Ecological restoration of farmland: progress and prospects

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Sustainable agricultural practices in conjunction with ecological restoration methods can reduce the detrimental effects of agriculture. The Society for Ecological Restoration International has produced generic guidelines for conceiving, organizing, conducting and assessing ecological restoration projects. Additionally, there are now good conceptual frameworks, guidelines and practical methods for developing ecological restoration programmes that are based on sound ecological principles and supported by empirical evidence and modelling approaches. Restoration methods must also be technically achievable and socially acceptable and spread over a range of locations. It is important to reconcile differences between methods that favour conservation and those that favour economic returns, to ensure that conservation efforts are beneficial for both landowners and biodiversity. One option for this type of mutual benefit is the use of agri-environmental schemes to provide financial incentives to landholders in exchange for providing conservation services and other benefits. However, further work is required to define and measure the effectiveness of agri-environmental schemes. The broader potential for ecological restoration to improve the sustainability of agricultural production while conserving biodiversity in farmscapes and reducing external costs is high, but there is still much to learn, particularly for the most efficient use of agri-environmental schemes to change land use practice.

Keywords: agri-environmental schemes; biodiversity; ecological restoration; external costs; sustainable agriculture; wildlife conservation

1. INTRODUCTION

The expansion and intensification of agricultural land is recognized as a major driver of contemporary global environmental change (Meyer & Turner 1992; Matson *et al.* 1997; Stoate *et al.* 2001; Benton *et al.* 2003; Baillie *et al.* 2004). Humans have converted an estimated 38.2% or 4973 million ha of the Earth's land surface area to agriculture (i.e. temporary or permanent crops and pastures) at the expense of natural habitats (FAOSTAT 2003). Unabated, this figure is forecast to reach 60% in the next 100 years. This conversion is regarded as being unparalleled by any other human-induced change in its combination of spatial extent and intensity of influence (Matson *et al.* 1997). Land cover change and intensification can vastly affect biological diversity, trace-gas emissions, the quality and flow of water, soil condition and climate at both local and regional scales that extend beyond farm boundaries (Meyer & Turner 1992; Benton *et al.* 2003; Cramer & Hobbs 2005; Tscharntke *et al.* 2005). Critically, this affects the delivery of cultural, provisioning, regulating and basic supporting types of ecosystem services, such as biological control, food production, gas regulation, nutrient cycling, pollination and water supply, which more generally benefit mankind but are also partly

provided and received by agroecosystems (Costanza *et al.* 1997; Pereira & Cooper 2006). Indeed, with policy decision makers as the intended audience, the recent Millennium Ecosystem Assessment (2005) highlighted the state of global ecosystems and their role for human well being. The assessment, a monumental effort of some 1360 scientists from 95 countries over 5 years, examined 24 ecosystem services and found that productivity of only four have been enhanced over the last 50 years (i.e. global climate regulation and production of aquaculture, crops and livestock), whereas 15 have been degraded (e.g. aesthetic values, air and water purification, biological pest control, freshwater supply and pollination; Millennium Ecosystem Assessment 2005; see also Stokstad 2005).

Besides the increasing concern over the impacts of modern farming practices on ecosystem services or function, there is also a general perception that the sustainability of agroecosystems themselves is under threat (Meyer & Turner 1992; Matson *et al.* 1997; Stoate *et al.* 2001; Robinson & Sutherland 2002; Tilman *et al.* 2002; Millenium Ecosystem Assessment 2005). Ecological restoration of farmland can contribute to sustainable agriculture by moving degraded ecosystems closer to their former state and thereby restoring ecosystem function. But what exactly is meant by ecological restoration and sustainable agriculture, and what does the former involve? This review

