

Effective Landscape Restoration for Native Biodiversity in Northern Victoria

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Introduction

Biodiversity is the variety of all life forms – species of plants, animals and micro-organisms, the genes they contain, and the populations, communities and ecosystems they create – and the *interactions* between and among them and the physical environment that generate ecosystem (or ecological) processes (SEAC 1996, Saunders 2000, SER 2004). Examples of ecosystem processes include carbon fixation by plants (photosynthesis), nutrient cycling by micro-organisms, nitrogen fixation by bacteria, decomposition of organic matter, water filtration, pollination of flowering plants by fauna and seed dispersal. Ecosystem processes that are of direct benefit to humans (e.g., carbon sequestration, water production, pest control) are called ecosystem services (Daily 1997, CSIRO 2001).

It is now widely accepted that current and past land-use practices on a global scale have caused massive biodiversity loss and damaged natural systems, including the provision of ecosystem services (Tilman et al. 2001, Foley et al. 2005). Biodiversity loss is most readily expressed in terms of species extinctions and imperilment. For example, the most recent inventory of species threatened with global extinction listed more than 16,000 plants and animals, including 10% of all described vertebrates (IUCN 2006). In Victoria, over a century of agriculture based on European farming traditions have compromised the viability of many native species. Many of the 550 taxa and 36 communities currently listed as threatened under the Victorian Flora and Fauna Guarantee Act 1988 (DSE 2006), including many in northern Victoria (DNRE 1997), have been negatively affected by agriculture. Paradoxically, land management practices have also degraded the biophysical environment upon which agricultural production depends: rising ground-water tables and dryland salinity, increasing soil acidification and erosion, loss of soil biota, eutrophication of waterways and wetlands, altered hydrological regimes, and the spread of exotic animals and weeds are symptomatic of dysfunctional landscapes. Such threats have generated an urgent need for remedial work in agricultural landscapes to restore ecosystem processes that underpin sustainable agriculture and natural ecosystems.

Although radical and widespread changes in land management are required to reverse current declines in biodiversity and ecosystem processes, the dominance of high-input farming practices coupled with market demand for ever-increasing production has obstructed the integration of ecologically sustainable land management with agri-production systems. This is not to suggest sustainable land management is an anathema to profitable agricultural production. On the contrary, without it, production will ultimately fail (Wright 2004, Diamond 2005). However, there are many reasons why land managers have been reluctant to change, including the lack of comprehensive policy shifts to stimulate change; economic paradigms that do not recognise environmental costs; vested commercial interests in current practices; financial costs; distrust of scientists and management agencies; mismatch between the scales of ecological research and on-ground management; ineffective communication between scientists and land managers; and technical deficiencies and complexity (Geurin and Geurin 1994, Saunders 2000, Yencken and Wilkinson 2000, Lefroy and Smith 2004, Carr and Hazell 2006, Pannell et al. 2006).

Despite these impediments, there is growing discontent with the status quo among sections of the farming community and acknowledgment that current practices are not sustainable (Milne 1995, Nicholson 2000, Clewell and Aronson 2006). This has stimulated a groundswell of local revegetation activities, although the primary motivation for revegetation has been the provision of shade and shelter for stock and rehabilitation of degraded land (Bennett et al. 2000). While production remains the focus for revegetation, most revegetation programs will be too small in scale and of inadequate design and quality to address biodiversity loss and ecosystem decay. More emphasis needs to be placed on restoring resilient ecosystems at large spatial scales with long-term timeframes. Thus, restoration includes, but is more than, revegetation.

Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SER 2004). Because ecosystem processes are the basis for self-maintenance in an ecosystem, a common goal for restoration is to recover self-renewing or *autogenic* ecosystem processes (Whisenant 1999); that is, to build ecosystem resilience. Thus, in the broadest sense, restoration may take many forms, including revegetation (e.g., replanting, regeneration), changes in farming practices (e.g., crop rotations, grazing intensity, fertiliser application, pasture species), manipulation of natural disturbances (e.g., fire regimes, hydrological flows), education (e.g., retention of logs), control of exotic species, and manipulation of biophysical habitats (e.g., soil crusts, hollows).

Restoration ecology is a young science; therefore, restoration activities have often been conducted without reference to a conceptual framework (Hobbs and Harris 2001). While some projects have achieved significant social and environmental outcomes, the lack of a cohesive framework has limited the effectiveness of many projects and perhaps stifled the development of better ways to repair landscapes. Rather, a variety of approaches and ideologies have arisen independently as practitioners grapple with the complexities of restoration in the face of the urgency of finding solutions. This has led to the development of numerous 'guidelines' or 'rules' for restoration, which although generated from location and context-specific experience tend to be disseminated widely. Thus, land managers seeking to undertake restoration are confronted with an unfamiliar lexicon and a bewildering (and sometimes contradictory) array of restoration approaches, without the support of a cohesive conceptual framework for implementation. In response to this, the aim of this discussion paper is to:

- (i) demystify the language of ecological restoration;
- (ii) synthesise current thinking on a conceptual framework for restoration; and
- (iii) critique common approaches for restoration planning.

The scope of this paper is the restoration of biodiversity and ecosystem processes in agricultural landscapes of northern Victoria, although research from around the globe will be canvassed. We focus on private land because while the public reserve system protects irreplaceable core areas, it is inadequate in extent and diversity to sustain all species or maintain broad-scale ecosystem processes (Bennett et al. 1995, Martin & Martin 2004, Fischer et al. 2006, Mackey et al. in press). We will briefly review the threats to biodiversity in northern Victoria, define landscape restoration and related terms, discuss goal-setting and indicators, and examine strategies for restoration planning (including some commonly advocated 'rules') in a northern Victorian context. Case studies from properties in the region will be used to articulate the values and limitations of restoration approaches and identify priorities for research. Our emphasis is to identify and critique approaches to *planning* restoration, and outline where and when different actions may be most valuable for improving biodiversity and ecosystem processes at a property to sub-catchment scale.

1 Setting the Scene: Study Region and Policy Context

This paper focuses on the four Catchment Management Authority regions in northern Victorian – Mallee, North Central, Goulburn-Broken and North East. The key bioregions are the Murray Mallee, Wimmera, Lowan Mallee, Goldfields, Victorian Riverina, Northern

Inland Slopes and Murray Fans with smaller but significant occurrences of other bioregions, notably the Victorian Volcanic Plain, Murray Scroll Belt, Robinvale Plains and Central Victorian Uplands. The predominant broad vegetation types across the region are temperate woodlands and grasslands, giving way to mallee ecosystems in the north-west and montane forests on the slopes of the Great Dividing Range to the east.

The area forms the southern part of the Murray Darling Basin, draining north from the Great Dividing Range to the Murray River. Rainfall varies from 2000 mm annually in the east to less than 250 mm annually in the north-west. A diversity of landscapes are represented with predominant landforms being plains and low rolling hills with smaller areas of more elevated hill country. Soils are generally shallow and low in fertility with a history of erosion due to over clearing. Problems of soil salinisation, sodicity and acidity (especially in higher rainfall areas) have arisen as a result of past and current land use. Much of the region is freehold land (around 75%) with large public land blocks predominantly in the Mallee and in higher elevations of the Great Dividing Range in the east. Land use is correspondingly varied along a gradient from south east to north west with grazing (sheep and cattle) predominating in higher rainfall areas to cropping associated with lower rainfall regions on suitable soils. However, land use patterns are changing rapidly associated with declining farm viability and proximity to urban centres driving an expansion of small lifestyle holdings (Barr 2005).

A diversity of ecological communities can be found across the study area represented as Ecological Vegetation Classes (EVC's). Approximately 300 EVC's have been described. EVCs are derived from land system (eg, geomorphology, rainfall), vegetation structure, floristic information and other environmental information including aspect, fire frequency and ecological responses to disturbance. They describe local patterns of vegetation diversity but are not bioregion specific. At a finer scale than bioregions, EVCs have been shown to be useful surrogates of biodiversity for birds, mammals and trees (but less so for invertebrates and reptiles). In combination with the bioregions, the EVC classification system is an important tool for regional strategic planning across as they provides valuable information about the level of depletion and threat status of different vegetation types. It can also inform the planning of on ground vegetation management activities and revegetation (North Central Native Vegetation Plan, 2006).

Landscape restoration is integral to a number of strategic policies with diverse objectives at both the national and state level. For example, at the national level, the importance of protection and enhancement of native vegetation has been recognised in a number of policies. The establishment of the first phase of the Natural Heritage Trust (NHT) in the mid 1990's led to the development of the Bushcare program, the first nationally coordinated approach to biodiversity conservation on private land. Subsequent programs including NHT 2 and the National Action Plan for Salinity and Water Quality (NAP) have continued to invest at a range of scales in native vegetation management activities with a focus on delivery through Regional NRM bodies (see www.nrm.gov.au). The National Framework for the Management and Monitoring of Australia's Native Vegetation, which is currently being reviewed, was designed to provide an agreed framework of best practice management and monitoring measures to reverse the long-term decline in the quality and extent of Australia's native vegetation cover.

In Victoria, restoration is a key component of the Victorian Greenhouse Strategy, Victorian River Health Strategy and Victorian Native Vegetation Management Framework. The latter "establishes the strategic direction for the protection, enhancement and revegetation of native vegetation across the State" and thus provides the policy foundations for management of native vegetation in Victoria (DNRE 2002a). The framework introduces the concepts of Net Gain, conservation significance, habitat hectares and other tools for setting priorities in vegetation management. A primary goal for vegetation management is identified as "a reversal, across the entire landscape, of the long-term decline in the extent and quality of native vegetation, leading to a Net Gain", to be achieved through application of the principles and approaches to vegetation management outlined in the framework. At a catchment scale, Regional Native Vegetation Plans have been developed across all CMA regions to translate the statewide aims and objectives of the framework to specific regional circumstances. This has been valuable in identifying the current extent and condition of native vegetation, nature and degree of threatening processes together with regional guidelines and approaches aimed at achieving "Net Gain".

State Government initiatives to actively promote the broader goals of biodiversity enhancement have also commenced. Two examples are Bush Tender and Bioregional Action Planning. Bush Tender applies principles of game theory to establish a market for biodiversity and ecosystem services for landholders willing to manage their remnant vegetation for biodiversity conservation. Bioregional Action Planning is underway in several catchments across the state and recognizes the imperative of planning for biodiversity at multiple spatial scales, emphasizing the hierarchical nesting of catchment, bioregional, landscape and local spatial scales (Platt and Lowe 2002). Bioregional Action Planning is a tool for communicating and implementing landscape restoration, engaging local communities to increase their awareness of biodiversity issues and their capacity to change the landscape for positive biodiversity outcomes. It promotes the mantra of “protect, enhance, restore” – protect existing remnant vegetation as the first priority, enhance the biodiversity value of remnant vegetation, and restore areas that have been severely disturbed. Such initiatives complement programs run by non-government organisations (e.g., Trust for Nature, Australian Bush Heritage, Greening Australia, GreenFleet, WildCountry) and a suite of Commonwealth-funded incentive programs for native vegetation enhancement and biodiversity conservation coordinated through the CMAs (e.g., MCMA 2003, NCCMA 2003).

In prescribing off-sets for clearing native vegetation, in promoting restoration of disturbed land, and in purchasing or covenanting parcels of land for biodiversity conservation, government policy assumes that we have the capacity to replenish biodiversity in production-orientated environments. In other words, policy now dictates that as a society, we should invest in landscape restoration for biodiversity purposes. This is a fundamental shift in emphasis from biodiversity conservation based in reserves, to holistic management of entire landscapes in a manner that integrates profitable agricultural production with sustainable populations of native flora and fauna.

Unfortunately, in the past, a culture of short-term funding cycles and lack of policy integration at the state and national level has contributed to on-ground actions from different programs being implemented in isolation. A priority should be to integrate investments and projects to achieve multiple benefits, where possible, for salinity, greenhouse, river health, soil quality, biodiversity and ecosystem services (Park and Alexander 2005). Further, these environmental outcomes are sought in landscapes that are primarily devoted to agriculture. What is the best way to achieve environmental improvements while maintaining productive and profitable agricultural systems? If a solution can be designed that more fully incorporates environmental outcomes, will it be economically feasible and socially acceptable?

2 Threats to Biodiversity in Northern Victoria

Loss of habitat through clearing of native vegetation has been, and continues to be, a significant threat to biodiversity across northern Victoria (Robinson and Traill 1996, Bennett et al. 1998, Lunt and Bennett 1999, Radford et al. 2005). As well as the removal of woody vegetation, habitat loss also includes the conversion of grassland to crops or ‘improved pasture’, which although less obvious in terms of structural changes to the vegetation, is equally destructive. Historically, the agriculture and mining sectors were primarily responsible for broad-scale clearing. Foremost among contemporary motives for clearing are residential developments and agricultural intensification (e.g., for vineyards, olives, irrigation infrastructure and precision agriculture). Habitat loss decreases the resource base (i.e., food, shelter and mates) for individual animal species resulting in smaller populations with lower genetic diversity, increasing the probability of local extinction. Impacts on native plant species include their direct removal from the landscape and the viability of the remaining patches (Young and Clarke 2000). As the amount of habitat in a landscape decreases, fewer species are able to sustain viable populations, leading to a decline in species richness. Typically, clearing also decreases the diversity of vegetation types (ecosystem diversity) further reducing the number of species for which suitable habitat exists. Moreover,

the most fertile parts of a landscape are often preferentially cleared resulting in landscapes that are not representative of the original vegetation composition.

Habitat fragmentation – the division of formerly contiguous tracts of habitat into two or more discrete patches – usually accompanies habitat loss, leaving smaller, more isolated patches of remnant habitat (Fahrig 2003, Kupfer et al. 2006). Smaller patches support fewer species because minimum patch size requirements are breached for area-sensitive species (e.g., Bolger et al. 1991). The spatial arrangement of native vegetation affects the movement of plants and animals at a range of spatial and temporal scales, and therefore the ability of organisms to forage, migrate or disperse successfully (Bennett 1999, Murphy and Lovett-Doust 2004, Soulé et al. 2004). For example, fragmentation may force animals to traverse gaps of inhospitable habitat more often, with consequent increases in predation risk (Lima and Dill 1990) and energy costs (Grubb and Doherty 1999). Populations confined to isolated remnants experience lower immigration rates, compromising the genetic health of the population (e.g., Stow et al. 2001, Frankham 2005) and reducing the chance of demographic 'rescue' or re-colonisation following local extinction. Habitat fragmentation often interferes with species interactions because species are differentially affected by fragmentation (Kupfer et al. 2006). For example, the abundance of a species may increase substantially following the loss of a key predator or competitor with subsequent impacts for other components of the ecosystem (e.g., increased grazing or herbivory). Some native species, such as the Noisy Miner (*Manorina melanocephala*), thrive in fragmented landscapes, often to the detriment of other species (e.g., Grey et al. 1998). Although the majority of northern Victoria has been cleared for several decades, it is likely that the effects of habitat loss and fragmentation are yet to be fully realised, with further declines in biodiversity predicted as unviable populations successively disappear from the region (Recher 1999).

Habitat degradation (i.e., modification but not complete removal of habitat) also poses serious threats to biodiversity in northern Victoria, either directly or by compounding the effects of habitat loss. Degrading processes diminish biodiversity because they remove critical habitat elements, alter habitat structure and composition, disrupt ecological processes or introduce non-natural disturbances. For example, many species are sensitive to logging because it removes mature trees, homogenises forest structure and simplifies the ground layer (Traill 1991, Gibbons and Lindenmayer 1996, Kavanagh et al. 2004). Similarly, fauna reliant on dead standing timber or fallen logs are severely impacted by firewood collection (Wall 1999, Mac Nally et al. 2001, Lindenmayer et al. 2002a). The widespread use of exotic pasture species and the addition of fertilisers can see these and other species become invasive weeds in natural systems. These weeds displace native plant species and usually increase the density of ground vegetation, reducing habitat quality for a range of ground-dwelling fauna (NPWS 2002, Maron and Lill 2005). Excessive grazing by stock, feral species (rabbits, hares, goats) and native herbivores (kangaroos, wallabies) leads to a suite of problems including reduced ground cover, soil compaction, eutrophication and soil erosion, with concomitant declines in biotic diversity (Jansen and Robertson 2001, Martin et al. 2006, Dorrough et al. 2006). Predation of native fauna by foxes and feral cats and dogs has not been quantified for northern Victoria but it is probable that their impact, particularly on mammals and reptiles, has contributed to population declines, as it has in other parts of Australia (e.g., May and Norton 1996, Kinnear et al. 2002, Olsson et al. 2005, Davey et al. 2006).

Ecological restoration strives to restore natural disturbance regimes, so it is important to identify intrinsic variability in resources arising from natural disturbances and distinguish this from external disturbances (Wyant et al. 1995, White and Walker 1997). In northern Victoria, natural disturbances associated with fire, hydrological flow patterns and herbivory have been disrupted through direct human intervention or as a consequence of habitat loss and fragmentation. Fire management is a vexed and complex issue, one in which different sections of the community advocate contrasting management options (e.g., ECC 2001). Protection of human assets has usually taken precedence over biodiversity in management of fire regimes. In some parts of the landscape, prescribed burning for fuel reduction has increased fire frequency and extent (area burned) but decreased fire intensity, whereas fire suppression and habitat fragmentation has reduced fire frequency in other parts of the landscape (Gill and Williams 1996),

increasing the risk of periodic wildfires. Each of these interventions has detrimental impacts for biodiversity. A sustained increase in fire frequency alters the structure and composition of ground and shrub vegetation and reduces logs and ground litter, contributing to declines in ground-nesting and ground-foraging fauna (Woinarski and Recher 1997, Garnett and Crowley 2000). Yet, total exclusion of fire also alters vegetation composition and structure, favouring late successional species and reducing habitat for some fauna.

River regulation and inputs of pollutants and sediments have severely diminished the physical condition, water quality and habitat value of instream, riparian and floodplain ecosystems (Yencken and Wilkinson 2000). Water impoundments, changes to surface water flows, and diversion of water for irrigation have supported increased agricultural production at the expense of stream condition, such that only 27% of reaches in major Victorian rivers or tributaries are considered in good or excellent condition whereas 34% are in poor or very poor condition (DNRE 2002b). Loss and degradation of riparian vegetation has compounded the problem. Riparian vegetation plays a critical role in maintaining water quality, buffering the inflow of pollutants, stabilising streambanks and providing habitat for both terrestrial and instream fauna (Price et al. 2004). The eradication of native herbivores and introduction of domestic stock and rabbits has radically changed the disturbance regime of ground flora. Continuous grazing and excessive stocking rates have altered the competitive dynamics among native ground flora, favouring winter-active, unpalatable annuals, with consequent loss of diversity and ecosystem resilience.

