Website: www.wetlandcare.com.au Email discussion list: list@wetlandcare.com.au)

BOOK REVIEWS

26.5

Feral Future. (1999) Tim Low, Viking (Penguin Books Australia, Ringwood, Vic.). ISBN 0670884650. NZ$32.95.

While most people can intrinsically recognize that there is something environmentally wrong with an oil spill or a clear-felled forest, it is much more difficult to recognize the damaging effects of invasive species. Those of us who work with exotic invaders are frequently confronted with a puzzled look and a 'so what?' attitude, or get told 'but it's so pretty'.

Tim Low sounds like somebody who has had enough. *Feral Future* is a book written to shake the public out of their apathy. As well as aiming to raise the profile of invasive species, it also clearly aims to raise a few hackles. Home gardeners, botanic gardens, pet shops, acclimatization societies and even the CSIRO are confronted with the role they have played, and are still playing today, in the introduction of exotic pests.

Feral Future starts with an unusual reference to overgrazing in the book of Genesis and moves through the writings of Theophrastus, Pliny and Cato before finally focusing on contemporary exotic invasions of Australia. The early history of exotic introductions into Australia is an interesting story and with the benefit of twenty-first century hindsight, it is also a bizarre one. How could anyone have entertained the idea that it would be a good idea to introduce feral monkeys 'for the amusement of the wayfarer whom their gambols would delight as he lay under some gum tree in the forest'? Few are likely to argue with the assertion that 'acclimatisation was one of the most foolish and dangerous ideas ever to infect the thinking of nineteenth-century men', at least not after reading Chapter 5. However, there is a sting at the end of the chapter, where CSIRO pasture scientists are described as the acclimatizers of today.

The section 'Careless Ambitions' discusses the deliberate introduction of some of Australia's worst pests. This is where the book starts to get challenging. For a start, it's still a big mental jump for some people to see pastoral grasses and legumes or trout as pests, even when there is convincing evidence of the damage they cause. And, unlike the nineteenth-century acclimatizers, those responsible for some of these introductions will still be alive.

As a New Zealander, I looked especially hard at the section 'Australians as pests', since this is the area I know best. There is a good range of Aussie invaders discussed, from the Brushtail Possum (*Trichosurus vulpecula*), our worst pest, to some of the obscure and recent plant invasions such as Coast Banksia (*Banksia integrifolia*) and Wonga Vine (*Pandorea pandorana*). To find out about these species takes a bit of extra effort — they are still invaders that few people have heard about in New Zealand. I was also interested to see the comment that 'a few' Redback (*Latrodectus hasselti*), Huntsmen (*Delena cancerides*) and Whitetail (*Lampona cylindrata* and *L.murina*) spiders live around New Zealand houses. This is true for the first two species, but Whitetail spiders are actually common here and have been the cause of a reasonable amount of public concern. Likewise, Bleeding Heart (*Omalanthus populifolius*) does not confine itself to the north of New Zealand; it is spreading around Nelson City. To be fair, this is very recent information. What it does suggest is that inaccuracies for New Zealand are underestimates rather than overestimates.

Tim Low deserves particular credit for bringing to the public attention some of the more complex issues related to invasive species, and in an easily understandable way. Some of the issues he tackles include: native plants becoming environmental weeds outside their natural range; the way many invasive species take years before they become invasive; the role of free trade in exotic species invasion; and new invasive species which are yet to reach Australia. These concepts are difficult for people with no background in ecology to grasp, but they are important if the general public is to make informed decisions in the future.

It is clear from *Feral Future* that Tim Low has a great passion for his subject. Part of the appeal of this book is the way his passion carries through. However, if asked for any criticism, I would have to say that some of the comments in *Feral Future* seem unduly personal. Those of us dealing with the present day consequences may feel aggrieved at the legacy of acclimatization, but there is little value in blaming the acclimatizers for their actions. The way that criticism is aimed at botanic gardens, permaculturalists, CSIRO scientists, AQIS and home gardeners may be more likely to make people defensive and miss the message of the book. On the other hand, perhaps these things need to be said in order to raise the issues. There is also a fair amount of emotive language such as 'African grasses that violate woodland and rampant garden vines that ruin rainforests'. This is prose, not science, but it is backed up elsewhere in the book by facts.

When I finished reading *Feral Future*, I have to say that I felt pleased. At last somebody was putting the problem of invasive species right in front of people, challenging them to look differently at an issue so many people don't even realize is there. It's readable, relevant and interesting — a book I can recommend to professional colleagues and non-ecological friends and family. Most of all I hope that it makes people think.

Review by Melanie Newfield, Weed Ecologist, Department of Conservation, Nelson/Marlborough Conservancy, Private Bag 5, Nelson, New Zealand. Email: mnewfield@doc.govt.nz

CONFERENCE REPORTS

27.7

Managing Grassy Woodlands Workshop. 23–24 March 2000, The University of Sydney (Orange Campus), Cumnock, NSW.

Grassy woodlands, once widespread on fertile soils of the slopes and plains of south-eastern Australia, are some of the most highly threatened ecological communities in Australia. Most woodland organisms have suffered severe habitat loss, and many continue to decline today.

This workshop brought together some of the increasing number of people involved in grassy woodland conservation. Held over 2 days, the workshop was jointly sponsored by the University of Sydney (Orange), Department of Land and Water Conservation, New South Wales National Parks and Wildlife Service (NPWS), Greening Australia, and Community Solutions The first day targeted scientists, extension officers, Landcare coordinators and other professionals involved in researching and promoting woodland conservation and management issues. The second day brought in hands-on managers including property owners, community group members and Rural Lands Protection Board officers.