begins with a working definition of each of these key terms (i.e. *what*). We describe the consequences of agricultural expansion and intensification to put into perspective why farmland ecological restoration may become necessary (i.e. *why*?). We then provide examples of focal plant and animal taxa that are targeted in farmland restoration programmes, often owing to their high conservation status or general sensitivity to impacts and hence as potential indicators of environmental condition (i.e. *which*?). We then explore the locality of farmland ecological restoration sites, meaning the prominent habitats, regions and countries (i.e. *where*?). This is followed by an examination of the timing of ecological restoration activities on farmland (i.e. *when*?). We then provide an overview of the different strategies, or tools that can be used and of the research done to develop or support these strategies (i.e. *how*?). As part of this section, we examine the various approaches that involve the use of flowering plants and permanent refuges, as these have attracted considerable interest among researchers. In doing so, we draw a distinction between 'shotgun' and 'directed' approaches for managing plant biodiversity. The discussion focuses on the ecological mechanisms by which restored or newly created areas operate rather than on the full range of socio-economic factors that must be evaluated before the methods can be implemented. We then include an evaluation of the conceptual frameworks and methods used to gauge restoration success. We conclude with an overview of the wide range of agri-environmental schemes in place, at least in 'developed' countries, to provide incentives for landholders to conduct farmland ecological restoration.

2. *WHAT* IS ECOLOGICAL RESTORATION OF FARMLAND?

The goal of ecological restoration is to shift an ecosystem towards its pre-disturbed state with respect to ecosystem structure, function and composition (Hobbs & Norton 1996). The approach emphasizes the use of quantitative practices for measuring and restoring ecosystem 'health', including its ability to deliver ecosystem services (Costanza *et al.* 1997). Sustainable agriculture 'refers to the ability of a farm to continue producing indefinitely with a minimum of outside inputs' (Anon. *a*), or put another way, 'is defined as agriculture that meets the needs of the present generation while conserving resources for the use of future generations.' (Anon. *b*). The continuity of production by using minimal inputs and creating few negative effects is emphasized. 'Farmland' primarily refers to the land use comprising temporary or permanent crops and pastures. For the purposes of the review, this also includes non-crop vegetation, such as hedgerows and remnants of native vegetation, and waterways that are situated on farmland, but not plantation timber or farm forestry. Although farmland is often derived from grassland and woodland, these habitat types *per se* are generally excluded from the review unless the principles involved in the restoration of these habitats are relevant to that of farmland (Hooper *et al.* 2002; Ryan *et al.* 2002).

3. *WHY* IS ECOLOGICAL RESTORATION OF FARMLAND NECESSARY?

What causes farmland to become degraded and what are the symptoms of farmland in need of ecological restoration? Farmland and its environs are susceptible to inadvertent or deliberate degradation in their physical, chemical and biological condition by a range of farming activities that primarily result in changes to air quality, biological diversity, climate, soil condition and the quality and quantity of water (reviewed by Meyer & Turner 1992; Matson *et al.* 1997; Stoate *et al.* 2001; Robinson & Sutherland 2002; Tilman *et al.* 2002; Benton *et al.* 2003; Millennium Ecosystem Assessment 2005). Soil erosion results from the loss of vegetation cover due to burning, grazing and cultivation. Changes in the fertility, structure, acidification and salinization of soils are caused by cultivation, drainage, irrigation and tree removal. Pollution of ground water and eutrophication of rivers and lakes results from off-farm movement of silt, pesticides and nutrients, i.e. fertilizers or animal effluent. Flow rate of rivers is affected by the construction of weirs and levée-banks, diversion of overland water flows to on-farm reservoirs and direct removal for irrigation. There are global impacts on atmospheric constituents (principally carbon dioxide, methane and nitrogen dioxide) and climate (chiefly temperature and rainfall) as a result of forest removal, biomass burning, fertilizers and livestock. Finally, land cover changes lead to both habitat loss and fragmentation, which threaten aquatic and terrestrial taxa (Meyer & Turner 1992; Matson *et al.* 1997; Stoate *et al.* 2001; Robinson & Sutherland 2002; Tilman *et al.* 2002).

Symptoms of degraded farmland include algal blooms and pesticide residues in waterways, pest outbreaks, plant disease epidemics such as 'rural dieback' of native Australian eucalypts, which is principally caused by the root rot fungus *Phytophthora cinnamomi*, and disease epidemics of livestock, such as foot and mouth disease and influenza A virus (H5N1, 'bird flu'). In addition, there is evidence of yield decline, loss of topsoil through water and wind, hedgerows and field margins removed or sprayed with herbicides, and a general reduction in species richness and abundance of plants and animals (Wills 1993; Stoate *et al.* 2001; Tilman *et al.* 2002; Millennium Ecosystem Assessment 2005).