Climate change is expected to increase temperatures in Victoria by between 0.7°C to 5.0°C, decrease rainfall by up to 25% and increase the frequency and intensity of extreme rainfall events, among other changes, by 2100 (DNRE 2002c). The direct impacts of climate change on biodiversity in northern Victoria are uncertain but it is likely that the altitudinal and latitudinal distribution of plant species will shift as they track climatic conditions to which they are adapted, with accompanying changes in competitive interactions (Hughes 2003). This will change vegetation community composition and structure, with knock-on effects for fauna. While the physiological limits of most fauna are unlikely to be exceeded in the study area, climate change is likely to cause changes in the distribution (range shifts southward and towards higher altitudes), behaviour (e.g., thermal regulation, foraging), movement patterns, community composition, host-pathogen dynamics, resource availability (e.g., nectar, fruit, insect emergence) and timing of critical processes (e.g., breeding, migration, spawning) of animal populations (Brereton et al. 1995, Chambers et al. 2005). From a restoration perspective, it is critical that areas are set aside or restored to allow for climate-induced migration or distribution shifts, highlighting the need for altitudinal and latitudinal biolinks (Mansergh et al. 2006).

3 Landscape Restoration: Definition and Objectives

Ecological restoration is an intentional activity that initiates or accelerates the recovery of a natural ecosystem that has been degraded, damaged, or destroyed, usually as a consequence of human activities (SER 2004). Ecological restoration can occur at a variety of spatial scales but for maximum benefits should be approached from a landscape perspective, one that explicitly recognises and is concerned with maintaining or restoring interactions and flows across adjacent ecosystems or elements of the landscape. We refer to ecological restoration at this scale as *landscape restoration*. Ecological restoration involves manipulation of abiotic and biotic components of the environment (i.e., management intervention) and aims to return a degraded system to its unimpaired state (Whisenant 1999). Thus, *restoration* overtly attempts to recover a pre-existing condition close to the original state, although this will rarely be possible in practice. The related practice of *rehabilitation* also seeks to improve the condition of degraded areas to resilient, self-supporting ecosystems but not necessarily in the direction of the pre-existing state (Bradshaw 1997). The allied concept of *conservation* relates primarily to the protection of existing natural areas and reserve design (e.g., Pressey and Nicholls 1989, Pressey et al. 1997, Bennett and Mac Nally 2004).

Restoration ecology – testing ecological theory through restoration projects and developing new theory specifically to repair damaged ecosystems – is the science that informs the applied practice of ecological restoration (Palmer et al. 1997, Lake 2001, SER 2004). Restoration ecology is a relatively new sub-discipline of ecology that developed from minesite rehabilitation (e.g. Nichols and Watkins 1984, Nichols and Bamford 1985), post-logging recovery and habitat manipulation for the management of game species (Scott et al. 2001). Early restoration efforts concentrated on site-specific processes, such as re-establishment of vegetation and re-introduction of species of interest in damaged sites. While this pioneered many technical methodologies, success was limited by a failure to incorporate ecological principles into restoration projects, particularly when the focus shifted from degraded sites within largely intact ecosystems to sites in highly modified landscapes or the repair of entire sub-catchments. Community ecology contributed understanding that is now well established in restoration practice (Palmer et al. 1997). For example, an understanding of community succession, species facilitation and mutualism, and the importance of natural disturbance regimes have increased restoration success (Young et al. 2005). The emergence of landscape ecology – the study of spatially heterogeneous land mosaics and the interactions between landscape structure and function as they change over time (Forman and Godron 1986) – complemented the need for a broader spatial perspective in restoration ecology. Integration of landscape ecology has addressed issues relating to natural recruitment of propagules to restored sites, flows and disturbances across patch boundaries, edge effects, landscape context and the role of the surrounding ‘matrix’ on restoration success (Bell et al. 1997).

That landscape restoration actively seeks to restore pre-existing conditions (biotic integrity and ecological processes) introduces the fundamental issue of setting restoration goals. Defining restoration (or rehabilitation) goals and constructing a ‘landscape vision’ are fundamental to the planning, implementation and success of restoration programs (Wyant et al. 1995, Hobbs and Saunders 2001, Lake 2001). There are two parts to this process: first, a reference site(s) or condition must be selected (what is a desirable end-point?) and second, indicators for that condition must be developed (when has restoration been successful?). The two most common sources of reference information for goal setting are historical data from the same site or contemporary data from reference sites that are assumed to match the environmental and biotic attributes of the restoration site prior to degradation (White and Walker 1997). That is, reference sites have high *ecological integrity* in that they maintain their structure, species composition and disturbance regime solely through natural processes (Brussard et al. 1998).

Spatial and temporal variation in nature means that finding identical reference sites is rarely possible but nor should it be expected. Moreover, given that degrading processes are rarely randomly distributed in ecological space, landscapes in need of restoration are unlikely to have pristine analogues. Therefore, it is prudent to devise restoration goals that incorporate uncertainty in ecological variables across space and time; that is, by combining information from multiple reference sites, restoration goals should reflect a range of ecological conditions rather than a single reference condition. Ecological variation is usually spatially and temporally correlated, so reference sites should be sought as close to the restoration site as is feasible, in terms of geographic distance and time since disturbance (White and Walker 1997). Lunt and Spooner (2005) contend that historical anthropogenic land uses fundamentally change the biophysical environment, resulting in new ecosystems that differ in vegetation structure, species composition and function, constraining the range of possible end-points. Under these circumstances, it is neither appropriate nor practical to aim to restore original conditions, and more realistic goals focusing on repairing ecosystem processes should be set (Whisenant 1999, Hobbs and Harris 2001).

The choice of indicators will depend on the goals of restoration. For single-species programs, indicators should monitor population parameters of the species of interest, and may include survival, reproductive success, recruitment, foraging success, range expansion or population size. Selection of indicators for community or ecosystem restoration is more complex. Species diversity (or richness) of the taxonomic group of interest is commonly used, although plants and invertebrates are often used as general indicators or surrogates for other taxa (Ruiz-Jaen and Aide 2005). It is recommended that the diversity of several taxonomic groups

across different trophic levels are monitored, even though recent research indicates that subsets of indicator species may efficiently predict up to 83% of the variation in combined species richness of multiple taxonomic groups (Fleishman et al. 2005). Simply monitoring species richness or diversity may conceal significant change or fluctuations in species composition of communities (e.g., McDougall and Morgan 2005). Thus, indicators based on community structure (species composition) are informative. Studies of aquatic flora and fauna have long used community structure as an indicator of river health in impact assessments (e.g., Blinn and Bailey 2001) and latterly for evaluating restoration success (e.g., Bond and Lake 2003, Bond et al. 2006). Indeed, Su et al. (2004) contend that patterns of species composition are more likely to co-vary across taxonomic groups (e.g., plants, birds and butterflies) at different sites than species richness. If the sensitivity of individual species to degrading processes is known, it is possible to generate 'sensitivity' indices based on community structure that reflect the level of anthropogenic impact [e.g., SIGNAL index for aquatic macro-invertebrates (Chessman 1995, Chessman et al. 1997); Bird Integrity Index for riparian vegetation (Bryce et al. 2002) and land use management (Glennon and Porter 2005)]. Recovery from a degraded state towards the reference condition could be monitored using indices of this type. Sensitivity indices provide appealing measures of community-level condition and function, but they are data-intensive to develop, susceptible to the ambiguities inherent in composite indices, may not be geographically transferable, and proximate causal relationships are not always clear.

A common approach is to use *indicator species* as surrogates for species richness or ecological integrity. The choice of which species to use as indicators is complex and restoration practitioners are often perplexed by the array of terminology and approaches associated with indicator species. In its simplest form, an indicator species is a species whose 'presence and fluctuations reflect those of other species in the community' (Simberloff 1998). However, use of a single indicator species is unwise because there is difficulty in determining what the species should indicate (e.g., species richness, community structure or ecosystem processes), how to choose an appropriate indicator and whether it is representative of the wider community (Landres et al. 1988, Simberloff 1998). Selection of a set of indicator species may improve performance, at least for predicting species richness of a taxonomic group (Fleishman et al. 2005). Threat-orientated indicator species may be useful for identifying environmental change (e.g., species sensitive (or tolerant) to chemical toxicity or logging activity) but there is little evidence that they represent a large number of species (Simberloff 1998).

The indicator species concept has spawned several distinct approaches that are often (erroneously) used synonymously, creating angst among scientists and confusion among restoration practitioners. An *umbrella species* is a species whose conservation confers protection to other naturally co-occurring species (Roberge and Angelstam 2004), and classically refers to the minimum area requirements of a population at the patch (minimum patch size) or landscape (minimum habitat cover) scale. The concept assumes that if the resource (usually area) requirements of the umbrella species are protected or restored, the requirements of a large number of species will simultaneously be met. Evaluation of area-limited umbrella species in conservation planning provides little support for their effectiveness (Roberge and Angelstam 2004). One reason for this is that umbrella species were traditionally chosen because they were threatened or endangered, not because they were good surrogates for the entire community.

The *focal species* approach (Lambeck 1997) attempts to overcome this by explicitly linking surrogate species with ecological processes, based on quantitative data. The focal species approach involves identifying the threatening processes in a landscape, identifying the species most sensitive to each threat (a focal species) and managing each threat at a level that will protect the associated focal species (Hobbs and Lambeck 2002). Lambeck (1997) suggests four threat categories should be considered for each habitat type: species limited by patch area, dispersal (patch isolation), resources (habitat condition) and processes (e.g., fire regimes). The result is a multi-species umbrella consisting of a set of focal species (one for each threatening process in each habitat type) whose requirements are assumed to include those of all other less sensitive species in the landscape. Management is then designed to meet the requirements of the focal species. The focal species approach has been criticised on the grounds that it

requires exhaustive field-sampling (which may delay urgent restoration actions), incomplete data risks misidentifying the *most* sensitive species, thorough application would require a large number of focal species rendering it inefficient, species-specific responses to fragmentation and degradation are contrary to the assumptions of surrogacy, focal species may not adequately represent other taxonomic groups (e.g. reptiles, invertebrates), small patches are under-valued, and it focuses exclusively on occupancy patterns without addressing population viability (Lindenmayer et al. 2002, Lindenmayer and Fischer 2003, Bennett and Mac Nally 2004).

Keystone species have functional impacts disproportional to their abundance or biomass (Lyons et al. 2005). For example, the loss of top predators may lead to an increase in herbivores and loss of plant diversity and environmental degradation through overgrazing. Watson (2001) contends that mistletoes are keystone species in Australian woodlands because they provide an array of resources for many other species (e.g., nectar, fruit, foliage, nest sites), and has shown that woodlands without mistletoes may have lower bird diversity (Watson 2002). *Ecosystem engineers*, species that directly modulate the availability of resources to other species by causing physical state changes in biotic or abiotic materials (Lawton 1994), are a special case of keystone species. Beavers *Castor canadensis* are archetypal ecosystem engineers, building dams that alter the flow and habitat of rivers (Naiman et al. 1986). In Victorian woodlands, the contribution of termites to the development of tree hollows qualifies them as both ecosystem engineers and keystone species. Simberloff (1998) suggests managing for keystone species may combine elements of ecosystem and species-based approaches "to the extent that the keystone is functionally crucial to a suite of other species, its management may maintain them." Simberloff (1998) continues that even if management of the keystone species itself were difficult, understanding the functional mechanisms of the keystone would increase understanding of the ecosystem, facilitating its overall management. However, several factors limit the utility of keystone species as indicators, although an understanding of their functional roles may be critical for successful restoration. Keystone species are difficult to identify, especially if they are rare and their function is not apparent until they are lost from the ecosystem. Moreover, it is not certain that all ecosystems have keystone species. Although many species depend on keystone species and their functional impacts, most species will have additional requirements that are not met by the keystone species. Consequently, managing for the keystone species alone will rarely be enough to ensure the survival of the dependent species. For example, although owls require hollows, managing for termites will not ensure owls are present.

A *flagship or icon species* is a charismatic species, usually a large mammal or bird, used to raise public awareness and galvanise support for a particular course of action (Simberloff 1998, Nickoll and Horwitz 2000). For example, the Red-tailed Black Cockatoo *Calyptorhynchus banksii graptogyne* was adopted as the mascot of the Melbourne Commonwealth Games to attract attention to its decline and the need for responsible environmental management. There is not necessarily an ecological reason for the choice of flagship species, except that they are often endangered, and they need not be a good indicator or surrogate species. Increasingly, however, flagship species are being selected on the basis of both public appeal and because they are indicators of ecological integrity (e.g., Murray Cod *Maccullochella peelii*; Malleefowl *Leipoa ocellata*).

Native vegetation is perhaps the most commonly used indicator for biodiversity (Williams 2005, Ruiz-Jaen and Aide 2005). A host of metrics have been developed to quantify the amount and arrangement of native vegetation at the landscape scale (e.g., Turner and Gardner 1991, Hargis et al. 1998, Bender et al. 2003). While many studies have demonstrated the influence of various aspects of landscape structure on components of biodiversity (e.g., Downes et al. 1997, Major et al. 2001, Radford et al. 2005), landscape metrics are relatively coarse indicators of trends in biodiversity, seldom allowing accurate and specific predictions. Nevertheless, percent vegetation cover remains one of the most powerful and frequently used indicators of biodiversity. At the site or patch scale, vegetation structure (e.g., stem density, height, diameter, number of strata) and plant diversity is frequently used to monitor restoration success. Vegetation structure and complexity is a key determinant of faunal diversity providing a sound ecological basis for the assumption that greater structural diversity equates to increased biodiversity (e.g., Hadden and Westbrooke 1986, Bennett

1990, Friedenberger 2001, Williams et al. 2002). A recent emphasis has been on combining measures of landscape structure with patch-level vegetation attributes in a single metric, as for example in the habitat hectares (Parkes et al. 2003) or biodiversity benefits index (Oliver and Parkes 2003) approaches. Remote sensing is now being employed to quantify the extent and condition of native vegetation across large spatial scales (e.g., Bastin et al. 2002, Newell et al. 2006).

The 'ecosystem management' approach (see 5.5 below) focuses on ecological processes and ecological systems as a whole rather than the identity of species, species richness or species composition (Brussard et al. 1998, Whisenant 1999). Indicators such as primary productivity, nutrient cycling, organic decomposition, pollination, seed dispersal, herbivory or parasitism are used to gauge restoration success (Ruiz-Jaen and Aide 2005). The ecosystem management approach has been applied in two slightly different ways. The first assumes that if the full spectrum of ecological processes is functioning properly then ecosystem resilience will be high and the natural composition and diversity of biodiversity will be present (Knight 1998): that is, the processes themselves are viewed as surrogates for biodiversity. Proponents put forth evidence that ecological functioning is impaired as species are eliminated from ecosystems (e.g., Lawton 1994, Tilman 1997). A fundamental reason for this is that different species often function optimally under different environmental conditions. Thus, uncommon species that contribute little to ecosystem functioning under current conditions may play critical roles following particular environmental triggers, such as successional change, natural disturbances, climatic variation or external shocks (Lyons et al. 2005). Similarly, rare species may also be important for ecosystem resilience (recovery after disturbance) or the resistance of a community to invasion by exotic species (Lyons et al. 2005). However, functional redundancy, whereby numerous species perform the same functional role, has also been demonstrated, such that the relationship between species richness and ecological processes reaches an asymptote and a minimum set of species allows proper functioning (Palmer et al. 1997). Current consensus is that full functioning (for a particular ecological process) can usually be obtained with 10-15 species but that the presence of different functional groups (*functional diversity*) is an important feature of functioning ecosystems (Young et al. 2005). However, Elmqvist et al. (2003) caution that greater diversity within a particular functional group increases ecosystem robustness because variability in species' responses to external disturbances decreases the risk that the function will be entirely lost from the system.

The second way ecosystem management has been applied is to assume that as key processes are restored through physical or biotic manipulation, improvements in biodiversity will follow (Whisenant 1999). However, this approach does not contend that the full roster of species must be present for ecosystems to function properly; that is, restoration of ecological processes is an end in itself, improving landscape condition and in most situations laying the foundation for increases in biodiversity. For many conservation biologists, a danger here is that ecosystem processes become more important than species composition, such that processes may be maintained or restored without the full complement of native species (or achieved with exotic species), and biodiversity losses are still incurred (Knight 1998, Simberloff 1998). While ecological processes may indeed be authentic indicators of ecosystem resilience and integrity, their adoption remains problematic for several reasons. First, knowledge of the system is often insufficient to determine how many and which processes should exist. Second, many processes are difficult to measure and monitor. Third, end-points for processes are just as arbitrary as for species measures – how much is enough? Finally, it remains unresolved as to whether ecological function begets biodiversity or vice-versa? For the restorationist, perhaps this is an artificial distinction and the pragmatic approach is to manage for both, or whichever is receptive to manipulation given the constraints and opportunities of a given landscape.

Effective landscape restoration entails more than using ecological theory to inform on-ground actions: the drivers of land use change - social, economic and political - must also be addressed. Harnessing societal motivations for restoration (Clewel and Aronson 2005), economic cost-benefit analyses (Holl and Howarth 2000, Hajkovicz and Young 2002), developing equitable policies that integrate restoration and production (Qureshi and Harrison 2002, Brennan 2004), engaging the community and facilitating uptake of

new ideas and approaches (Pannell et al. 2006) are all part of the restoration process that are reviewed in other publications. Conjecture surrounds the role of science in these processes. Turner (2005) posits that through detailed observation, experiments and critical thinking, science brings clarity to restoration by increasing understanding, leading to improved efficiency in restoration efforts, confidence in proposed outcomes and acceptance of uncertainty. Winterhalder et al. (2004) insist science is integral to identifying degraded landscapes, selecting reference conditions and defining restoration goals, arguing that effective decisions and policies must be based equally on ecological, economic and social imperatives. Similarly, Brennan (2004) sees a role for scientists not only in defining environmental issues and providing technical expertise but also in policy development and articulating solutions to complex problems. In contrast, Davis and Slobodkin (2004) argue that setting restoration goals is fundamentally driven by personal and social values, whereas the science of restoration ecology is integral to the implementation of restoration actions. They contend that restorationists compete with other stakeholders in the community using social, ethical, economic and cultural arguments to justify their stance, and that recognising and embracing the value-based nature of goal setting will increase the effectiveness of restoration projects. Turner (2005) suggests that ecological restoration and social restoration are reciprocal: that there is a positive feedback between protection and restoration of the environment and the 'health' of a society. Turner (2005) cites an analysis of wetland management in 90 countries by LaPeyre et al. (2001) that emphasised the importance of social development (measures of health and education) and open and inclusive government for 'successful' wetland protection and restoration, concluding that "both good science and social capital are essential elements of restoration success".

4 A Conceptual Framework for Landscape Restoration

In northern Victoria, the necessity for landscape restoration is widely, although not unanimously, acknowledged. Scientists, extension staff and catchment planners have contributed to this perception through raising awareness of environmental issues; for example, declines in populations of native flora and fauna, dryland salinity, waterlogging and soil erosion. Farmers have also played a key role in recognising environmental problems, particularly those pertaining to ecosystem processes that directly affect their enterprises. However, there is often a gulf between acknowledging a need and acting upon it. Effective restoration needs to address the ecological and biophysical processes that underpin functioning landscapes; science is pivotal for both recognising problems and articulating solutions. Ultimately, however, people undertake restoration, and thus, social and economic factors will shape the type, extent and success of restoration. A conceptual framework that integrates these disciplines lays the foundation for effective restoration.