Day 1 Prior to the meeting, participants were invited to provide a one-page summary of their work on grassy woodlands. This provided a good background, allowing greater detail than could be presented on the day, and included information from those unable to attend. The day was structured around a series of short talks summarizing the state of knowledge of woodland biology, covering a range of biological groups (understorey plants, trees, birds, reptiles, bats) and perspectives (individual organisms, patches, landscapes, ecological communities). Ample question time allowed wide discussion around these areas. After splitting into groups, participants considered management options for a number of case studies of real sites. The day concluded with an hour-long open discussion on woodland issues.

Several key issues arose during the meeting. As more information becomes available from different research areas, it is becoming clear that the needs of different groups of organisms may be quite different, and conservation goals for one group may be decoupled from, or even conflict with, goals for another. This is evident at several scales. Rainer Rehwinkel (NPWS) described the dilemmas facing managers when one site contains up to seven threatened species with widely differing ecologies. Knowledge on the needs of cryptic fauna, including many reptiles, is also still limited but it is likely that management regimes such as burning that favour some plant species will disadvantage some fauna, at least in the short term and at a small scale. At a broader scale, several studies showed that many woodland birds such as Hooded Robins (*Melanodryas cucullata*) need large remnants with structurally complex vegetation and scattered shrubs, while some uncommon native forbs such as Yam Daisies (*Microseris lanceolata*) thrive only in little-grazed (and hence small), regularly burnt remnants with few shrubs.

The fundamental problem facing managers is that for some ecological communities and some landscapes there are very few remnants left and many of them are very small and widely scattered. Management and reconstruction needs to be applied in carefully designed mosaics to allow the continued survival of all elements. Mosaic management is needed at all levels: patch, landscape and ecological community. A key approach then is to view each remnant in relation to other remnants within a landscape and within an ecological community, rather than in isolation. This will allow for the connectivity and diversity required by organisms that utilize a range of remnants, and ensure that broad representation of each ecological community can be achieved. As well, it will help to address landscape-scale problems such as salinization which can threaten the very survival of remnants.

Other issues discussed included: the need for improved integration across different land uses and different organizations, especially government agencies; potential funding sources that might allow large-scale rehabilitation of grassy woodland landscapes (e.g. carbon credits, salinity control programs); and the difficulties farmers face in the current economic climate. Many gaps in our knowledge were highlighted by the meeting (e.g. the needs of bats, reptiles, insects, microorganisms; reconstruction and enhancement techniques for native understorey; the utility of crash grazing as a management tool; and a good overall understanding of the way these ecological communities function). However, gaps in knowledge should not detract from on-ground application of those things we know how to do, now.

Day 2 Over 100, mainly land managers, attended day two of the workshop, focusing on the question: 'Is a fence enough to protect and retain our grassy woodland remnants?' Kevin Thiele set the scene at the morning session by describing grassy box woodland; and David Goldney, having suggesting that many remnants would have disappeared in 10–30 years, asked: 'How do we put back critical landscape functions and how do we farm in a sustainable landscape?'

Donna Windsor spoke about managing disturbed grassy woodlands, with a striking message that the presence of various species can indicate the disturbance history of a site. For example, Heliotrope (*Heliotropium europaeum*) and Skeleton Weed (*Chondrilla juncea*) give an indication that the area was a stock camp, while the presence of Pinrush (*Juncus filicaulis*) indicates waterlogging. Sea Barley Grass (*Hordeum* sp.) indicates saline soils; Sorrel (*Rumex acetosella*), Bracken (*Pteridium esculentum*) and Catsear (*Hypochoeris radicata*) indicate acid soils. A predominance of Red Grass (*Bothriochloa macra*), Wiregrass (*Aristida ramosa*) and annual weeds imply the area has been heavily grazed. From there you can look at potential management strategies. For example, if an area has been a stock camp she found that scalping and scarifying is a strategy that can stimulate tree regeneration.

Ian Lunt spoke about managing higher quality woodlands, identifying the need to: maintain or enhance the size and connectivity of the site; prevent large soil disturbance and inputs of water or nutrients; and use plantings to enlarge or connect sites (for fauna) and to buffer sites (for flora). He identified that grazing is a major factor affecting plant diversity and that most of the diverse sites have been rarely or intermittently grazed. The take-home message, therefore, was not to increase the stocking levels at sites that have been rarely grazed.

The afternoon was spent visiting two field sites, (i) a travelling stock reserve which had rarely been grazed and had a high diversity of plants and animals, and (ii) a grazed property with regeneration. There were a number of strategies proposed for the second site including burning and cell grazing (grazing small areas for short periods to allow time for root growth and regeneration). The take-home message was that fencing is a very useful management tool that allows a grassy woodland remnant to be managed for vermin and grazing control. Fences are often not enough, however, and grazing, burning or other strategies may also need to be introduced to maintain healthy grassland populations and to prevent domination by some weed species.

The most stimulating part of the workshop was the question and answer session. Questions were written down during the day and given to the facilitator who arranged them into topics and then posed them to the group. I'd recommend this method to other workshop organizers.

Day 1 report by Suzanne Prober and Kevin Thiele, CSIRO Division of Plant Industry; Grassy Box Woodlands Conservation Management Network Project. Email: kevin.thiele@pi.csiro.au
Day 2 report by Helen Ryan, Grassy Ecosystems Networker South East Australia, 10 Parliament Place, East Melbourne, Vic. 3002. Email helenr@vnpa.org.au