Importantly, agricultural practices have both local and landscape-scale impacts that transcend farm boundaries (Meyer & Turner 1992; Benton *et al.* 2003; Cramer & Hobbs 2005; Tscharntke *et al.* 2005). Local intensification includes adverse effects such as shortened crop rotation cycles and increasing input of agrochemicals. On a landscape scale, fields have been amalgamated and enlarged, resulting in simplified landscapes with few or no non-crop habitats remaining (Tscharntke *et al.* 2005). The total annual external (off-farm) costs of agriculture on natural resources (air, soil and water), biodiversity and human health (pathogens and pesticides) have been estimated for the United Kingdom at £1149–3907 million between 1990 and 1996 (Pretty *et al.* 2000) and £1514 million in 2000 (Pretty *et al.* 2005), and for the USA at £3256–9678 million in 2002 (Tegtmeier & Duffy 2004). This equates to £208 ha^{-1} of arable land and permanent
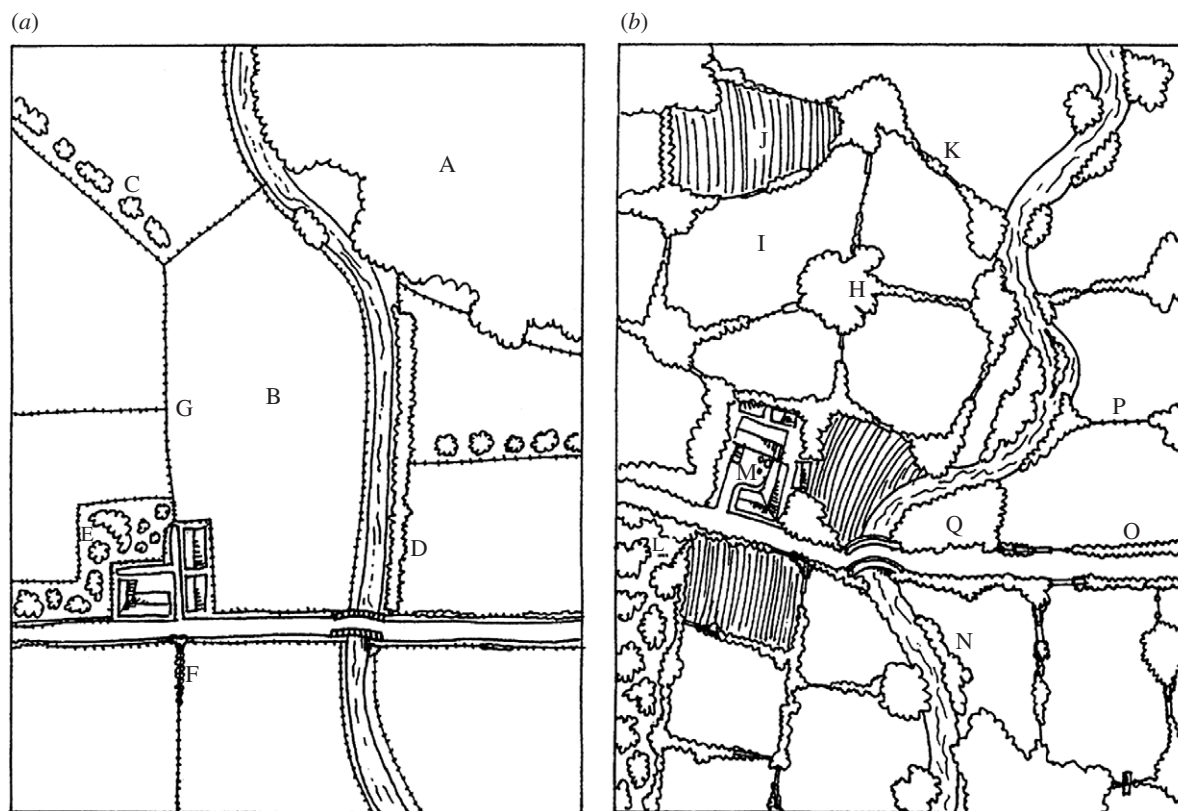
Figure 1. Contrast between (*a*) a simple (or homogeneous) and (*b*) a complex (or heterogeneous) farm landscape. Note the heterogeneity of habitats and the connectance features in the complex landscape compared with the uniformity of the simple farm landscape. A, indigenous plant reserve: these tend to be large tracts of land not integrated with farmland; B, pasture and exotic grasses; C, typical shelter belt (e.g. popular, *Cupressus macrocarpa* and *Pinus radiata*); D, riparian vegetation (e.g. willow, grasses and some indigenous species); E, farmhouse garden; F, small areas of patchy gorse, *Ulex europaeus*; G, wire fences: common field boundaries; H, small wood lot: a highly used but sustained feature; I, pasture; J, ploughed field; K, hedge fence; L, orchard; M, farmhouse garden; N, riparian vegetation; O, roadside vegetation, hedges, trees, etc; P, wire fences or stone walls; and Q, woodland. Modified from Keesing & Wratten (1997).