The development of a conceptual framework for ecological restoration has received considerable attention from academics (e.g., Wyant et al. 1995, Hobbs and Norton 1996, Palmer et al. 1997, Brussard et al. 1998, Hobbs and Harris 2001, Lake 2001). A set of core principles has emerged from these discussions that together comprise a general framework for restoration applicable in most landscapes.

- i. Define the ecosystem or landscape to be restored. This involves defining biophysical boundaries (e.g., bioregion, vegetation community, riparian zone, sub-catchment or cluster of adjoining properties), the social landscape (e.g., pastoralists, dairy farmers, cereal farmers, hobby farmers, or lifestyle landholders), and the politico-economic context (e.g., legislative and regulatory obligations, economic constraints and incentives, voluntary agreements).
- ii. Assess the current condition and trends of the ecosystem or landscape to be restored and identify natural and anthropogenic disturbances. This will require indicators of ecological and biophysical condition to be developed. This identifies components of the system that require restoration and degrading processes that need to be reversed.

- iii. Construct a 'landscape vision': a set of specific, ecologically-informed and feasible restoration goals that take into account the current state of the system. The landscape vision should clearly specify goals for biophysical processes (e.g., hydrology, soil condition, nutrient cycling), fauna-mediated processes (e.g., pollination, animal dispersal / movement, genetic diversity), species recovery (e.g., threatened or declining species), community integrity (e.g., soil fauna, cryptogams, flora, faunal groups) and landscape structure (e.g., extent of native vegetation, desirable patch sizes). Restoration in agricultural landscapes will never recreate 'pristine' or reference ecosystems; therefore, goals should focus on the desired characteristics of future landscapes rather than attempting to recreate pre-disturbance conditions. It is important to seek input from a range of stakeholders when constructing the landscape vision.
- iv. Articulate a set of restoration actions that link the current state to the landscape vision (i.e. *how* to move from the current system to the desired system). Actions should be based on sound ecological knowledge and draw on local knowledge, where available. Actions must consider social and cultural context, cost of restoration, methods of payment, risk assessment and technical aspects of the proposed restoration.
- v. Establish transparent and measurable success criteria based on relevant ecological and biophysical indicators. Success criteria (indicators) must reflect the restoration goals, be responsive to restoration actions and lack ambiguity, and ideally, be relatively easy and economic to sample.
- vi. Implement restoration actions in a manner consistent with adaptive management. Restoration projects should 'build in' opportunities for testing theory and learning, and feedback mechanisms for adjusting restoration activities contingent on restoration outcomes. This should include, where possible, collection of baseline (pre-restoration) data, replication of the restoration action at independent (treatment) sites, establishment of control (degraded, not restored) and reference (not degraded, not restored) sites, an unbiased sampling regime, and consistent application of restoration actions.
- vii. Monitor indicators at scales appropriate for the restoration actions. This requires consideration of spatial grain (sampling unit) and extent (area over which sampling is conducted), and temporal frequency (interval between sampling events) and longevity of monitoring. Time lags between restoration actions and ecological responses are likely so it is important that monitoring has a long-term perspective. It is critical to also measure the restoration actions themselves (e.g., extent of revegetation, decline in area infested by weeds, river flows) and potentially confounding co-variables that may not be part of the restoration project (e.g., climate, exotic predators, regional factors) to establish causal relationships between restoration and ecological responses.
- viii. Adjust management based on cost-benefit assessment of restoration inputs (costs) and ecological and/or biophysical responses (benefits). Thus, an ongoing process of implementation – monitoring – evaluation – adjustment is established, with learning accrued from well-planned 'experiments' during each implementation phase. It is important to re-visit the landscape vision prior to re-setting restoration actions so that adjustments are in line with the restoration goals.

5 Application of Planning Approaches for Restoration in a Northern Victorian Context

Each of the Catchment Management Authorities (CMA) in northern Victoria have set targets for extent and condition of native vegetation and biodiversity that are detailed in their respective Regional Catchment Strategies (MCMA 2003, NCCMA 2003, NECMA 2003, McLennan et al. 2004). The CMAs have adopted an asset-based approach that identifies biodiversity assets and threatening processes, and then develops targets and actions (including implementation plans) to diminish the threats and enhance biodiversity and other land condition criteria (Table 1). The catchment strategies contain many of the elements outlined in the conceptual

framework above, including: (i) a defined domain of reference; (ii) summary of current conditions and assets; (iii) long-term (> 50 years) aspirational goals (i.e., a 'landscape vision') for each of the assets; (iv) medium term (10-30 years) targets for asset (resource) condition; (v) clearly defined management actions (1-10 years); and (vi) success criteria for management action targets. In the main, the catchment strategies are founded on sound ecological principles with appropriate, if ambitious, targets and goals. Further attention is required in developing ecologically responsive indicators, co-ordinating management actions in an adaptive management framework, developing low cost techniques that will allow large areas to be revegetated, and strategic monitoring. However, the most challenging task is planning and implementing on-ground actions such that the regional targets enshrined in the catchment strategies are achieved.

Management actions are usually implemented at spatial scales smaller than those used for setting targets. This mismatch of scales may mean that site-level actions are delivered without reference to the regional context, so the full biodiversity potential of restoration work is not realised. What approaches can be used to ensure the integrity and spirit of the regional targets are maintained when transferred to smaller scales, and that management (conservation, restoration or rehabilitation) effectively achieves the desired biodiversity benefits? Bennett and Mac Nally (2004) reviewed approaches for determining priority areas for conservation and restoration, and grouped them into use of general ecological principles, species-based approaches, quantitative methods for assessing representativeness (i.e., multi-species optimisation), and landholder-driven 'bottom-up' actions. Here, we ask how might these (and other) approaches be used to guide restoration planning at the property or sub-catchment scale in northern Victoria? As pointed out by Bennett and Mac Nally (2004), these approaches should not be considered mutually exclusive: adopting one approach does not prohibit using others. It may be best to adopt a mix of strategies, depending on restoration objectives, resource constraints and existing natural capital and land uses.

5.1 Species-based approaches

Species-based approaches (e.g., indicator species, focal species, keystone species, flagship species – see section 3) are appealing because they identify tangible foci for restoration efforts to which most people can relate: a rallying point for community involvement and agency funding. There are many examples of restoration projects in northern Victoria that revolve around threatened species (e.g., Superb Parrot *Polytelis swainsonii*; Regent Honeyeater *Xanthomyza phrygia*; Grey-crowned Babbler *Pomatostomus temporalis temporalis*; Malleefowl *Leipoa ocellata*; Carpet Python *Morelia spilota metcalfei*; Eltham Copper Butterfly *Paralucia pyrodiscus lucida*; Spiny Rice Flower *Pimelia spinescens*; threatened orchids *Caladenia* spp.). These represent flagship species because the projects have been specifically designed to meet the requirements of the target species without consideration of surrogacy values, although benefits often incidentally flow to other species. These projects have been successful in attracting funding and/or community involvement and restoring habitat; however, success in terms of reversing population declines in the target species (let alone other species) has been mixed. The capacity of government agencies, non-government organizations and communities to undertake such intensive restoration projects falls well short of the number of species in need of assistance in northern Victoria. Thus, while flagship species play a valuable role in connecting people with the environment, the efficacy of this approach for conserving or restoring biodiversity in the broader context is questionable. While it is not possible to conduct population viability modelling for all species, an efficient use of resources may be to objectively assess extinction risks for a suite of non-target species under different restoration scenarios based on population viability assessments for several flagship species (Lindenmayer et al. 2003, Nicholson et al. 2006).

The focal species approach was developed in response to the piecemeal approach of single-species methods (Lambeck 1997). Brooker (2002) provides a comprehensive demonstration of the focal species approach for landscape design in the Gabbi Quoi Quoi sub-catchment (~300 km²) in the Western Australian wheatbelt. She identified four focal species in seven threat categories, and then based on existing landscape structure and soil conditions, identified priority areas for revegetation. Priority areas were designed to

increase the size of small remnants above the threshold defined by the patch size focal species (i.e., 'area-limited') in areas between extant remnant vegetation, but within distances defined by the isolation threshold of the dispersal focal species (i.e., 'distance-limited'). This produces a map identifying priority areas for revegetation but does not specify *how much* revegetation is required. The focal species approach has merit in that it is evidence-based, considers multiple species and multiple threats including processes, and is spatially explicit. The quantitative targets (e.g., for patch size and isolation distances) also hold intuitive appeal for landholders who are therefore more inclined to undertake on-ground restoration activities. The social hook (*sensu* Lindenmayer and Fischer 2003) of an animal that a landholder is familiar with, or at least can see in a field guide, is a powerful tool that should not be underestimated. Complete focal species analyses are rarely undertaken given the considerable amount of data that needs to be collected (see section 3 and below). The ability of this approach to encourage on-ground action may therefore be one of its stronger points.

In theory, this approach could be conducted in northern Victoria but there are questions about its efficiency as a planning tool. Several local area plans have been developed in northern Victoria, purportedly using the focal species approach. However, rarely have these plans been based on survey data of sufficient sampling intensity (coverage or sample size) to reliably estimate patch size or dispersal thresholds, let alone identify focal species for other threatening processes (*cf.* Western Australian examples in Lambeck 1999, Brooker 2002). An incomplete inventory of species' requirements renders the focal species approach impotent in its intended form – sensitive species may be missed, thresholds may be incorrect and threatening processes not recognised. It would be preferable to develop pseudo-focal species using expert opinion but this could be criticised for lacking scientific rigour. In some parts of northern Victoria, it may be possible to meet the data requirements of the focal species approach if a database could be established to consolidate all sources of survey data (e.g., Birds Australia atlas surveys, research projects, consultants' surveys, monitoring projects). Supplementary surveys could then be commissioned to fill gaps. However, the limitations of using focal species may not justify such expense. First, the results of a comprehensive focal species analysis are unlikely to be socially acceptable or practical, which may stifle restoration action and generate community resentment. For example, viable populations of the Black-eared Miner *Manorina melanotis* require patches of continuous mallee vegetation exceeding 13,000 ha (Clarke et al. 2005). This is important information for *conservation* but is unreasonable for *restoration* in agricultural landscapes. Redefining the focal species until an 'acceptable' species is found (e.g., one with smaller patch size requirements) compromises the integrity of the whole approach such that it becomes a flagship species rather than one grounded in empirical evidence. Second, the focal species approach is susceptible to unintentional abuse due to the lack of transferability of focal species between regions. For example, the Hooded Robin *Melanodryas cucullata* is commonly used as a patch size focal species, with a minimum patch size requirement of 100 ha cited from studies in ACT/NSW (Freudenberger 1999, Watson et al. 2001). Yet, Hooded Robins are frequently present in small (<10 ha) patches of *Allocasuarina* woodland in western Victoria (Maron and Lill 2005) and *Casuarina* woodland in the Mallee (JQR, personal observation), and elsewhere occur in revegetation sites as small as 1 ha (Taws 2001). Applying a minimum patch size of 100 ha risks undervaluing smaller patches and missing opportunities for effective restoration in smaller patches. A third impediment to applying species-based approaches to individual (or a cluster of) properties relates to spatial scale. There is always variation and uncertainty in occupancy patterns of native species. Landholders may become disheartened if adhering to restoration prescriptions fails to attract the target species due to variation in species distributions. Further, it is misleading to consider focal species or flagship species for individual properties – the scale of operation of these methods is considerably larger – although individual properties obviously contribute to regional plans and objectives.

Restoration of keystone species is a promising approach for relatively small scales because the focus on ecological processes is likely to derive substantive benefits for a wide range of species, although this may not include charismatic vertebrates. While all approaches that involve surrogacy carry risk that the anticipated collateral benefits do not occur, approaches that focus on processes are more likely to address underlying causes rather than symptoms of a dysfunctional landscape. In many cases, restoration actions

directed towards the keystone species will have collateral benefits for other species (i.e., an umbrella species effect), as well as functional benefits derived from the keystone species. The major obstacle, however, is lack of knowledge about which species (or group of species) are pivotal to particular ecological processes (e.g., ectomycorrhizal fungi for nutrient cycling (Tommerup and Bougher 1999), key pollination vectors (Paton 2000, Paton et al. 2004), organic decomposition agents, symbiotic relationships). If this can be determined, restoration actions can be directed to improve conditions for keystone species or inoculate restoration sites with keystone species, stimulating improvements in ecosystem processes, and subsequently, biodiversity.

5.2 Multi-species optimisation

The species-based approaches discussed in 5.1 assume that meeting the requirements of one or more indicator species will satisfy many other species as well. In contrast, multi-species optimisation is not predicated on surrogacy; rather it considers each species in its own right and attempts to identify relatively small areas with high species richness. This may include discrete sites (i.e., 'biodiversity hotspots') or complementary sets of sites that together represent the full range of biodiversity, or environmental variation, within a defined region in the most efficient manner (Margules et al. 1988, Williams et al. 1996). For example, Ceballos et al. (2005) recently used complementarity techniques to identify 11% of the Earth's land surface that together represent 10% of the geographic range of all terrestrial mammal species. Complementarity analyses may be based on records of species occurrence at specific survey locations (e.g., Arponen et al. 2005, Radford and Bennett 2005), species range maps (e.g., Hulbert and White 2005), or potentially, maps of habitat suitability derived from spatially explicit habitat models (e.g., Ferrier et al. 2002, Guisan and Thuiller 2005).

For restoration planning, combining habitat models with optimisation algorithms in a GIS framework to identify areas that are potentially suitable for a large number of species may prove very effective. Conceptually, this involves overlaying maps of projected habitat suitability under various restoration scenarios and choosing the option(s) that satisfies the habitat requirements of the largest number of species. Filters could be applied to choose options most befitting groups of species of special interest. The benefits of multi-species optimisation are: (a) it does not invoke surrogacy – the species predicted to benefit from restoration are identified directly from habitat modelling; (b) predicted outcomes are explicit and specific, improving the capacity to assess competing restoration options; and (c) habitat models can include a variety of predictor variables such that factors affecting occurrence can be accurately identified for each species. However, there are some significant limitations to this approach. Multi-species optimisation is only as good as the underlying data, and whether based on distributional records or habitat models, this approach also requires extensive data collection. Insufficient data will mean the models have poor predictive ability and large confidence intervals, which reduces their value for assessing restoration options. The complex statistical modelling and GIS optimisation procedures require expert skills and resources to implement, which are not readily available to all agencies involved in restoration. This is not an approach that can be sketched out in the paddock or a single community workshop! Despite the sophisticated methods, models will usually be derived from occurrence data, so may not accurately reflect population persistence. Recent advances in reserve selection algorithms designed to minimise extinctions across multiple species, based on population viability models, suggest this may be overcome (Nicholson and Possingham 2006, Nicholson et al. 2006). Finally, the habitat models may simply not be accurate for restored habitat if it differs in vegetation structure and composition from remnant vegetation.

5.3 Ecological principles

Ecologists are torn between the demand for producing universal quantified guidelines for restoration and the knowledge that the complexity of natural systems and species-specific responses means it is unlikely such guidelines will be accurate in every situation (Hobbs and Yates 1999). It is foolhardy and counter-productive to give prescriptive restoration guidelines in the absence of specific

information. However, agency staff and landholders actively involved in restoration desperately seek such information. An alternative is to couch recommendations in relative terms (trends, ranks and gradients) based on generally accepted ecological principles that are applicable across a range of situations (Bennett and Mac Nally 2004). This allows contingencies specific to each restoration project to be considered within the context of the general principles to arrive at the 'best management practice'. It is noteworthy that many of Australia's foremost landscape ecologists advocate landscape design based on general ecological principles (e.g., Saunders et al. 1991, Wilson and Lindenmayer 1995, Hobbs and Yates 1999, Bennett et al. 2000, Ive and Nicholls 2001, Platt 2002, Bennett and Mac Nally 2004, Soulé et al. 2004, Fischer et al. 2006).

Ecological principles often relate to landscape design at the patch and landscape scale. For example, Wilson and Lindenmayer (1995) outlined 17 'design principles for the development of corridors and corridor networks', including 'that corridors be designed to provide both suitable habitat for wildlife and to maintain and/or enhance connectivity between remnant populations of plants and animals' and 'that a set of key design principles be established for corridor development, including minimising edge effects, minimising the impact of disturbance from surrounding land use practices, recreating the complexity of vegetation structure and plant diversity, minimising gaps between remnant and planted patches and ensuring continuity'. Bennett et al. (2000) used general principles to develop recommendations for restoration at a range of spatial scales, from the site level to the regional level (Table 2). Similarly, the Heartlands project, run by the Murray Darling Basin Commission and CSIRO, returned to first principles to construct a framework for its restoration activities. By outlining 15 ecological design principles (Ive and Nicholls 2001) restoration activities are grounded in "best-ecological practice". These principles focus on re-establishing native vegetation in the landscape, and include issues such as local provenance for seed-stock, re-establishing original ecological entities, multiple representation of ecological entities, re-establishing functional groups within remnants and on-going maintenance. Fischer et al. (2006) outline five general principles relating to landscape 'pattern' (i.e., maintain large, structurally complex patches; maintain structural complexity in the 'matrix'; create buffers around sensitive areas; maintain or create corridors and stepping stones; and maintain landscape heterogeneity) and five pertaining to 'processes' (i.e., maintain key species interactions and functional diversity; apply appropriate disturbance regimes; control invasive species; minimize threatening ecosystem-specific processes; and maintain species of particular concern) to improve biodiversity, ecosystem function and resilience in agricultural landscapes.

A significant advantage of adopting ecological principles is that on-ground practitioners can approach restoration with the confidence that their plans are based on best available science, even without extensive background ecological data. Of course, if data exists for a particular location, restoration plans can be more detailed and customized, increasing the probability that restoration will be successful. There are limitations associated with using ecological principles (Bennett and Mac Nally 2004): (a) they lack specificity about which species will benefit and by how much; (b) they lack specificity about the magnitude of the action required; (c) they may not be transferable between geographic areas or ecosystems; and (d) they can confuse on-ground practitioners with the range of options (paralysed by choices). However, without more specific data or instruction, abiding by the principles contained in the documents listed above is more than likely to improve landscape condition and enhance biodiversity.

5.4 Passive wildlife restoration

There is mounting evidence that restored and revegetated sites are colonised and used by native fauna to some extent. A growing list of studies have documented the occurrence of birds (Ryan 1999, Taws 2001, Arnold 2003, Paton et al. 2004, Kavanagh et al. 2005, Jansen 2005), mammals (Nichols and Nichols 2003, Law and Chidel 2006), reptiles (Nichols and Bamford 1985, Webb and Shine 2000, Kanowski et al. 2006), frogs (Nichols and Bamford 1985, Hazell et al. 2004) and insects (Bonham et al. 2002, Catterall et al. 2004, Cunningham et al. 2005) in restored sites or mixed-species plantations. We assume that restoration benefits fauna either directly by increasing the extent of habitat in the landscape, size of patches or landscape connectivity, or indirectly by buffering

remnant vegetation from the contextual effects of surrounding land uses. However, there is actually very little data about how different taxa are using restored sites. Are fauna breeding in restored sites? Are fauna resident in restored sites all year round? Are there sufficient resources to support a self-sustaining population?