pasture for the United Kingdom (UK) in 1996 and £17–55 ha^{-1} of arable land in the USA.

To put these costs into perspective with the external benefits provided by agriculture, the pivotal paper by Costanza *et al.* (1997) calculated the combined economic value of three ecosystem services (biological control, pollination and food production) from *worldwide* cropland to be USD$128.8 (£73.802) billion per year or USD$92 (£53) per ha. A caveat here is that Costanza *et al.* (1997) assigned a nil value to the ecosystem service of habitats or refugia for resident and transient taxa because this service 'do(es) not occur or (is) known to be negligible', hence the true value of cropland is likely to be underestimated. Nevertheless, by these calculations, the worldwide ecosystem service benefits from agriculture are estimated to be £53 ha^{-1} yr^{-1}, yet the external costs of intensive agriculture in countries like the UK are £208 ha^{-1} yr^{-1}. Equally compelling calculations estimated that the economic benefit to world society from biodiversity is USD$2928 billion. This value included the benefits of activities such as biological pest control, ecotourism, pollination and waste disposal (Pimentel *et al.* 1997). It is evident that more sustainable agricultural practices in conjunction with ecological restoration methods on farmland are necessary to reduce the unacceptably high external costs of agriculture that are borne by the community. In addition, 'ecological engineering'

techniques are available to enhance ecosystem services on farmland, including habitat manipulation tactics for beneficial arthropods that are responsible for biological pest control and contribute to biodiversity in general (see §7 and Gurr *et al.* 2003, 2004).

Agriculture and biodiversity conservation have been traditionally viewed as incompatible, with agriculture considered a major driver of species loss for many plant and animal taxa, such as bumble-bees (*Bombus* spp.) and bird species like skylarks (*Alauda arvensis* L.) since 1945 (Stoate *et al.* 2001; Robinson & Sutherland 2002). Agriculture represents the dominant land use throughout much of western Europe and a significant part of European biodiversity is associated with this habitat. Agroecosystems, however, are very hostile to a wide diversity of species owing to the conversion of complex natural ecosystems to simplified managed ones and the intensification of resource use. Firstly, there is a tendency for simplified cropping systems to be applied to increasingly consolidated land areas, leading to the loss of non-crop habitats, such as field margins and hedgerows, together with the decline in traditional mixed arable and livestock farming. As a result, remnant native vegetation has become fragmented into different patches and there are fewer 'nodes' where field corners join (figure 1). These nodes can be rich 'hotspots' of invertebrate, vertebrate and plant diversity (Keesing & Wratten 1997). Secondly, there is

intensification of resource use in the cropping systems themselves, including greater pesticide and fertilizer usage and shorter fallow periods (Stoate *et al.* 2001; Pywell *et al.* 2005*a*). However, more recently, there has been an important move beyond conservation efforts to an appreciation of the value of natural, undisturbed remnants and to a better recognition of the role that highly modified landscapes play in maintaining native biodiversity (Tscharntke *et al.* 2005). As Novacek & Cleland (2001) pointed out 'we are obviously past any point where strategies that focus on preservation of 'pristine' habitats are sufficient for the job. Greater attention must be placed on human-dominated landscapes that … (surround) the less disrupted areas'. In this way, agriculture can make important contributions to high-diversity habitats, while also benefiting from ecosystem services provided from different land use types. We know that invertebrate natural enemies of crop pests visit different habitat types before colonizing agricultural fields (Silberbauer *et al.* 2004) and improved biological pest control and crop pollination may directly increase farmers' income (Östman *et al.* 2003; Ricketts *et al.* 2004).