How do fauna reach restored sites? It is often simply assumed biodiversity increases will follow restoration owing to 'passive wildlife re-colonisation': animals will flow down a density gradient from surrounding habitat into restored sites (Scott et al. 2001). However, passive re-colonisation of restored sites requires careful planning. Individuals must be able to move from the source populations, through the landscape to the target (restored) sites. This requires consideration of connectivity (Bennett 1999) and corridors (Saunders and Hobbs 1991, Beier and Noss 1998), mosaic permeability (McIntyre and Barrett 1992, Fischer et al. 2005), metapopulations (Hanski and Gilpin 1991), conspecific attraction and social facilitation (Stamps 1988, Muller et al. 1997), source-sink demographics (Pulliam 1988) and density-dependent habitat selection (Fretwell and Lucas 1969) to increase the likelihood of colonisation following restoration. Further, the temporal sequence of restoration actions will also influence the probability of re-colonisation. Passive wildlife re-colonisation compels restoration planners to focus on processes such dispersal, demographics, habitat selection and social interactions. Restoration projects that lack the strategic planning necessary to facilitate passive re-colonisation may result in restored habitat without any fauna and fail to achieve the desired biodiversity benefits.

5.5 Ecosystem management

Ecosystem management explicitly recognises the role of humans in the management of ecosystems but emphasises a holistic approach that focuses on ecological systems and processes rather than a reductionist view of the component parts. Brussard et al. (1998) define ecosystem management as:

“managing areas at various scales in such a way that ecosystem services and biological resources are preserved while appropriate human uses and options for livelihood are sustained. Ecological services are biological, physical, and chemical processes that occur in natural or semi-natural ecosystems and maintain the habitability of the planet. The major services are allocation of energy flows, maintenance of soil fertility, and regulation of the hydrologic cycle.”

Insofar as the focus is on maintaining broad-scale ecological processes, this definition resonates with the WildCountry Project, which emphasises connectivity to promote and maintain seven key ecological processes: (1) trophic relations and highly interactive (*keystone*) species (e.g., higher-order predators, pollinators, decomposers, seed dispersers); (2) dispersal and migration of individuals and propagules; (3) natural disturbances (e.g., fire, flood, herbivory) at local and regional scales; (4) biotic adaptation to climate change; (5) hydroecology (i.e., the interaction between vegetation and surface and sub-surface water, and hence water availability to plants and animals); (6) coastal zone fluxes; and (7) evolutionary processes (i.e., potential for adaptation to changing environments and for speciation) (Soulé et al. 2004, Mackey et al. in press). Although these processes may not necessarily be visible at the farm scale, the underlying principles of managing for ecological processes are relevant and in many cases have production as well as biodiversity benefits. For example:

- Maintenance of predator-prey relationships provides pest management services in production systems. For example, leaf damage from herbivorous invertebrates was 3.5 times higher in the absence of insectivorous birds (Eveleigh et al. 2001); insectivorous microbats foraging in farmland may consume up to half their own body weight in insects per night (Lumsden and Bennett 2003); pesticide use in pasture can be reduced significantly where vegetation is nearby (Salt et al. 2004).
- Provision of shade and shelter by native vegetation reduces heat and cold stress leading to increases in milk production, weight gain and lambing success (Reid and Bird 1990, Blackshaw and Blackshaw 1994).
- Restoration of perenniality has the potential to increase water uptake and reduce recharge, ameliorating the effects of dryland salinity (Farrington and Salama 1996, White et al. 2001), although the effects of variability in climate, hydrogeology,

topography and land systems on the effectiveness of trees as a means of controlling salinity at landscape scales is yet to be fully understood (Passioura 2005).

- Flows of nutrients, pollutants and sediments from terrestrial to aquatic systems can be filtered by riparian vegetation, improving instream habitats, water quality and downstream sedimentation (Kimber et al. 1999, Salt et al. 2004).
- Local plantings can contribute to carbon sequestration, with new plantings capable of sequestering 7 to 10 tonnes of carbon per hectare per year (Wilson 2002).
- Vegetation buffers can reduce total loads of herbicide in run-off by up to 85%, and sediments by up to 93% (Popov et al. 2006).

How might ecosystem management be implemented on farms in northern Victoria? What are some of the critical processes for returning resilience and biological resources to the landscape?

Grazing management Historically, grazing management has consisted of set stocking rates in large paddocks for extended periods of time. This has favoured winter-active annual grasses because the palatable herbs and forbs are preferentially grazed without opportunities for recovery or regeneration, and summer-active grasses are displaced by annual grasses that grow rapidly in winter and spring. The addition of fertiliser and sowing introduced pasture species exacerbates this shift towards annual pastures. There is mounting evidence that strategic management of stock to mimic natural herbivory regimes (e.g., intensive 'crash' grazing between extended rest periods, seasonal variation in grazing) can restore native flora and increase perennality (Dorrrough et al. 2005, Davidson 2006, LWWNTF 2006, Handley 2006, Wong et al. 2006). This not only has biodiversity benefits but leads to a more resilient and heterogeneous grazing system, one capable of providing reliable summer-autumn fodder (in the form of summer-active C4 grasses) as well as winter-spring annuals. Increases in stocking rate and improvements in ground cover may also be achievable with strategic grazing (Kahn et al. 2005). The success of grazing management in facilitating natural regeneration of native flora depends on the soil characteristics (e.g., fertility, pH, moisture etc.), site history (regeneration potential is diminished with prior cultivation and fertiliser), proximity to seed sources (including status of the soil seed bank) and climate (Dorrrough and Moxham 2005, Dorrrough et al. 2006). In some cases, soil manipulation (e.g., ripping, harrowing) may improve success. The management skills of the grazier are also critical to success.

One tool that may be useful for monitoring resilience and functionality in concert with changes in grazing management is Landscape Function Analysis (LFA) (Ludwig and Tongway 1995, Ludwig et al. 1997). This protocol uses three sets of indicators (soil stability, infiltration and nutrient cycling) to identify processes regulating the availability of resources at relatively fine scales (indicatively, m² to ha). Zones of resource loss and resource gain are identified based on features that interrupt, divert or absorb runoff and transported materials. Originally developed for semi-arid rangelands, recent research suggests LFA is transferable between ecosystems and is applicable to temperate grassy ecosystems (D. Duncan, Arthur Rylah Institute, pers. comm).

Fire Fire is an important natural disturbance for both native plants and animals. Some plant species require fire to reproduce and fires release nutrients previously bound in living and dead plant material, which can stimulate a surge in growth. Fire also influences vegetation community dynamics, creating space, light and resources that may favour 'early successional' species, thereby increasing structural heterogeneity and plant species diversity. Whelan et al. (2002) found however that no one size fits all, and there are many possible responses of native plant and animal populations to fire. They identified four key stages of an organism's life cycle for manager's to focus on that may contribute to patterns of population change after fire. Although the 'natural' fire regimes in northern Victoria are largely unknown (Parr and Andersen 2006), it is safe to surmise that the fragmented landscapes that characterise northern Victoria are unlikely to support 'natural' fire disturbance regimes. Human intervention has generally resulted in either more frequent, low intensity fuel reduction fires or less frequent, more intense wildfires following long periods of fire exclusion.

Vegetation structure and species composition in northern Victoria has undoubtedly been affected by fire management: where fire has been suppressed (particularly grasslands), 'late successional' communities may be dominated by species that increase in the absence of fire, whereas in other areas, frequent fuel reduction burns may have eliminated some fire sensitive species and trapped the community in an early successional stage. Other habitat components are also influenced by fire patterns: litter, logs and debris on the ground, hollows, nectar resources, canopy cover, and seed production.

Incorporating biodiversity considerations into fire management has been hampered by insufficient knowledge about the response of flora and fauna to fire frequency, season, extent and intensity (Whelan et al. 2002, Olsen and Weston 2005). The functional response of different components of the biota to fire varies enormously, from species that require fire to regenerate (heat and/or smoke) to fire-sensitive species that only inhabit long unburnt vegetation (Gill et al. 1999, Bradstock et al. 2002, Clarke et al. 2005). Thus, fire management for biodiversity should seek to mimic the natural fire frequency and extent for the relevant vegetation community, avoiding uniform and inflexible prescriptions across the landscape. In the absence of more informed objectives, a 'patch mosaic burning' model has been promoted by fire ecologists - management should aim for a mosaic of different time-since-fire age classes, patch sizes and fire intensities, with prescribed burning conducted at an appropriate time of the year (Gill 1999, Bradstock et al. 2005). Despite the uncertainty that surrounds the scale of mosaic patches, the intensity at which they should be burned, the relative proportion of fire-age classes across the landscape, the interval and frequency of fires (the 'invisible' mosaic), and the juxtaposition of fire-age classes, the goal of creating a range of vegetation types of different fire-age classes (with some not burnt at all) remains the best advice for promoting diversity. However, Parr and Andersen (2006) warn that the ecological consequences of the 'pyrodiversity' paradigm of patch mosaic burning are rarely supported by empirical evidence specific to the system in which it is being applied, and that operational guidelines for implementation are poorly defined.

Hydrological flow patterns There are two key aspects of hydrological flow patterns relevant to ecosystem management. The first is retention of water in the landscape, particularly in the soil and in sub-surface flows. Water plays a critical role in many ecological processes, the most fundamental of which is uptake by plants and supporting primary production. Water is also an important movement vector for soil nutrients, bacteria, soil fungi, spores and seeds. In degraded landscapes, water moves rapidly through the system, reducing its availability for life-sustaining ecosystem processes, removing nutrients and sediment, and often scouring the landscape in the process (Greene 1992, Yates et al. 2000). The farm scale is appropriate for instigating ecosystem management aimed at slowing the rate of water movement, thereby retaining it in the landscape for longer. Management that increases the porosity of the landscape (e.g., increasing ground cover, reducing soil compaction, improving soil biota, increasing the irregularity or porosity of the soil surface) will slow the flow of water across the surface and increase infiltration into the soil (Eldridge and Freudenberger 2005). Even within drainage lines and stream channels, management actions to slow water movement (e.g., re-establishment of riparian and instream vegetation, re-snagging with logs and rocks) can provide fauna habitat and benefit water retention (Bunn and Arthington 2002).

The second aspect is restoring natural flow regimes, including the magnitude and timing of flood events, to streams, rivers and wetlands (Bren 1993, Lake 1995, Kingsford 2000, Nilsson and Svedmark 2002). It is critical that periodic flood events breach the stream channel to replenish nutrients, sediments, propagules and moisture in the floodplain, to fill ephemeral wetlands and to flush out billabongs (Pinay et al. 2002, Wiens 2002). In most cases, this is beyond the influence of individual landholders because it requires a revolution in water management, including restructure of water capture in the upper catchments, new infrastructure for water delivery, phasing out unsustainable agricultural industries, re-evaluation of water allocations and water trading, removal of flood mitigation structures. However, such changes will require support from individual landholders and adjustments to their land management.

Animal movements Movement through the landscape is necessary to sustain faunal populations through essential processes

such as dispersal, migration, foraging, access to refuge areas, escape from disturbances, colonisation and finding mates. Plant-animal interactions such as pollination and the dispersal of spores and seeds are also dependent on animal movement. Management actions at the farm to sub-catchment scale can enhance local movement pathways and contribute to increased permeability at larger scales. For example, restoring linkages in the landscape (Bennett 1999) and managing for increased structural heterogeneity in production areas and reducing contrast between adjacent land-uses (i.e., 'softening' the matrix, *sensu* Fischer et al. 2005) will facilitate animal movement. Restoring riparian vegetation may be a cost-effective priority for increasing landscape connectivity because riparian zones have disproportionately high species diversity, are natural linear elements in the landscape and dispersal pathways for a range of biota, and are often the foci for landholder-driven restoration activities (Tzaros 2001, Nilsson and Svedmark 2002, Wiens 2002).

5.6 Rules of thumb

In the absence of specific information, on-ground restoration has been driven by "rules of thumb" propagated through government funding programs, local experience and expert opinion. Table 3 lists some common guidelines or 'rules' for landscape design and restoration. Rules of thumb have intuitive appeal for land managers – they are conceptually simple, often provide quantitative guidelines and hold promise of measurable improvement. In most cases, they are sensible, best estimates based on some empirical evidence. Nearly all guidelines are couched in terms of "protect – improve – revegetate" and advocate the use of local native species, strategies to increase habitat complexity (e.g., patchiness of vegetation structure, logs and debris on the ground) and increase connectivity among patches. They aim to balance the capacity of landholders and communities to undertake restoration with the change necessary to improve biodiversity and ecological processes at a site and landscape level. Thus, they are useful for stimulating action because they provide a tangible framework around which restoration can be based. However, generalised rules also have several limitations, as the following list indicates.

- The applicability of such rules across different regions or in different restoration contexts has seldom been questioned, let alone tested. This may lead to inappropriate design or inefficient allocation of restoration resources if rules are taken out of the context in which they were generated.
- There is a tendency for each new funding program or research project to develop a new set of guidelines or 'rules'. This may confuse landholders faced with contrasting rule sets.
- The application of rules of thumb across a range of scales, or across taxonomic groups requires further investigation. For example, creating a patch of 0.5 ha may provide habitat for insects but not for mammals. Further, depending on the landscape context, that same 0.5 ha patch may increase landscape connectivity for mammals but not permanent habitat. Many guidelines are developed for birds and may not be applicable for other groups.
- There is significant variation in recommendations, which may confuse practitioners, stifle action or produce ineffective outcomes. For example, Paton et al. (2004) recommend restoring patches in the order of 20-100 ha to provide breeding habitat for woodland birds yet Fischer and Lindenmayer (2002a) highlight the conservation value of small patches (<1 ha) for birds in agricultural landscapes. The important role of individual scattered trees in the landscape for birds and mammals is also increasingly being identified (Law and Dickman 1998, Fischer and Lindenmayer 2002b). Kirkpatrick and Gilfedder (1995) also noted that small degraded patches of vegetation can provide habitat for threatened plant species. These examples demonstrate that the adage 'bigger is better' can be counter-intuitive to conservation outcomes if taken out of context.
- Most rules of thumb relate to landscape structure (e.g., patch size, corridor width etc.) with few specific recommendations for ecological processes.

6 Case Studies

Two case studies from north-central Victoria are presented to illustrate a range of restoration approaches and options. “Glendemar” and “Nil Desperandum” are both large sheep-growing properties located on the northern plains of the North Central CMA (Fig. 1). Although both properties are farmed with a strong emphasis on environmental sustainability, they differ in mode, history and philosophy of management. We visited both properties and conducted interviews with the principal land managers to gain insights into the operating principles and approach to landscape restoration on each property. We then examined spatial information on a GIS to devise a management plan for Glendemar and document the history of restoration actions on Nil Desperandum.

Glendemar has been home to the Duxson family for over 100 years. Two years ago, Dwain Duxson switched from a conventional mixed cropping and grazing enterprise to production based entirely on sheep farming using ‘holistic grazing management’ principles. The focus at Glendemar is ecosystem management: they radically altered their farming system from a high-input, high-cost business to a low-input, low-cost business in an effort to improve productivity and sustainability, and create a healthy environment for the next generation. Dwain and his family are just embarking on their journey.

Like Glendemar, Nil Desperandum has also been in one family, the Twigg family, for 100 years. For the last 40 years, Bill and Gwen Twigg have conceived and trialled many innovative ideas to improve the landscape and increase productivity. Some worked, some didn’t, some worked in unexpected ways, but Nil Desperandum is a living legacy to the evolution of ecological restoration for both biodiversity and production. With no sign of a farming heir, Bill concedes his focus is now continued environmental improvement with a commitment to biodiversity while maintaining the productivity and infrastructure of the farm.

6.1 *Glendemar: A case study in ecosystem management*

Glendemar is ~2400 ha in total, divided into three blocks on land that has all been cropped at some stage (Fig. 2). In 2004, Dwain Duxson turned away from high-input farming. Dwain’s motivation for change was his realisation that farming was not sustainable using conventional farming practices: to reverse land degradation and pass on a healthy landscape to his children, he needed to farm with the natural assets and capability of the land rather than work against it by continually adding chemicals and introducing exotic biota. He stopped cropping entirely, removing the need for expensive fertilisers, herbicides and pesticides, and adopted ‘holistic grazing management’ rather than a conventional set stocking grazing regime. He now runs sheep for meat and wool on ‘improved’ pasture of exotic annuals (annual rye, barley, lucerne; estimated at 70%) and native perennial grasses and forbs (~30%). Dwain’s goal is to continually increase the proportion of native perennials in his paddocks, with the ultimate goal of achieving 100% (mostly native) ground cover year-round. Dwain describes many advantages of a grazing system based on native perennials: increased ground cover; healthier, ‘softer’ soils due to increased water infiltration and organic matter; increased seed return to the soil; less soil loss through erosion and wind-drift; fewer weeds because the natives out-compete exotic species in the absence of fertilisers and pesticides; and fewer invertebrate pest problems (e.g., red-legged earthmite). After only two years, these ecosystem services have started to deliver commercial returns as well. There is less need for supplementary feed because the native perennials are a reliable and nutritious food source all year-round. Dwain’s sheep are healthier, with improved condition and no worms, making drenching unnecessary. Lambing survival is also up because the longer grass provides shelter and reduces losses from exposure. Stock management on Glendemar is guided by ‘holistic grazing management’ in which sheep are used as tools to manipulate the composition and density of ground cover (Savory and Butterfield 1999, LWWNTP 2006). Holistic grazing is based around grazing rotations: intensive grazing by large mobs of sheep for short durations in relatively small paddocks. The underlying premise is that most of the property is not grazed for most of the year, allowing for long periods of rest during which plant species best adapted to the natural conditions flourish. In the absence of fertilisers and other chemicals, this encourages the return of native perennials,

especially summer-growing grasses and palatable herbs and forbs, at the expense of exotic annuals. Stocking rates and duration of the rotations on Glendemar vary according to dry feed content in the paddocks, which is monitored closely, but are based around 30-day rotations in the plant growing season and 90-day rotations in the non-growing season. Dwain is confident he could accelerate the rate of pasture improvement if he could reduce his typical paddock size from 30-35 ha to 15 ha thereby enabling 'faster rotations' and more flexible grazing management. Although there are considerable establishment costs associated with fencing and watering, once in place, holistic grazing reduces inputs and costs facilitating the shift to a low-cost, profitable and ecologically sustainable business.