While the rate of conversion of land to agriculture has slowed in more developed countries like the United Kingdom (Robinson & Sutherland 2002), ecosystem function in fragmented remnant vegetation often remains severely disturbed due to continuing effects of microclimatic change and isolation of taxa (Saunders *et al.* 1991). Saunders *et al.* (1991) pointed out that populations that are too small to be viable might still persist for long periods, simply owing to the longevity of remaining individuals. For example, remnant native areas of the Western Australian wheat belt, which now represent only 7% of 14 million ha (Saunders *et al.* 1993), contain female trapdoor spiders *Anidiops villosus* (Rainbow) that can live for over 23 years (Main 1987, cited by Saunders *et al.* 1991). Thus, the presence alone of a species in a remnant is no guarantee of its continued existence there, as it also requires successful reproduction and recruitment, the success of which can be revealed by a closer examination of the species' age structure. A key objective in conservation and restoration ecology is the determination of the minimum viable population sizes for a given habitat size that can sustain a given community of plants and animals, as well as the metapopulation effects of fragment shape and position in the landscape (Simberloff & Cox 1987; Saunders *et al.* 1991). Vegetation 'corridors' that link otherwise isolated patches have been suggested for improving the persistence of taxa, and there is a general consensus that they are beneficial (Simberloff & Cox 1987). Thus, agroecosystems do support biodiversity that contributes overall to ecosystem services such as crop pollination and biological pest control, and ecological restoration can to some extent ameliorate the effects of agricultural expansion and intensification. Finally, while farmland ecological restoration methods may serve their immediate purpose of contributing to sustainable agriculture, other non-restoration or 'multi-function' (Gurr *et al.* 2003) benefits may be achieved. For example, the advantages of planting trees on farmland include: bioenergy production, carbon sequestration, erosion control, habitat restoration, increased water use (reduced secondary salinity) and wood production (farm forestry; Ryan *et al.* 2002), as well as aesthetics, cultural, landscape conservation and other benefits.

4. WHICH ARE THE TARGET SPECIES OF FARMLAND ECOLOGICAL RESTORATION?

There has been considerable debate in the ecological restoration literature about whether the requirements of single species should be the basis of designing conservation programmes, or whether the analysis of landscape attributes that support entire communities should underpin conservation planning. In a single-species approach, rare or vulnerable species or groups of species that are considered to represent important components of biodiversity are often the targets. But it has been argued that the single-species approach can be costly and ineffective in dealing with the urgency of threats to ecosystems and their functioning. Conversely, it is considered unwise to ignore the requirements of individual species when seeking to define the attributes of an improved landscape that will ensure community survival via enhancement of species richness and evenness (reviewed by Lambeck 1997). The concepts of 'flagships', 'umbrellas', 'biodiversity indicators', 'focal-species' and other classes of taxon-based restoration schemes have been developed by the proponents of the single-species approach as an alternative to studying each species separately. In focal-species schemes, the species are purportedly selected on the basis that their requirements encapsulate those of the regional taxa. In practice, the species are either classified as being area sensitive, dispersal limited, resource limited or limited by ecological processes such as fire and predation. However, critics have challenged the underlying validity of the assumption that such taxa can be adequately chosen to represent the regional taxa (Andelman & Fagan 2000; Lindenmayer *et al.* 2002). For instance, Andelman & Fagan (2000) examined the outcome of how well threatened, rare or 'of concern' species that were deliberately selected for inclusion in at least 14 different surrogate schemes (e.g. large carnivores, charismatic species, habitat specialists) actually do represent the interests of the total pool of species across all schemes. Also, the area of habitat that would be protected compared with conserving suites of species that were randomly selected from each of three conservation databases is a concern. The results showed that the various surrogate schemes were generally poor at protecting the total background pool of taxa (Andelman & Fagan 2000). As an alternative to a focal-species approach alone, Lindenmayer *et al.* (2002) urged the adoption of a mix of approaches across different landscapes, leading to economically, politically and socially acceptable results. This form of a bet-hedging strategy would help in the event that a single approach was unsuccessful in a given location. Examples include habitat restoration to a nominal proportion of vegetation cover, expansion of the size of existing patches of remnant vegetation, and restoration of a watercourse and associated riparian vegetation. Lindenmayer *et al.* (2002) urged that the response of

a large set of species within ecosystems should be measured in response to ecological restoration changes. They also highlighted the need for continued basic autecological research, although data intensive, to provide the foundations for ecological restoration practices, such as (i) abundance and distribution changes, (ii) dispersal ability, (iii) the influence of adjoining land use types on behaviour, (iv) the spatial scale at which organisms respond to disturbances such as fire and grazing, and (v) the cumulative effects of disturbance. Further, it was recognized that it is better to explain openly than to hide the complexities and uncertainties involved in ecological restoration and to desist from the promotion of 'simple rules' for restoration to landholders, like the focal-species approach (Lindenmayer *et al.* 2002). The complexities of species-diverse food webs are well illustrated by Snyder *et al.* (2006), who considered the concepts of species identity (i.e. the 'sampling' effect) and niche complementarity when managing biodiversity to enhance pest suppression in farmland.

Despite the above cautions, it is insightful to consider the taxa that have been the subject of single- or focal-species farmland ecological restoration approaches. These taxa can be assigned to eight broad categories and examples for each are provided in table 1. Often the target species are classified as rare or threatened (at risk of extinction) based on their population size and trend, like the skylark (*A. arvensis*) in Europe. Host or prey species that are important in the diet of the species under conservation are also targeted, such as various insects. Conversely, predator species that threaten the existence of species under conservation are targeted, such as the red fox (*Vulpes vulpes* L.) and ferret (*Mustela furo* L.). Species may be considered good indicators of habitat quality, such as butterflies and insectivorous bats (e.g. *Pipistrellus pipistrellus* (Schreber)). Other species may be common and fulfil important ecosystem services, such as biological control, nutrient cycling and pollination. Examples include species of carabid beetles and bumble-bees. Finally, attempts are made to reduce populations of non-native or native species that cause disruption to agro-ecosystem function in a given locale; examples are a freshwater fish the common carp (*Cyprinus carpio* L.) and lantana weed (*Lantana camara* L.). It is recognized that species that fall into the above category could also be classed as agricultural or invasive pests, which are categories not represented here, but the intention is to demarcate targets of agricultural pest management and biosecurity eradication programmes within farmland ecological restoration. An evaluation of agri-environmental schemes in Europe recorded the response of birds in 29 published studies, plants in 20, insects and spiders in 20 and mammals in only one (this review included six studies involving more than one taxonomic group; Kleijn & Sutherland 2003).

A salient consideration in ecological restoration is the concepts of multitrophic interactions and polyphagy. The work of Prasad & Snyder (2006) is a recent example of how 'beetle banks' (raised strips of unsprayed and uncultivated perennial grasses) increased predatory beetle (Coleoptera: Carabidae and Staphylinidae) abundance adjacent to the banks,

but not rates of predation on fly eggs due to intraguild predation and the presence of alternative prey in this area. The presence of the noisy miner (*Manorina melanocephala* Latham), an aggressive bird species native to Australia, can affect avian diversity and abundance in remnant patches of woodland on Australian farmland. Moreover, their presence is believed to exclude small insectivorous birds that play an important role in reducing the abundance of leaf-feeding insects that contribute to 'rural dieback' of native trees like eucalypts (Grey *et al.* 1997).