Dwain is primarily concerned with ecological function (e.g., ground cover, primary productivity, nutrient cycling, water filtration and retention, soil structure, water quality in the streams and springs). Increases in biodiversity are welcome but not the primary driver. Dwain recognises the benefits of a species rich grassland for production, that diversity provides insurance against environmental variability (e.g., fluctuation in climate, soil variability, insect outbreaks) and that different growing seasons ensure that his sheep always have something to eat. These concerns resonate with the principles of ecosystem management: manage processes to improve land condition and ultimately biodiversity will flourish. However, Glendemar was originally a mosaic of grassy woodland and riparian woodlands, with some box-ironbark forest on the upper slopes (Fig. 3), a situation vastly different to today. Glendemar currently has about 5% tree cover, mostly along creeks and on the foothills (Fig. 2) and this is unlikely to increase under the current management priorities. Several small remnants are fenced which are grazed intermittently. Mature paddock trees are valued for shade and shelter, and Dwain leaves fallen logs and branches on the ground to protect surface structure and recycle nutrients. As there are no plans for revegetation or restoration to increase the extent of woodland cover on the property, there is little prospect of recovery for biota reliant on woodland.

So, how could Dwain complement his ecosystem management approach with restoration specifically geared towards increasing biodiversity on Glendemar? Using the conceptual framework outlined in section 4 and the planning approaches in section 5, we have devised a biodiversity restoration plan for Glendemar that would increase the habitat potential of the property. The restoration plan relies heavily on natural regeneration of grassy woodlands and native pastures. Dwain has already ceased adding fertiliser and established a strategic grazing regime, thereby removing several of the major impediments to natural regeneration (Dorrrough and Moxham 2005). However, natural regeneration requires a seed source, which will depend on the viability of the soil seed-bank and proximity to other native vegetation, and favourable climatic conditions. The restoration plan (Box 1) and associated structural plan (Fig. 4) presented is not the only option – myriad possibilities exist depending on the objectives; nor is it a case of 'all or nothing' – each restored parcel will bring some biodiversity benefits.

The restoration plan increases the area of wooded vegetation from the current level of around 5% to 30% woodland cover. Two points of clarification are required here. (1) Tree cover estimates are derived from GIS layers based on 'solid' woodland patches: the gaps between trees are included in the area estimates so it is not an estimate of canopy cover per se. Actual canopy cover in grassy woodlands is no more than 30%, so in effect, the plan recommends establishing 30% canopy cover across 30% of the property. (2) Areas designated for restoration (enhancement, regeneration or revegetation) are not necessarily excluded from grazing – they are not to be 'locked up' and taken out of production. Once new growth has established (including regeneration of some eucalypts), woodland patches could be included in grazing rotations on Glendemar, particularly outside of the spring flowering season. Indeed, crash grazing in late summer or early autumn may enhance establishment of native species should appropriate rainfall occur, and grazing should continue to be used as a tool for controlling weeds (Dorrrough et al. 2005).

Restoration, even if the plan were to be implemented in its entirety, would be more effective if neighbouring properties and public land managers undertook similar actions (Fig. 5). In particular, restoration of creeks and drainage lines that extend through the landscape requires collaboration across several properties. Neighbourly coordination of revegetation or regeneration efforts can also

have synergistic effects: restoring adjacent areas on different properties creates a larger patch with greater biodiversity value than an equivalent area in two separate patches. Increases in connectivity, whether via habitat corridors, stepping-stones or mosaics of semi-natural vegetation must occur at the landscape scale. Again, strategic planning can increase effectiveness. For example, if neighbours each plant a 20 m buffer along a boundary fence, the resulting 40 m wide corridor will be of much greater value than two 20 m wide strips. Finally, it is unlikely that all landforms and vegetation types will be represented on a single property. Therefore, to increase representation, particularly of highly depleted vegetation types (often on the most productive land), restoration must occur at multiple sites across the landscape.

6.2 *Nil Desperandum: A case study in revegetation history*

Bill and Gwen Twigg have always farmed by a philosophy of custodianship: “we don’t own this land, we’re just looking after it”. Bill has noticed a steady improvement in land condition and biodiversity under his custodianship, and with it, farm productivity. When Bill and Gwen began managing Nil Desperandum (Fig. 6) 40 years ago, they possessed a “burning ambition to change the landscape”. Bill inherited a stressed and treeless farm with declining productivity, and wanted to improve land condition because he sensed productivity and profitability could be increased by farming in accordance with the land’s capability. Bill also has an innate love for trees and plants. Thus, although the terminology didn’t exist then, ‘intrinsic biodiversity’ (i.e., biodiversity for biodiversity’s sake), landscape aesthetics and spiritual well-being were central to Bill’s motivation to restore the land. However, he wasn’t sure *how* to improve landscape condition for there were few examples of alternative farming practices for guidance. So began 40 years of trial and error that Bill happily acknowledges continues to this day.

The Twigg’s first challenge was to return perenniality to the landscape. Bill was a pioneer of lucerne-based pastures; around 80% of the 1325 ha farm is now lucerne-based pasture (with approx. 20% cropping, mostly wheat), mixed with other exotic pastures such as rye and subclover, and a variety of other “palatable weeds”. Native species (mostly wallaby grass *Austrodanthonia* spp.) comprise about 5% of the pasture. Bill contends this system, combined with relatively light stocking rates and rotational grazing, has proved successful, with consistently high primary productivity, reliable fodder in dry times, lower water tables, and improved ground cover and soil structure. However, Bill concedes that lucerne does “rob” the soil of its nutrients and has noticed productivity decreases on a ~20-year cycle, after which some fertiliser or re-sowing is required. Bill is now interested in increasing the carrying capacity of his land by increasing the native component in his pastures, particularly Kangaroo Grass *Themeda triandra*, which is slowly returning, and saltbushes *Atriplex* spp., as lucerne production falls. He anticipates this will be achieved through a combination of strategic grazing and active re-establishment.

We estimated that the original vegetation on Nil Desperandum was predominantly Plains Grassy Woodland (~70%), with some Plains Grassland (~15%), dissected by several drainage lines (~15%) (Fig. 7). By the time Bill took over, remnant native vegetation cover, including scattered paddock trees, had declined to approximately 3%, all of which is now rated of ‘high’ conservation significance (Figs. 6 and 8). Paddock trees and the remaining patches of remnant vegetation were (and still are) valued for shade and shelter, and dead trees for their contribution to biodiversity, which in turn helps control invertebrate pests. Remnant patches have been fenced to protect them from grazing and in their place, Bill has established ‘forage’ plots of acacias and saltbushes, which provide biodiversity and land condition gains, as well as an alternative fodder source (Fig. 8). Bill began revegetation in 1958, when a copse of “bushy sugar gums” was planted. Nowadays, the use of non-indigenous species in a single-species planting in a small and isolated patch would not be recommended but in 1958 it was rare for anyone to be planting *native* vegetation at all.

Many early plantings on Nil Desperandum were long strips or shelterbelts, most only 2-3 rows of trees wide (Fig. 8). Strips and shelterbelts have localised benefits for decreasing wind shear, providing shade and shelter for stock and increasing litter fall and nutrient cycling. They provide habitat for some fauna, particularly invertebrates, but are not suitable for the majority of native animals.

Although such plantings were in vogue in an attempt to 're-connect' the landscape, the elongated shape of these patches means they consist entirely of 'edge' habitat, and consequently tend to be dominated by a handful of species that are able to persist in farmland (e.g., Noisy Miner *Manorina melanocephala*, Willie Wagtail *Rhipidura leucophrys*, Eastern Rosella *Platycercus eximius*, White-plumed Honeyeater *Lichenostomus penicillatus*). The influence of Noisy Miners is especially pervasive because they aggressively exclude many other bird species from small or narrow patches (Loyn 1987, Grey et al. 1998), which may reduce the natural control of defoliating insects and ultimately result in a decline in tree health (Reid and Landsberg 1999). Further, many strips were 'corridors to nowhere' (i.e., did not connect existing or new native vegetation) and followed fences or roads with little reference to natural contours and pathways of the land. Thus, many early strip plantings on Nil Desperandum and elsewhere have not produced the anticipated biodiversity benefits of increased habitat availability and connectivity. On a positive note, they did energise the community and introduce the concept of landscape-level revegetation, and re-vegetated strips may yet provide useful habitat as it matures or serve as the skeleton for an improved network of wider links.

Block plantings were the next revegetation strategy (Fig. 8). Blocks were recommended on the basis that less 'edge' would provide more 'interior' habitat, and therefore support more woodland species. Several of the revegetation blocks on the Twigg property incorporated remnant scattered trees, further increasing their habitat value. The revegetated blocks have probably played a key role in attracting and supporting several bird species (e.g., Superb Fairy-wren *Malurus cyaneus*, Grey Shrike-thrush *Colluricincla harmonica*, Common Bronzewing *Phaps chalcoptera*) that Bill has noted have returned to Nil Desperandum in recent decades. Yet, the relatively small size of the blocks (<3 ha) and their location away from other native vegetation may have limited the biodiversity benefits that potentially could accrue from revegetation. Care also needs to be taken to ensure revegetation matches the original vegetation type at the site: it appears that two sites on Nil Desperandum that were originally Plains Grassland have been re-planted with woodland species (i.e., trees and shrubs).

Five years ago, Bill embarked on creating an artificial wetland behind the house block (Fig. 8). Although there were no natural wetlands on the property, Bill has successfully created a wetland and fringing vegetation that provides a variety of habitats for fauna. Consolidating connections between Bill's wetland and Serpentine Creek, which is 5 km to the west will bring further biodiversity benefits. As Bill's focus shifts more towards landscape aesthetics and biodiversity, his aim is to re-create a "natural landscape" that is still productive. Recently Bill has fenced and revegetated (with direct seeding and tubestock) the natural depressions and drainage lines through Nil Desperandum in wide 'swathes' (Fig. 8). This is 'best management practice' in terms of landscape design: it is wide (~100m), follows the natural flow of the landscape, is faithful to the original vegetation type, and connects existing areas of native vegetation. Complementary action by neighbouring landholders is now required to establish landscape level connectivity. Similarly, on "Elmswood", a smaller property (668 ha) on the Loddon River (Fig. 6) purchased recently by the Twiggs, restoration will be more compatible with features of the landscape. Initially, a wide buffer (50-150 m) adjacent to the riparian zone has been fenced allowing natural regeneration to occur. As Bill continues to experiment, watch and learn, it is inevitable he will modify his methods and approach to improving the landscape – adaptive management in action.

7 Guiding Principles for Restoration

This section draws on the preceding discussion to formulate a set of guiding principles for restoration at the property level. It is not possible to be specific because site attributes (values, risks and threats) and personal aspirations will vary from property to property. Here, we try to reconcile restoration science with the "bottom-up" approach of landholder driven action, which tends to be opportunistic and ad-hoc, to develop principles that are relevant to individual landholders.

Principles for Restoring Landscape Resilience

1. Increase the areal extent of native vegetation with an appropriate species mix and sufficient structural complexity that it provides habitat for a range of flora and fauna. There is unlikely to be a quantitative target for revegetation that will ensure species diversity or landscape resilience is restored to former levels – this will depend on the species and communities of interest, quality of existing and new habitat, landscape context and extent of disturbance to ecosystem processes.
2. Emphasise repair of ecosystem processes: e.g., nutrient cycling, retention of water and limiting resources, soil stability, pollination, gene flow, animal movement. This may involve revegetation (natural regeneration, direct seeding, tubestock) as well as other management actions such as strategic grazing, fire management, soil manipulation, landscape engineering, control of aggressive or invasive species, and maintaining keystone habitat features (e.g., paddock trees, fallen logs, deep pools in streams).
3. Within the context of the restoration plan follow the edict of:

Protect – Improve – Enhance – Reconstruct

Protect maintain existing native vegetation through fencing or sympathetic management. There are several financial incentive schemes available to landholders interested in remnant protection. In some cases, temporary protection will be adequate to protect habitat elements at critical times.

Improve improve quality of existing native vegetation by removing or controlling threatening processes (e.g., weeds, feral animals, firewood collection, inappropriate fire regimes).

Enhance supplement and enlarge existing patches of native vegetation through revegetation of habitat gaps or buffers (particularly around sensitive areas such as riparian zones).

Reconstruct create new patches of native vegetation through replanting or manipulation of physical processes to promote natural regeneration. Priority should be given to reconstructing large patches of under-represented vegetation classes.

Although the logical progression of actions *at a given site* is from protection through to reconstruction, there may be circumstances when different parts of a property are receptive to different restoration actions at the same time. Thus, resources permitting, remnant protection may occur concurrently with reconstruction in other parts of the property.

4. Build spatial variation (i.e. patchiness) and landscape heterogeneity into landscape design - seek variety in patch types, patch shape and size (though larger patches are preferred), patch boundaries and landscape position. For example, increased bird species richness may be achieved by establishing new habitat (i.e. revegetation for biodiversity) in landscapes with existing remnant vegetation compared with landscapes without existing remnant vegetation.
5. Revegetation should seek to simulate natural processes by representing original ecological vegetation classes and functional vegetation types (e.g. nectar, seed and fruit producing plants).
6. Promote continuity of vegetation along environmental gradients (e.g. rainfall, geographic, altitudinal, topographic). Connectivity at this scale is important to allow movement in response to changes in resource availability over time, natural catastrophes and climate change.
7. Employ strategies to counter habitat fragmentation (i.e., restore landscape connectivity):
 - i. Expand area of existing remnants.
 - ii. Increase number of patches through reconstruction, particularly between existing patches of native vegetation.
 - iii. Create landscape linkages, including corridors (linear strips that link patches of native vegetation) and stepping stones (small patches located between existing native vegetation) (see below).
 - iv. Amalgamate nearby patches to form a single larger patch.

- v. Reduce the hostility of the matrix to native fauna by 'softening' boundaries between landscape elements, maintaining habitat elements in the matrix (e.g., paddock trees, fallen logs, rocks), strategic arrangement of different land-use types, reducing intensity of land-use across the landscape (for example, increasing the area of native pastures) and incorporating refuge areas in high land-use intensity zones.

Guidelines for Landscape Linkages

1. Clearly define biological purpose of linkage in terms of target species or faunal groups, spatial (i.e., extend over what distance) and temporal (i.e., used over what timeframe) scale and ecological function (e.g., seasonal migration, access to irregular resources, natal dispersal).
2. Consider design, dimensions, vegetation type and management required to meet purpose. Knowledge of ecology and behaviour of target species is required here.
3. Retain existing natural links where possible (especially at the planning stage if habitat is to be lost) rather than create new habitat.
4. Connectivity is more than 'wildlife corridors'. Stepping stones, increased permeability of non-habitat (see 7 v above), alternative land-uses and ephemeral links may also achieve desired outcomes.
5. Ensure habitat quality and diversity in linkages is suitable for target species. Wildlife will not enter linkages if quality is poor, even if destination is pristine.
6. Structural priorities for landscape linkages:
 - i. The wider the better – ultimate test is the maintenance of connectivity. Aim for twice the width of edge effects (e.g. light penetration, habitat structure differences, floristic composition, weed invasion) to ensure there is some 'interior' habitat (but see Table 3).
 - ii. Longer linkages must be wider to provide 'habitat for the journey' (i.e. increased resources).
 - iii. Including nodes (small patches built into the linkage) increase use by wildlife but do not negate the need to maximize the width to length ratio.
 - iv. Where appropriate, fill in gaps in existing linkages.
7. Location priorities for landscape linkages:
 - i. Follow natural movement pathways if known – e.g. migratory routes, daily foraging patterns.
 - ii. Follow natural environmental features – rivers, creeklines, drainage lines, ridges, and gullies but attempt to incorporate all habitat types (multiple paths) in one or several links. These are often irregular rather than straight lines between two patches.
 - iii. Include existing natural vegetation, where possible.
 - iv. Unique or irreplaceable linkages should be afforded highest priority (but network of multiple connections usually functions more effectively).
 - v. Locate away from sources of human disturbance, including freeways / roads.
8. Design linkages that enable passive wildlife recolonisation. That is, recognize that restored sites must be colonised from existing source habitats. Thus, providing links to known or potential source populations is critical to success. Habitat quality in recipient patch must also be adequate to support populations of target species.
9. Monitor success of linkage against original objectives. Can effectiveness be increased through adaptive management (e.g. provision of nest boxes, habitat manipulation, or increased width)?

8 Future Research Directions

There are several priorities for future research, presented here as key questions, to improve methods and approaches for landscape restoration.

1. Disturbance regimes – what is the scale, frequency, severity, intensity and seasonality of natural disturbance regimes (e.g., fire, flood, herbivory, drought, storms)? How do organisms and ecosystems cope with natural and modified disturbance regimes? Based on our current knowledge, we are poorly placed to implement or recreate disturbances that will benefit the system under repair.
2. Climate change – putting aside uncertainty about the predictions of climate change itself, estimates of changes in species' distributions under different climate scenarios are inexact, especially at relatively small spatial scales. There are several questions about climate changes that can guide research. What will be the impact of climate change on critical population processes, such as breeding, migration, seed set, flowering and fruiting? It is probable these processes will occur earlier than at present, but different species will respond to different degrees. What effect will this have on inter-specific interactions (e.g., pollination, herbivory, predator-prey dynamics, competitive interactions)? What are the implications of changes in rainfall – less in winter/spring, more in summer – for revegetation, in terms of species selection, ground preparation and timing of activities? What are the implications of the interaction between climate change and disturbance regimes (fire frequency, drought, floods and storms)?
3. Time lags – we know there are lags in population responses following habitat loss (extinction debt), but lags in recovery following restoration are poorly understood. Key research questions concern the delays in availability of critical resources (Vesk and Mac Nally 2006) and how this affects trajectories of recovery? A thorough understanding of time lags will greatly benefit restoration planning.
4. Hysteresis – do communities re-assemble following restoration in reverse order to which they disassembled during habitat loss (i.e., is the last species to disappear the first to recover)? Do the same environmental variables and landscape factors that contribute to the loss of species (e.g., extent of habitat, patch size, isolation) trigger the return of species (i.e., if a species disappears as patches are cleared below 20 ha, will it return once it is restored to 20 ha)?
5. Keystone species – what are the ecological processes that are influenced by which keystone species? What is the precise role of keystone species and thus, what are the implications of removing them from the system? How can their recovery be accelerated? What is the extent of functional redundancy among species, and therefore what are the opportunities for functional surrogacy during restoration? Could this increase the effectiveness and speed of restoration?
6. Soil biota – what role do soil biota (bacteria, fungi, invertebrates) play in natural regeneration (e.g., through mutualism, parasitism, symbiotic relationships, nutrient cycling) and replanting? Can we manipulate soil conditions to expedite such processes? What influence does fire have on soil biota (e.g., McCarthy and Brown 2006)? What is the relationship between soil biota and the ground layer (soil crusts, lichens and cryptogams, litter, logs etc.)?
7. Functional responses – we still have only a primitive understanding of the relationship between landscape structure and functional responses of species. For example, how does the arrangement of native vegetation in the landscape (e.g., patch sizes, connectivity, juxtaposition of land uses) constrain or facilitate the movement of animals, and hence impact on processes such as dispersal, migration, pollination, seed dispersal, trophic relations? What are the genetic consequences of disrupted movement for population persistence? Why do different species respond differently to landscape change?

8. Scale issues – what is the operational scale of ecological processes, functional responses of fauna to landscape change, or community assemblage patterns? How do relationships between environmental factors and ecological responses hold across multiple spatial and temporal scales? What is the interaction between scale and disturbance processes? How can scale issues be incorporated into restoration planning?
9. Landscape design for native vegetation – much of the literature and guidelines on landscape design and restoration focus on native fauna and its habitat. Similar research, modelling and guidelines are needed for native plant species and communities that addresses their long-term viability in the landscape as entities in their own right. What is the coherence between landscape design for native plants, and that proposed for groups of native fauna (e.g., birds, mammals, reptiles)?
10. Synergies for conservation and production – which restoration actions or particular aspects of landscape design enhance both conservation and production? Are there particular synergies that will increase cost-efficiency of restoration?

Acknowledgements

Funding for this project was provided by the National Action Plan for Salinity and Water Quality through the Multiregional Project Laying the Foundation. Jim Radford is also supported by the Australian Research Council (LP0560309). Jim Radford would like to thank Richard MacEwan (Dept. of Primary Industries) for infrastructure support. In particular, we appreciate the support and involvement that Dwain Duxson and Bill Twigg have given this project.

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Table 1. Examples of aspirational goals, resource condition targets and management actions for native vegetation and biodiversity conservation, as set out in Regional Catchment Strategies. Resource condition targets and management actions in coloured text relate to aspirational goals in same colour, where specific linkages apply.

Region	Vision	Aspirational Goals (>50 years)	Resource Condition Targets (10-30 years)	Management Actions (1-10 years)
Mallee CMA ¹	To maintain ecological processes and to protect and improve the extent and quality of biodiversity in the Mallee.	<p>The extent, diversity and condition of all EVCs maintained above self-sustaining thresholds.</p> <p>Threatened ecological communities and threatened species populations recovered to self-sustaining levels and secured against further decline.</p>	<p>30% native vegetation cover across each bioregion.</p> <p>Increased extent of each EVC to at least 15% of pre-1750 extent</p> <p>20% improvement in condition across all conservation significance levels using the 'Habitat Quality Assessment' method.</p> <p>No decline in populations for a yet to be determined number of rare or threatened species.</p> <p>Increase in size, range and number of populations for a yet to be determined number of rare or threatened species to (yet to be determined) stable levels.</p>	<p>Baseline conditions determined and target levels set.</p> <p>Vegetation extent and quality assessments completed at priority sites using 'Habitat Quality Assessment' method.</p> <p>200 priority reserved covered by management documents.</p> <p>A yet to be determined no. of hectares restored to native vegetation as habitat for native species in priority areas.</p> <p>Remnants of EVCs <15% of pre-1750 coverage subject to management agreements with land managers.</p> <p>10 recovery plans established for nationally endangered species in 5 years.</p> <p>Priority remnants linked with corridors, with 10% completed in 5 years.</p> <p>Population monitoring of priority populations.</p> <p>Reintroduction of regionally extinct species.</p>
North Central CMA ²	<p>The ecological function of indigenous vegetation communities will be maintained and, where possible, improved.</p> <p>Populations of threatened native plant and animal species will be restored to viable levels.</p> <p>Threatened vegetation communities will increase in extent and improve in quality to achieve a net gain.</p>	<p>Increase native vegetation cover to 30% of the region.</p> <p>Increase the coverage of all EVCs to at least 15% of their pre-1750 distribution.</p>	<p>Increase native vegetation coverage to 20% of the region.</p> <p>Improve the quality and coverage of all vulnerable or endangered EVCs and any others with <15% of pre-1750 distribution by 10% (as measured by habitat hectares).</p> <p>Maintain or improve existing viable populations of significant threatened species.</p> <p>No further bioregional extinctions.</p>	<p>Identify areas supporting high conservation significance native vegetation that is threatened by sub-division, fragmentation and tree decline, salinity etc. Develop and implement Biodiversity Action Plans for these areas.</p> <p>Develop vegetation and environmental protection overlays for priority threatened species and communities.</p> <p>Provide financial support to landholders wishing to protect and enhance remnant vegetation. Run annual Bush Tender auction in biodiversity priority areas.</p> <p>Develop and implement roadside management plans.</p> <p>Support landholders and public land managers in eradicating or implementing long-term control over new and emerging weeds, priority weeds and controlled weeds.</p> <p>Develop and implement appropriate fire management regimes to sustain ecological processes in key remnants.</p>

Region	Vision	Aspirational Goals	Resource Condition Targets	Management Actions
Goulburn-Broken CMA ³	To protect and enhance natural assets and their ecosystem processes and functions in a way that provides benefits for native biodiversity, social and economic aspects.	The community will work in partnership with Federal and State Governments and other agencies to protect and enhance ecological processes and genetic diversity to secure the future of native species of plants, animals and other organisms in the Catchment.	Maintain extent of all native vegetation types at 1999 levels. Improve the quality of 90% of existing (2003) native vegetation by 10%. Increase the cover of all endangered and applicable vulnerable EVCs to at least 15% if their pre-European cover. Increase 2002 conservation status of 80% of threatened flora and 60% of threatened fauna.	By 2007 maintain all of the 715,000 ha of 1999 native vegetation and 8,000 of 'new' native vegetation. By 2007 protect 6,000 ha of remnant vegetation on private land. By 2007 plant, direct seed or naturally regenerate 8,000 ha of native vegetation. Implement relevant Action Statements and Recovery Plans. Capture opportunities for protecting and enhancing native biodiversity as land use changes from agricultural to less intensive uses over large areas.
North East CMA ⁴	Diverse, healthy landscapes; vibrant communities.	Maintain the quality of all EVCs. Achieve net gain in biodiversity across the region. Decrease in number the most highly threatened flora and fauna species and communities to levels that support self-sustaining ecosystems.	Improve the quality of priority EVCs by 10% of 2005 levels measured by habitat hectares. Achieve on-going 'net gain' for all EVCs ensuring a positive gain in extent, distribution and quality as measured against the previous year. Increase where possible the extent of native vegetation for endangered EVCs to 15% and for vulnerable EVCs to 30% relative to 1750 extent. Maintain or improve the 2003 conservation status of 80% of threatened flora and 60% of threatened fauna species by 2023.	Benchmark native vegetation condition in priority EVCs. Protect, enhance and restore 10,000 ha of priority EVCs through management agreements, fencing, pest plant and animal management and revegetation by 2009. Benchmark native vegetation extent. Baseline data obtained on native vegetation removal on all land tenures. Benchmark resource condition of significant native species and ecological communities using NRM Matters for Targets indicators. Implementation of Recovery Plans and Action Plans by 2009 for threatened species. On-going development of more effective management tools to determine number of threatened species. Biodiversity Actions Plans completed and implemented by 2005.

¹ Mallee Catchment Management Authority (2003); ² North Central Catchment Management Authority (2003); ³ McLennan et al. (2004); ⁴ North East Catchment Management Authority (2003)

Table 2. Principles for enhancing the value of revegetation for wildlife (adapted from Bennett et al. 2000).

Site level
Use locally indigenous plant species where viable
Match plant species to the landform
Establish natural layers in the vegetation
Promote fine-scale patchiness of vegetation
Provide ground-layer components as resources for wildlife and to assist restoration of ecosystem processes (<i>the 'messier the better'</i>)
Manage the composition and structure of restored habitats
Control disturbance and degradation

Patch level
Establish larger patches for large populations (<i>the larger the better</i>)
Ensure patches meet area requirements of particular species of concern
Create large patches for diverse animal communities
Increase width to reduce edge effects (<i>the 'rounder' the better</i>)
Position revegetation to increase opportunities for recolonisation (close to existing populations relative to the mobility of species and 'resistance' to movement of the landscape)
Build on to existing remnant vegetation (including 'gaps' in linear remnant vegetation)
Locate new vegetation away from known sources of disturbance

Landscape level
Increase the total area (extent) of habitat in the landscape (<i>the more the better</i>)
Establish multiple populations (<i>the more the better</i>)
Provide for wide-ranging species that use different types of habitat (all required habitats must be present)
Enhance connectivity by establishing different types of habitat linkages (corridors, stepping-stones, mosaics of semi-natural vegetation) (<i>connected is better than isolated</i>)
Promote corridors wide enough to provide for habitat specialists (<i>the wider the better</i>)
Give priority to streams and watercourses as 'natural' corridors
Establish links along topographic features (e.g., ridges, gullies)
Promote habitat contiguity from 'crest to creekbend' (i.e., along an altitudinal gradient)
Re-establish poorly represented vegetation types and restore remnants of degraded and depleted vegetation types (especially those of fertile soils)

Regional level
Develop a long-term vision for the region
Identify regional ecological priorities for restoration
Implement effective monitoring programs (<i>quantitative</i> monitoring of outcomes of revegetation actions, effectiveness of techniques and change in land use)

Table 3. Examples of restoration rules of thumb.

Source	Approach	Extent of habitat	Patch size	Connectivity / Isolation	Other notable guidelines
Wallatin catchment, WA (Lambeck 1999)	Focal species (birds)		>25 ha	Distance between patches <2 km Linear strips > 30 m wide for heath and >60 m for woodland	Inter-fire periods of >50 years
Birds on Farms (Barrett 2000)	Surveys	30% of total farm area	>10 ha	Linear strips > 50 m wide	Maintain shrub cover in >33% of area of patches
Revegetation and Wildlife (Bennett et al. 2000)	General principles		>10-20 ha	Linear strips 20-50 m wide	Give priority to riparian areas to enhance connectivity
Sustainable grazing thresholds, Qld (McIntyre et al. 2000)	Expert opinion	30% woodland cover	>5-10 ha		Maintain 60-70% ground cover with tussock grasses Limit intensive land uses to 30% of area
Saltshaker Project, NSW (Freudenberger 2001)	Focal species (birds)		>10 ha	Create linkages between patches >1.5 km apart, using either linear strips > 25 m wide or stepping stone patches >10 ha	
Bringing Birds Back, ACT (Taws 2001)	Surveys		>2 ha	Connect sites with windbreaks (as wide as possible) or stepping stone patches >1 ha	
Gabbi Quoi Quoi catchment, WA (Brooker 2002)	Focal species (birds)		>40 ha for heath >30 ha for woodland	Distance between patches <2 km for heath and <4 km for woodland	Habitat condition important
Landscape GIS tool, Vic (Wilson and Lowe 2003)	Literature review	15% cover	>40 ha for fauna >10 ha for flora	Linear strips >40 m; riparian strips >100 m	Include all vegetation types
Managing riparian widths (Price et al. 2004)	Various				Recommends minimum riparian vegetation widths for a range of objectives: 5-10 m (either side) to improve water quality, reduce streambank erosion, maintain natural light and temperature, and provide food inputs and aquatic habitat; 5-30 m to provide habitat for fish; 10-30 m to provide habitat for terrestrial fauna.

Box 1. Biodiversity Restoration Plan for Glendemar

Define the ecosystem / landscape to be restored

Property boundaries of Glendemar located in the Avon – Richardson sub-catchment of the North Central CMA region. Ideally, neighbouring properties would be included in a single, integrated plan but this plan is confined to Glendemar. Glendemar is comprised of three discrete parcels: a northern block adjacent to the Avon River (343 ha), a main central block in the Anderson / Wallaloo Creek catchment (1713 ha), and a south-western block adjacent to the Richardson River (357 ha) (Figs. 1 and 2).

Includes slopes, riparian zones, drainage lines and plains. Original vegetation (Fig. 3) was predominately Box-Ironbark Forest and Low Rises Woodland on the slopes, Floodplain Riparian, Creekline Grassy Woodland and Drainage Line Woodland along watercourses, and Grassy Woodland and Plains Woodland on the plains.

Current land tenure is privately owned and managed by the Duxson family. Glendemar falls outside zones for incentives schemes relating to salinity, biolinks, Bush Tender and other programs. Have received some CMA assistance for fencing along waterways.

Assess current condition of the ecosystem

Landscape context

Glendemar is close to two large blocks of native vegetation; it is 2.5 km west of Mt. Bolangum Flora and Fauna Reserve (2765 ha) and < 1 km north-west of Morri Morri Nature Conservation Reserve (2050 ha). There is almost contiguous native vegetation between Morri Morri NCR and Glendemar (Fig. 2). Several small patches of disturbed woodland occur on nearby properties; most of these are surrounded entirely by farmland. Some roadside vegetation is present in the landscape but it is not an extensive network. To the west and north extend the cleared (or naturally treeless) plains of the Victorian Riverina (correct bioregion?). Several significant (although intermittent) watercourses run through Glendemar, including the Avon and Richardson Rivers, and Anderson and Wallaloo Creeks. These streams retain a narrow band of fringing riparian woodland. In addition, many drainage lines (most entirely cleared) traverse the property. Land-use surrounding Glendemar is grazing (predominantly sheep on exotic pastures) and cropping.

Native vegetation

Current extent of wooded vegetation is approximately 5%. Fully 75% of the wooded vegetation is rated 'high' conservation significance, with the remainder rated 'medium' conservation significance. Pasture is comprised of exotic annuals (estimated at 70%) and native perennial species (~30%).

Action: Establish sites for habitat condition assessments (e.g., Habitat Hectares, Vegetation Quality Assessments) in each of the existing EVC remnants and selected paddocks. Conduct habitat condition assessments, including detailed plant species inventory using quantitative estimates of cover.

Native fauna

No surveys of native fauna have been undertaken on the property. The main faunal groups likely to be present on Glendemar and surrounds include bats, birds, reptiles, frogs, and arboreal, terrestrial and subterranean invertebrates.

Action: Conduct baseline fauna surveys in woodland remnants (plains, slopes and riparian) and paddocks on Glendemar, in 'control' paddocks and creeklines that will not be restored, as well as in nearby 'reference' sites (e.g., Mt. Bolangum FFR; Morri Morri NCR; selected riparian sites). Priority groups for survey include bats, birds, reptiles and soil invertebrates.

Processes and disturbance regimes

Current management objectives are closely tied with ecological processes, such as primary productivity, succession of pasture species, nutrient cycling, organic decomposition, soil moisture and surface water flows. The managers therefore undertake some regular monitoring and have a good 'feel' for these processes. However, quantitative benchmarks of current condition are needed to monitor the effects of restoration over time.

Action: Establish monitoring sites for assessment of selected ecosystem processes and commence monitoring. Priorities for monitoring include soil structure, soil moisture and nutrient analysis, bare ground (or vegetative ground cover), plant biomass and sediment loads in creeks. The 'Landscape Function Analysis' tool could be used to track fine-scale patchiness and soil stability, infiltration and nutrient cycling.

Fire can help maintain the health and diversity of native pastures, as well as control woody plants or promote their regeneration. Native pastures should only be burnt when pasture condition indicates that it would be useful. Experience shows that burning small patches in the morning or evening creates a patchwork of areas at different stages of growth, providing both shelter and green feed. The best time to burn is when plants have finished flowering and setting seed and when wildlife have finished their breeding cycle. In South-eastern Australia this is usually in autumn, which fortuitously is also the time when seasonal conditions are good for controlling fires.

Action: Establish some control and trial paddocks to assess the value of controlled burning for pasture recruitment and growth.

Construct a landscape vision (for biodiversity)

To restore Glendemar such that it provides habitat for indigenous flora and fauna, offers pathways for the movement of populations, individuals and genes through the landscape, and increases landscape heterogeneity.

Specific restoration goals (for a 50 year time horizon) may include:

- (1) increase woodland cover to 30% of the property.
 - (2) increase landscape connectivity by buffering all riparian zones, revegetating drainage lines, linking patches with habitat corridors and establishing new patches as 'stepping-stone' habitat.
 - (3) increase representation of native species in ground cover to 80%.
 - (4) increase richness and diversity of native plant species by 200%.
 - (4) increase ground cover to no lower than 80% year-round.
 - (5) increase landscape heterogeneity by restoring a range of vegetation types in different parts of the landscape.
 - (6) reduce rate of surface water flows in creeks and drainages by 100% (double the time water remains on property).
 - (7) reduce soil loss and gully erosion to nil.
-

Articulate restoration actions

Pasture management

Construction of fences to reduce paddock size; installation of mobile watering troughs and a water delivery system; burning trials to examine role of fire in pasture management and natural regeneration.

Native vegetation

Figure 4 presents a structural plan for where restoration may occur on Glendemar. Actions include replanting vegetation (i.e., tubestock / direct seeding) in areas without viable soil seedbanks or distant from seed sources, encouraging natural regeneration (may require soil disturbance, fire and temporary removal of grazing), and improving existing remnant vegetation through adding missing vegetation components (e.g., tussock grasses, shrubs) or other habitat features (e.g., fallen timber, rocks) as required. The plan includes 200 ha of replanting (plains – 116 ha; riparian – 44 ha; drainage – 40 ha), 393 ha of natural regeneration of grassy woodland (plains – 280 ha; riparian – 88 ha; slopes – 25 ha) and 78 ha of habitat improvement (slopes – 71 ha; drainage – 7 ha).

This plan attempts to: increase the extent of habitat in the landscape; increase the size of existing patches and establish new patches of 'usable' size; increase connectivity in the landscape; maximize width riparian vegetation and other linear vegetation elements; establish 'nodes' along longer linear elements; prioritise riparian zones and drainage lines; restore a range of vegetation types; connect the highest and lowest points in the landscape with continuous vegetation; provide pathways for passive wildlife colonisation; increase the permeability of grazing land; buffer water courses; reduce edge effects; and 'funnel' species into preferred movement pathways.

Economic considerations

Farmers should not carry the costs of landscape restoration alone. Cost-sharing, incentives, creation of new markets (e.g., carbon credits, biodiversity credits, ecosystem service payments, farm forestry), custodianship payments, Bush Tender, rate rebates, tax rebates and premiums for 'clean, green' products are areas that could contribute to easing the financial costs of restoration. Some of these options exist but many require policy development and/or political and social reforms before they become available.

Establish success indicators

Indicators would depend on current conditions but may include population increases for particular species or faunal groups (e.g., 50% increase in abundance of skinks), increases in species richness (e.g., bird richness = 50 resident or regular species; plant species richness increases by 200%), increases in soil moisture and organic content to specified levels, improvements in land condition (e.g., gully erosion, surface scouring), increase representation of native species in ground cover to 80% with maximum of 20% bare ground. It would be useful to set interim success indicators for 10, 20, 30 and 40 years as well.

Implement restoration actions

The order in which restoration actions are implemented is important. The aim should be to facilitate passive wildlife recolonisation. This involves drawing colonists (plants and animals) from existing remnant vegetation and source blocks (e.g., Mt Bolangum FFR and Morri Morri NCR) along habitat linkages such that they can 'flow' down onto restored habitat. Priorities include:

1. Restoring patches contiguous with source patches.
2. Restoring sites relatively close to source patches and linking these to existing remnants, preferably via riparian corridors or natural topographical features (gullies, ridges).
3. Filling gaps in existing linear networks along creeks, roadsides, drainage lines etc.
4. Establishing more remote (i.e., further from existing remnants) restoration sites.

5. Providing links to remote sites, again preferably by following natural features, but inevitably some will be along fences and roadsides.

Concurrently, steps should be taken to decrease resistance to movement through the production areas. This is achieved by maintaining scattered paddock trees and ensuring enough regeneration for replacement of mature trees as they senesce, increasing native pasture, and maintaining or providing sheltering habitat (e.g., logs, rocks, natural springs) in paddocks.