There seems to be no worldwide compendium of species that have been targeted in farmland ecological restoration programmes. This is unfortunate, but is certainly a task that is beyond the scope of this review. It probably reflects the wide range of habitat types that farmland as a land use type occupies. There are, however, various publications that list the conservation status of individual species present by biome, habitat or ecoregion, such as grassland, savannah and shrubland (Olson *et al.* 2001; Baillie *et al.* 2004), which would represent farmland. The most notable publication is the IUCN World Conservation Union's Red List of Threatened Species, which presently recognizes 15 589 species that are seriously threatened with extinction of the 38 047 assessed species (i.e. 41% threatened; Baillie *et al.* 2004). The seriously threatened species comprise 5274 species of vertebrates, 1992 invertebrates, 8321 vascular plants and two lichens from a range of countries across the world. However, it is claimed that this figure is an underestimate of the total number of threatened species as it is based only on an assessment of less than 3% of the world's 1 545 594 described species (Baillie *et al.* 2004). This key publication has inspired numerous separate listings for various countries. For example, the population status of 247 bird species has been assessed in the United Kingdom; 40 species are critically threatened ('red-listed'), 121 are moderately threatened ('amber-listed') and 86 are rare or not presently threatened ('green-listed'; Gregory *et al.* 2002). An action plan for Australian butterflies also exists, which is aimed at conserving threatened butterfly species and boosting their numbers (Sands & New 2002). The action plan reviewed 220 of Australia's 654 butterfly species and found that 26 were threatened. For each species reviewed, there is a brief commentary on its distribution, specific habitat requirements, major threatening processes, historic and present conservation status and a recovery action plan (Sands & New 2002). An unfortunate common feature of the above lists of threatened organisms is the recognized inability to formulate the conservation status and recovery plans for a large number of species either because few have been assessed or owing to a deficiency of data. Further, in many cases, it is not clear what the definitive cause of population decline was, although a wide variety of threats are generally implicated. The loss of habitat from agricultural expansion is generally the main reason suggested, but the evidence can be weak if it is based on anecdotal observations, unpublished data or 'grey' literature such as annual reports rather than empirical evidence in refereed scientific publications. Nevertheless, these lists are regarded as an extremely

Table 1. Examples of plant and animal taxa that have been targeted in farmland ecological restoration programmes.

rationale	target group	species	country	reference
threatened or rare	farm birds	corn bunting (*Miliaria calandra*); skylark (*Alauda arvensis*); yellowhammer (*Emberiza citronella*)	United Kingdom	Wakeham-Dawson & Aebischer (1998); Brickle *et al.* (2000) and Thomas *et al.* (2001)
	game birds	grey partridge (*Perdix perdix*); pheasant (*Phasianus colchicus*); red-legged partridge (*Alectoris rufa*)	France	Bro *et al.* (2004)
	raptors	hen harrier (*Circus cyaneus*); NZ falcon (*Falco novaeseelandiae*)	United Kingdom, New Zealand	Amar & Redpath (2005) and McKinnon (2005a)
	mammals	field vole (*Microtus agrestis*); harvest mouse (*Micromys minutus*)	United Kingdom	Robinson & Sutherland (2002)
	insects and other invertebrates	bumble-bees (*Bombus* spp, Hymenoptera); butterflies (Lepidoptera); wetas (Orthoptera)	Australia, New Zealand, United Kingdom	Goulson *et al.* (2002); Sands & New (2002); Carvell *et al.* (2004); Pywell *et al.* (2005b) and Bowie *et al.* (2006)
	plants	kanuka (*Kunzea ericoides*); kowhai (*Sophora microphylla*); totara (*Podocarpus totara*)	New Zealand	McKinnon (2005b) and Fiedler (2006); S.D.W. 2005, personal observation
prey ('chick-food')	insects and other invertebrates	beetles (Coleoptera); butterflies (Lepidoptera); grasshoppers (Orthoptera); harvestmen (Opiliones); spiders (Araneae); true bugs (Hemiptera)	United Kingdom	Brickle *et al.* (2000)
predators of threatened species	mammals	American mink (*Mustela vison*); Australian brushtail possum (*Trichosurus vulpecula*)[a], badger (*Meles meles*); ferret (*Mustela furo*)[a]; hedgehog (*Erinaceus europaeus*); weasel (*Mustela nivalis*); red fox (*Vulpes vulpes*); stoat (*Mustela erminea*)	France, New Zealand, United Kingdom	Byrom (2002); Bro *et al.* (2004) and Evans (2004)
indicator of habitat quality	birds	Port Lincoln ringneck (*Platycercus zonarius*); singing honeyeater (*Lichenostomus virescens*); yellow-throated miner (*Manorina flavigula*)	Australia	Fortin & Arnold (1997)
	insects and other invertebrates	beetles (Coleoptera); butterflies (Lepidoptera); flatworms (Turbellaria); spiders (Araneae)	Finland, New Zealand, United Kingdom	Bowie & Frampton (2004); Pywell *et al.* (2004, 2005a) and Pöyry *et al.* (2005)
	mammals	bats (*Pipistrellus pipistrellus*)	United Kingdom	Wickramasinghe *et al.* (2003)
	vascular plants	heather (*Calluna vulgaris*)	Denmark	Degn (2001)
biological control agents	insects and other invertebrates	beetles (Coleoptera: Carabidae, Staphylinidae); spiders (Araneae); wasps (Hymenoptera)	Australia, New Zealand, United Kingdom, USA	Thomas *et al.* (1991); Baggen *et al.* (1999); Hossain *et al.* (2002); MacLeod *et al.* (2004) and Prasad & Snyder (2006)
decomposers	insects and other invertebrates	dung beetles (Coleoptera: Scarabaeidae)	Ireland	Hutton & Giller (2003)
pollinators	insects and other invertebrates	bumble-bees (*Bombus* spp, Hymenoptera); hoverflies (Diptera)	New Zealand, United Kingdom	Carvell *et al.* (2004) and Pontin *et al.* (2006)
damaging (non-native or native)	birds	noisy miner (*Manorina melanocephala*)	Australia	Grey *et al.* (1997)
	fishes	common carp (*Cyprinus carpio*)	Australia	Koehn (2004)
	insects and other invertebrates	honeybee mite (*Varroa destructor*, Acari: Varroidae)	New Zealand	Zhang (2000)
	mammals	European rabbit (*Oryctolagus cuniculus*)	New Zealand	Byrom (2002)
	plants	lantana (*Lantana camara*)	Australia	Macleay (2004)