Opportunities for learning need to be incorporated in the implementation stage. Thus, particular restoration actions (e.g., replanting, regeneration) should be repeated at several sites across the property. Ideally, restoration actions would be replicated in several properties across the sub-catchment. Implementation and replication of strategic grazing occurs at the property scale.

Monitoring

In order to evaluate the success of restoration actions it is necessary to monitor restoration actions and ecological indicators, not only at restored sites, but nearby 'reference' (not degraded, not restored) and 'control' (degraded, not restored) sites as well. Monitoring must occur at appropriate spatial and temporal scales, and be replicated at the paddock (where relevant) and property scale if possible. For example, monitoring may include (but is not limited to):

- Extent of woodland cover – property scale, every 5 years.
 - Pasture composition (species composition, ground cover) – plots within paddocks within properties, twice yearly.
 - Habitat condition (Habitat Hectares, Vegetation Quality Assessment) – plots within remnants within properties (land-use parcels), twice yearly.
 - Bird community (species composition, richness) – remnants within properties, spring and autumn surveys every three years.
 - Soil properties (pH, moisture, organic content, nutrient analysis) – plots within paddocks within properties, twice yearly.
 - Erosion (end of property sediment loads in creeks, soil movement profiles) – plots within paddocks within properties, annually.
 - Co-variables (stocking rates and duration of rotations, climate variables, fires, water table depth, etc.) – as appropriate.
-

Evaluation

Evaluation is then required in terms of progress towards goals and success indicators and costs of restoration efforts. Adjust management based on trade-off of restoration inputs (costs, resources) and ecological and/or biophysical responses (benefits).

Figure 1 Location of case study properties in Victoria and North Central CMA

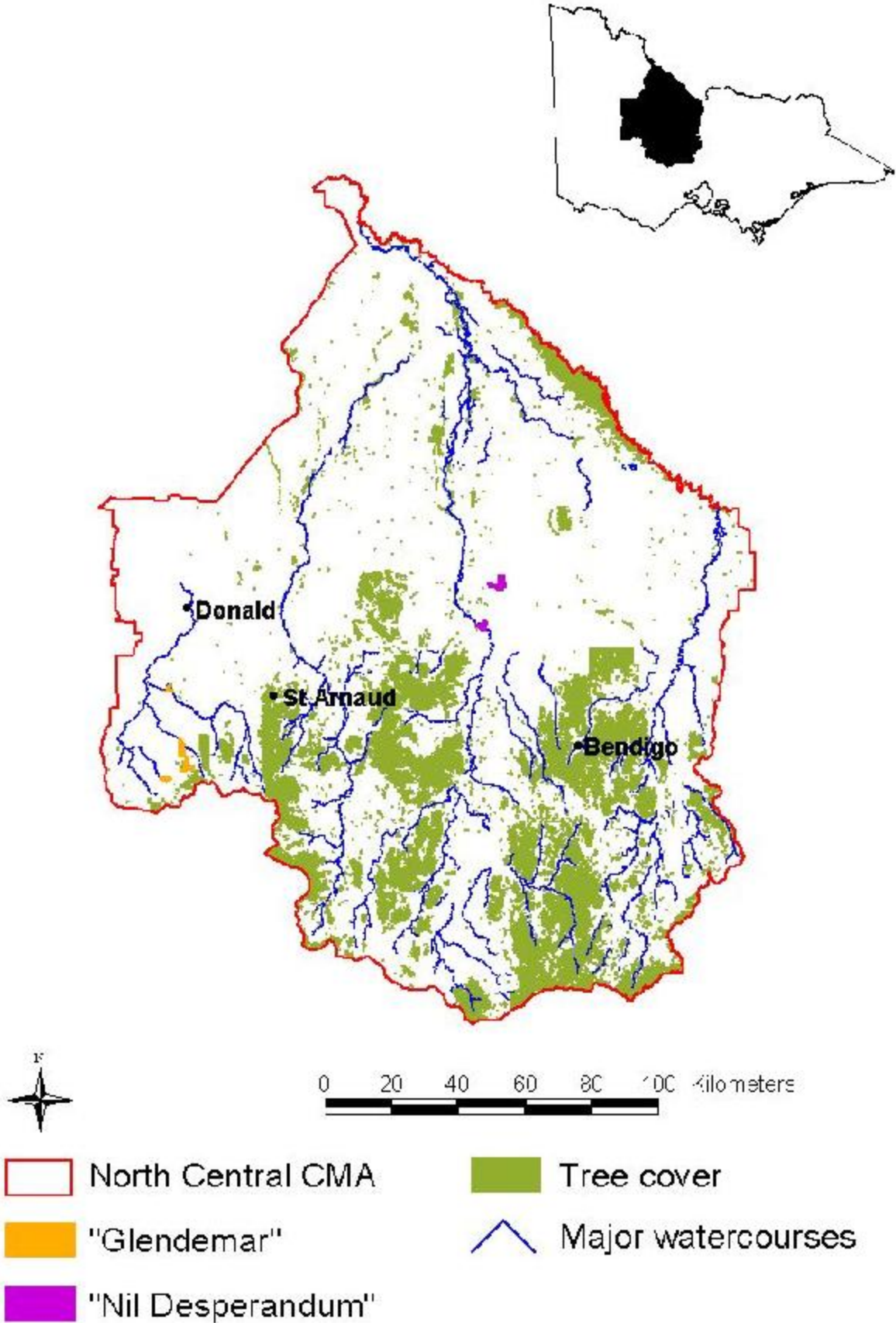
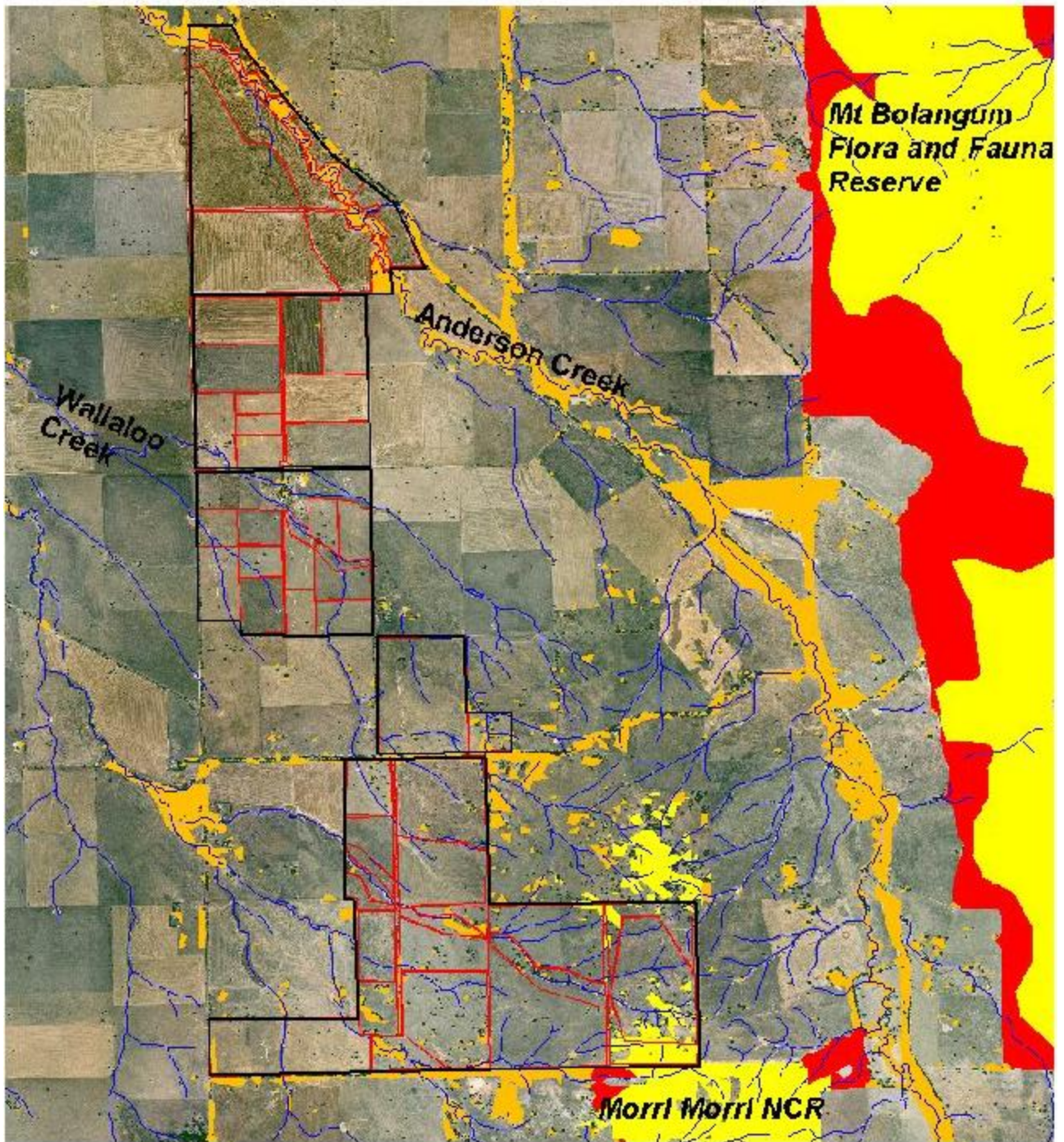


Figure 2a. Current layout and conservation status of extant native vegetation on Glendemar – central block.

Central paddocks



- ▭ Glendemar
- ▭ Paddock layout
- Surface hydrology
 - ▬ Stream
 - ▬ Gully - perennial watercourse
- Conservation significance
 - Very High
 - High
 - Medium
 - Low

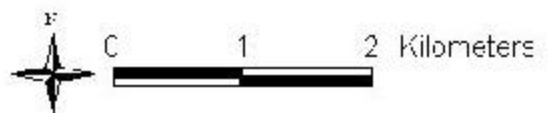
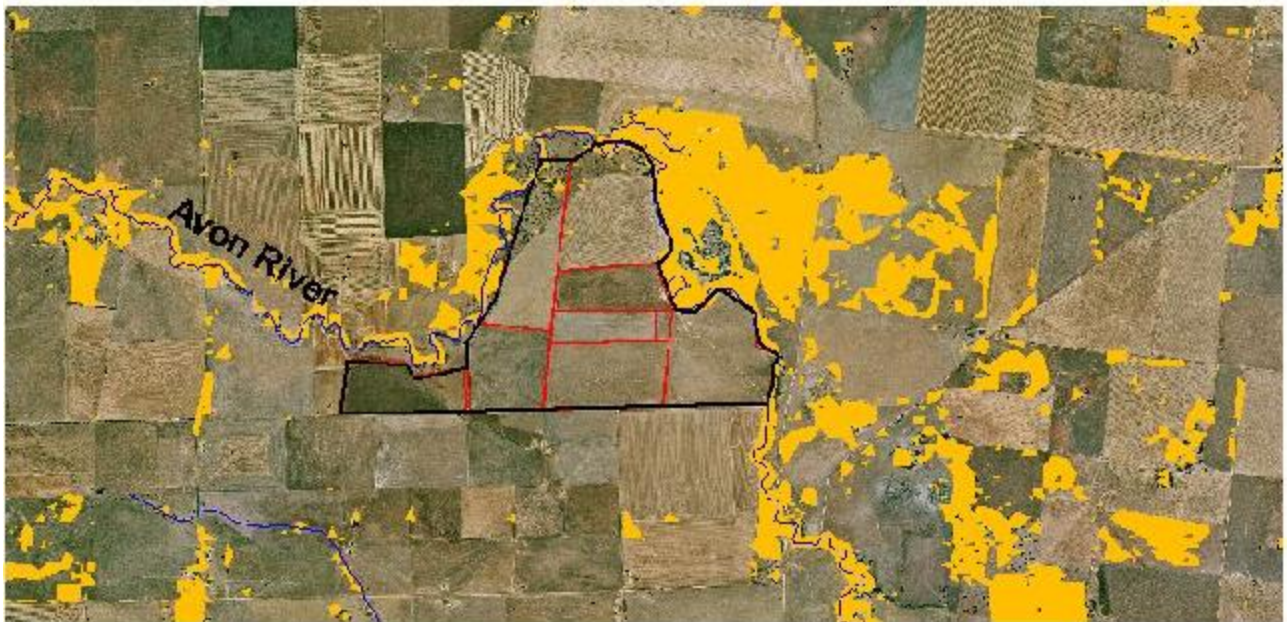
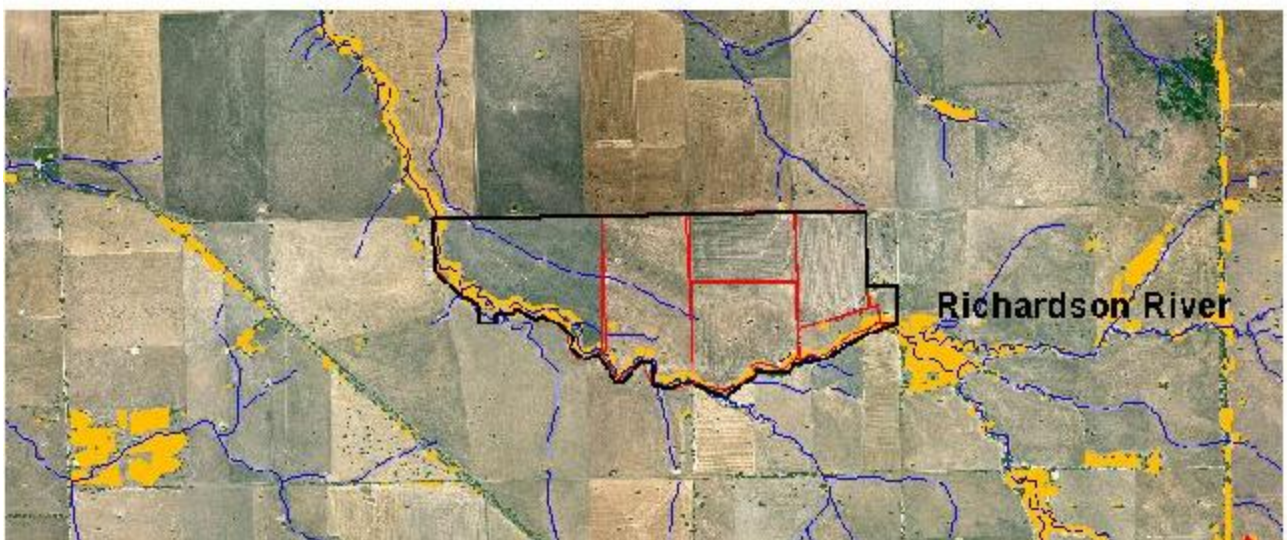


Figure 2b. Current layout and conservation status of extant native vegetation on Glendemar – northern and south-western blocks

Northern paddock



South western paddock



- ▭ Glendemar
- ▭ Paddock layout
- Surface hydrology
- ~ Stream
- ~ Dully-perennia watercourse
- Conservation significance
- Very High
- High
- Medium
- Low

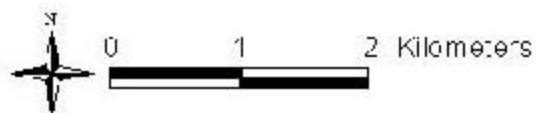


Figure 3. Pre-1750 distribution of Ecological Vegetation Classes on Glendamar

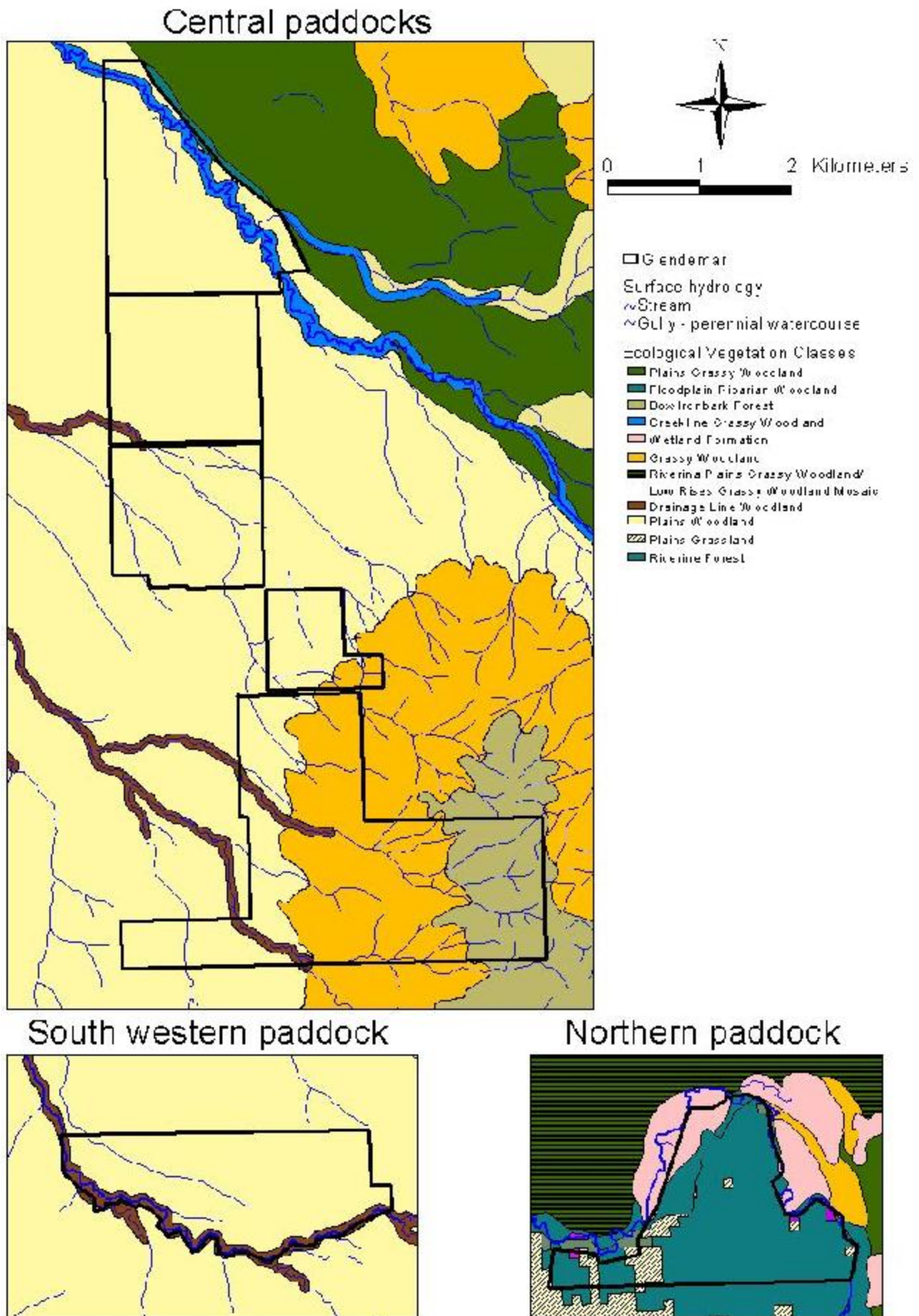
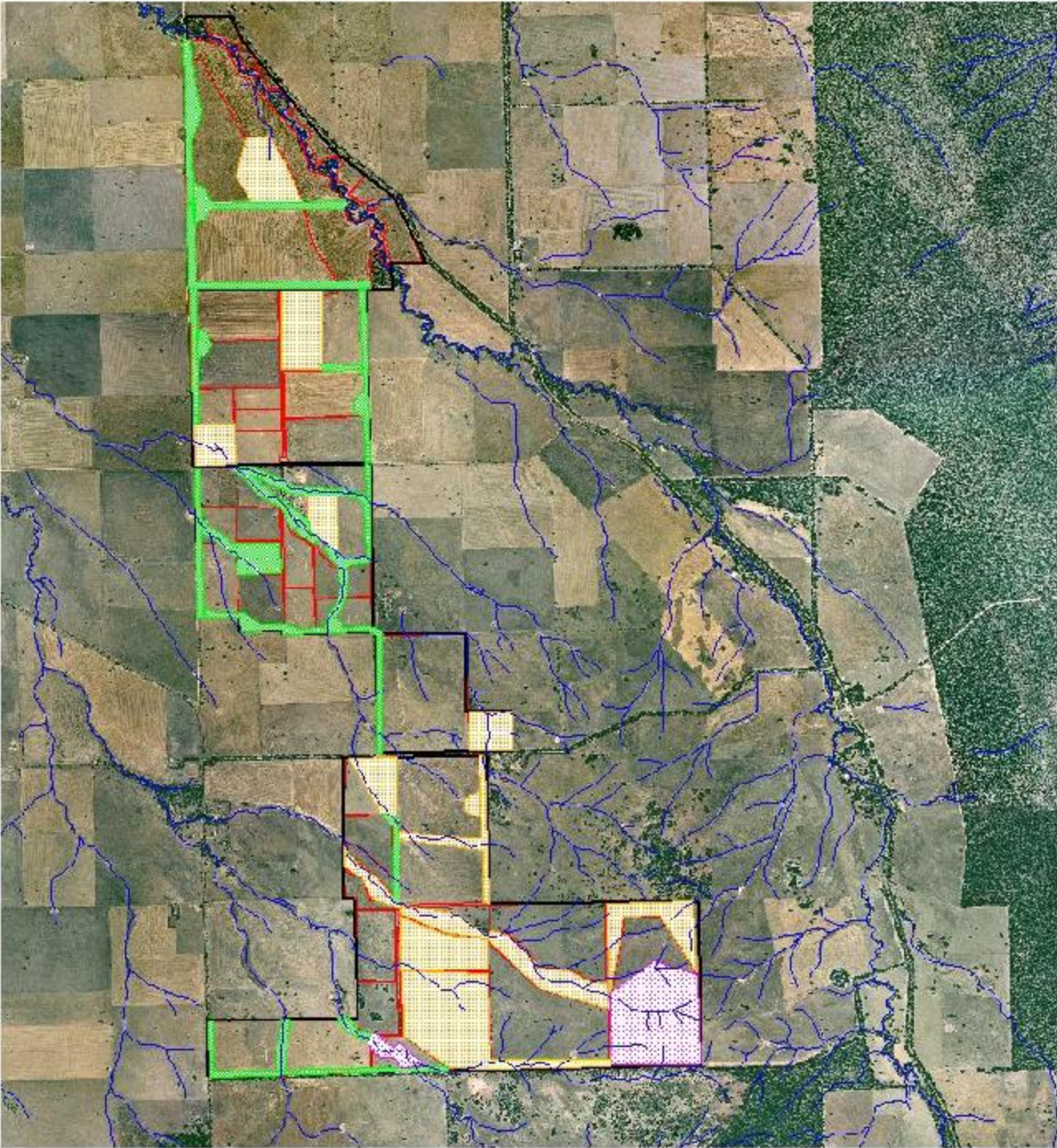


Figure 4a. Structural restoration plan for "Glendemar" - central block.

Central paddocks



- Glendemar
- Paddock layout:
- Surface hydrology
 - ~ Stream
 - ~ Gully - perennial watercourse
- Restoration action
 - Improve
 - Regenerate
 - Replant

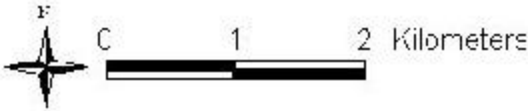
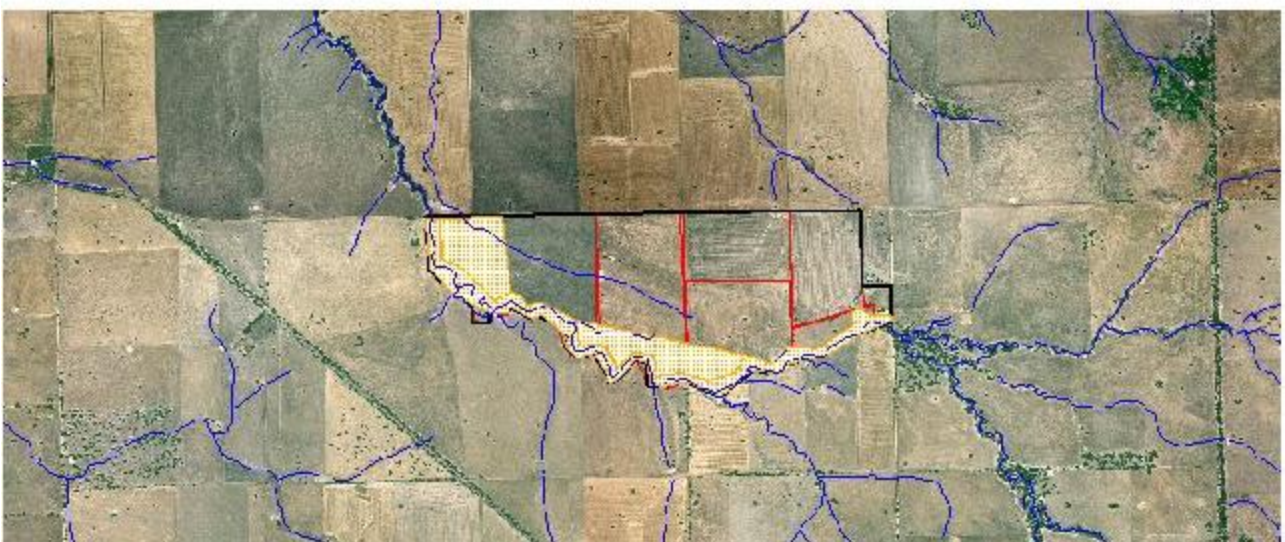


Figure 4b. Structural restoration plan for “Glendemar” – northern and south-western blocks.

Northern paddock



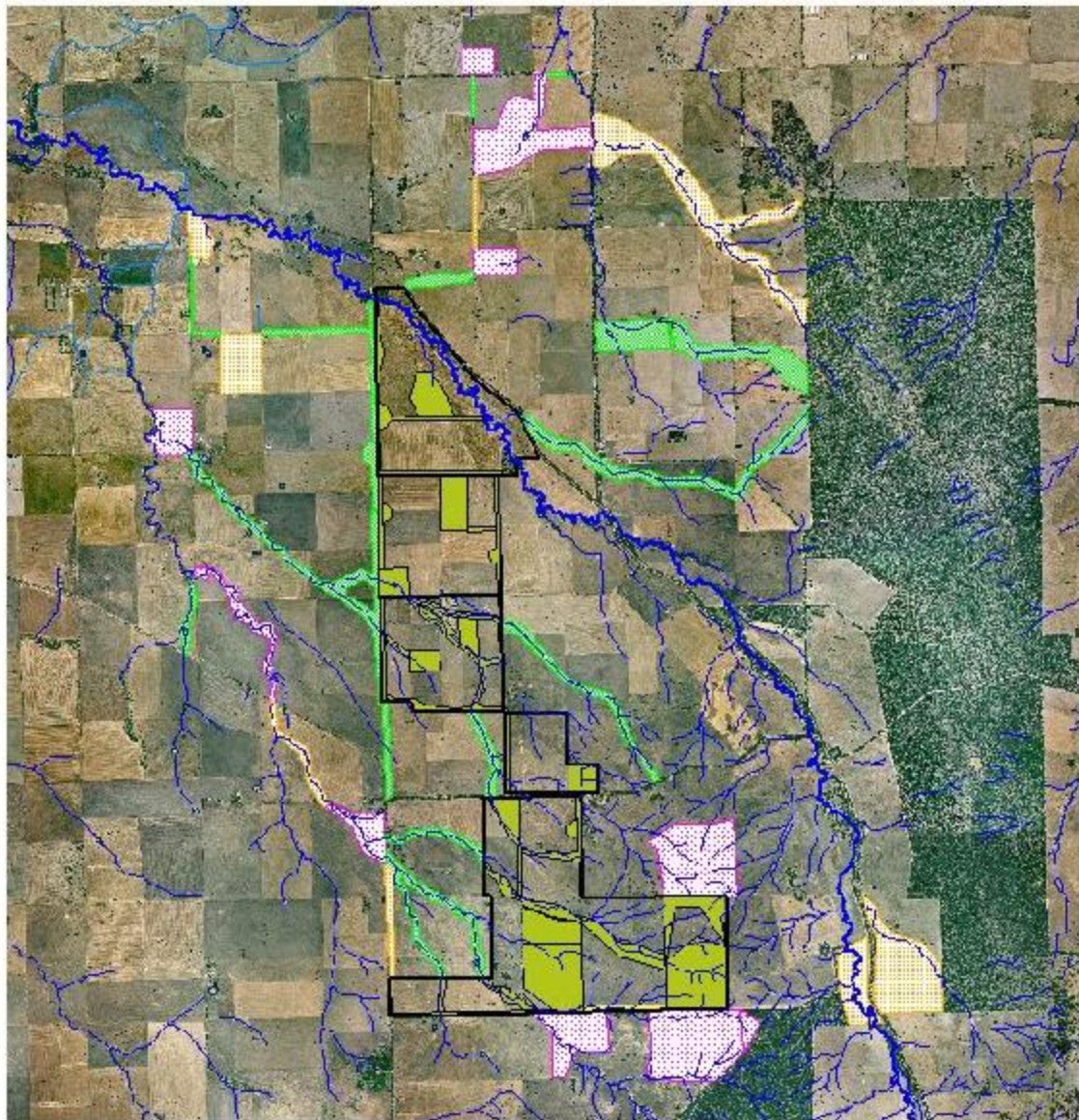
South western paddock



- Glendemar
- Paddock layout
- Surface hydrology
- ~ Stream
- ~ Gully - perennial watercourse
- Restoration action
- Improve
- Regenerate
- Replant



Figure 5. Restoration actions on neighbouring properties.



- ▭ Glendemar
- Surface hydrology
- ~ Stream
- ~ Gully - perennial watercourse
- Glendemar restoration
- Restoration action
- Improve
- Regenerate
- Replant

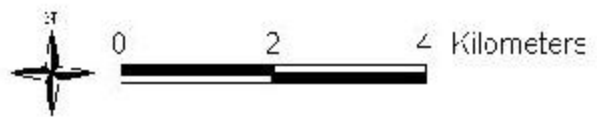


Figure 6. Current layout and conservation status of extant native vegetation on Nil Desperandum and Elmswood.

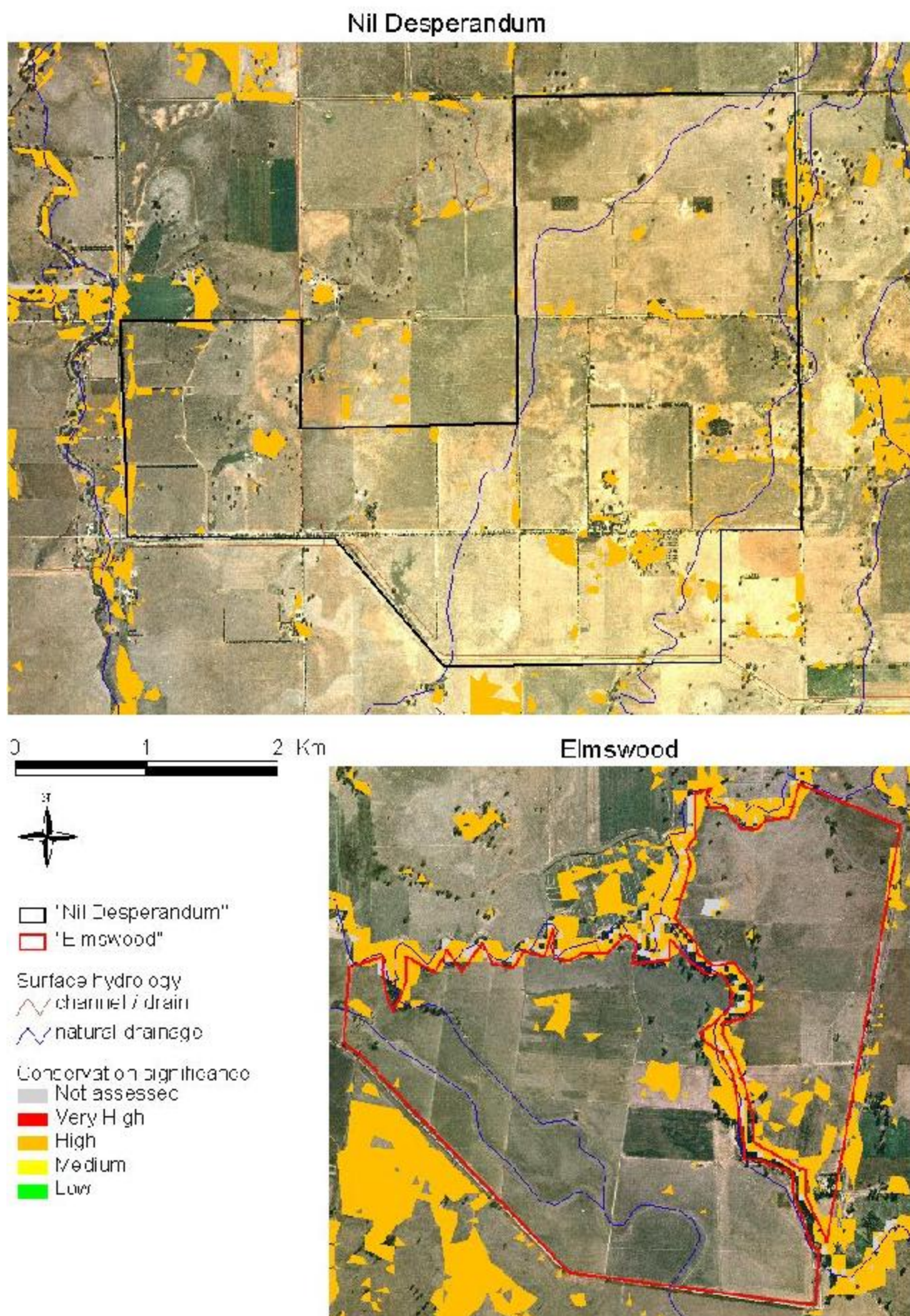


Figure 7. Pre-1750 distribution of Ecological Vegetation Classes on Nil Desperandum and Elmswood.

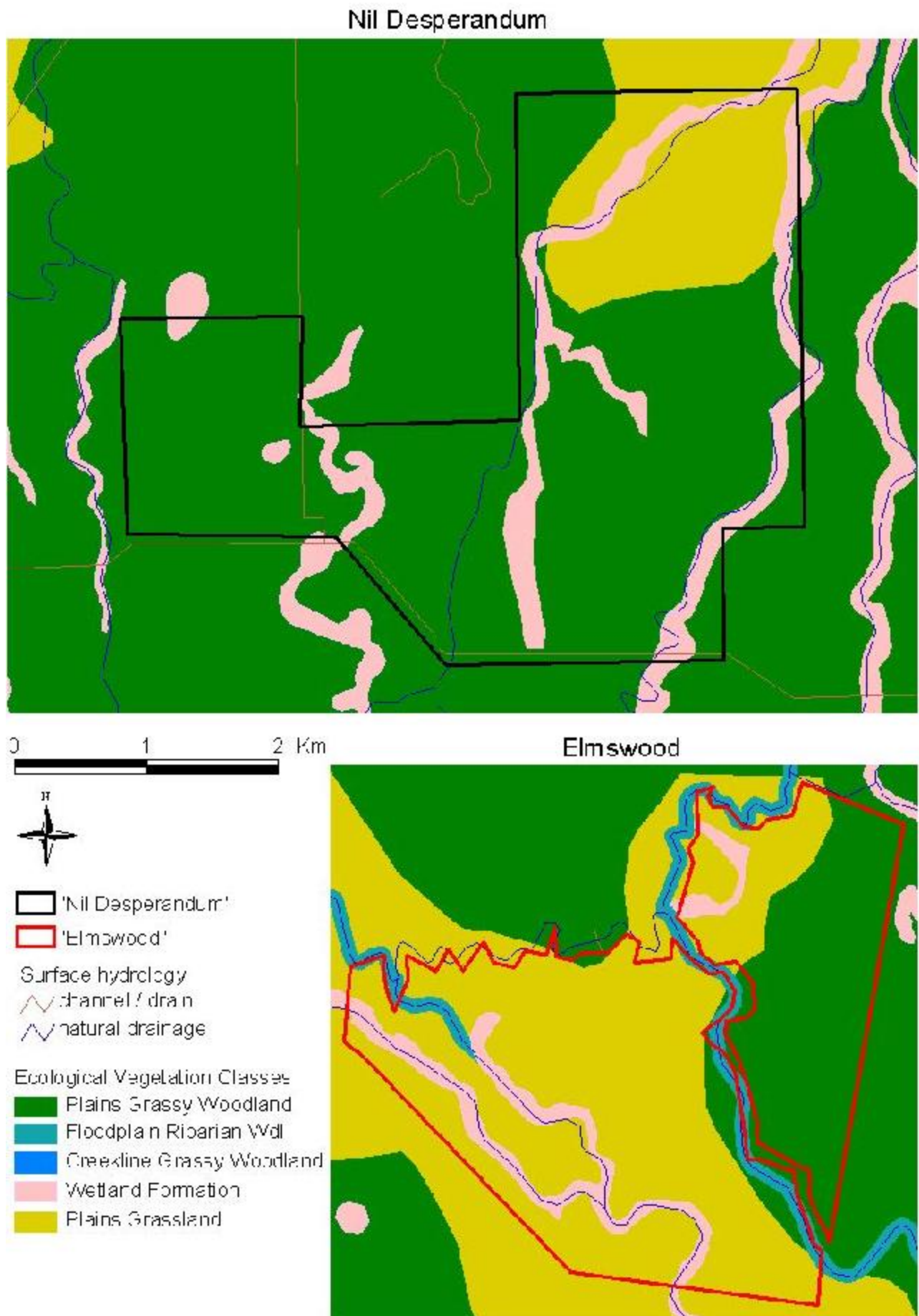


Figure 8. Restoration projects that have been completed on Nil Desperandum and Elmswood.