[a] Also regarded as pest species because they vector bovine tuberculosis to cattle and deer in New Zealand (Byrom 2002).

valuable tool for conservation planning, management, monitoring and decision making (Rodrigues *et al.* 2006). However, what is the relevance of these lists for farmland ecological restoration? Key values are that they provide a framework to help identify species of conservation importance, suggest causes for decline and propose a recovery plan. In this context, it would be very useful if they explicitly listed which species also colonize farmland within a recognized habitat type, like native grassland, but they appear not to do this.

Empirical research has been used to support ecological restoration programmes by determining which species are more or less suitable for restoration. For instance, large-seeded and shade-tolerant native tree species were recommend rather than small-seeded and shade-sensitive species for reforestation of abandoned farmland in Panama (Hooper *et al.* 2002). Intrinsic biological traits for a range of taxa have been identified as good correlates of susceptibility to local extinction (Purvis *et al.* 2000). Depending on the taxa, susceptible traits include low abundance, high habitat specificity, large body size and slow reproductive rates. Interplay exists between these traits with external threats; for example, habitat loss tends to affect mostly those species that are ecologically specialized, while introduced predators may have more of an impact on species with long generation times. The relative contribution of external threats and intrinsic traits to extinction risk has been investigated for some taxa; among mammals approximately 50% of the variation in extinction risk is explained by variation in species' biological traits (Purvis *et al.* 2000), with the remainder being attributable to human pressures and the interactions between these and biological traits.

5. *WHEN* IS ECOLOGICAL RESTORATION CONDUCTED?

It is intuitive that ecological restoration of degraded farmland should be initiated as soon as it is feasible and subsequent monitoring may last an indefinite period. In practice, however, monitoring rarely lasts for more than 5 years, even though this may be too short a period (Ruiz-Jaen & Aide 2005). The specifics of determining how long the ecological restoration takes in terms of reaching an 'endpoint' and how frequently monitoring should be conducted is highly site-specific and rather controversial, partly because it is hard to determine (Ormerod 2003; Ruiz-Jaen & Aide 2005). This issue will be discussed further in §7. Nevertheless, the ecological succession of plant and animal communities has often been the focus of several farmland restoration assessments. One of the best examples is provided by Degn (2001), who examined the succession of vascular plant communities from farmland to heathland over a period of 22 years after cultivation ceased. The succession of plants and arthropods has also been studied in beetle banks over 7 years (MacLeod *et al.* 2004). The grasses *Agrostis stolonifera* L., *Dactylis glomerata* L. and *Holcus lanatus* L., but not *Lolium perenne* L., became well established when sown in single-species plots, and in mixtures *L. perenne* was completely replaced with *D. glomerata* by the third year. The abundance of predatory carabid and staphylinid

beetles (Coleoptera) and linyphiid and lycosid spiders (Araneae) fluctuated over the seven consecutive winters; densities tended to be higher in the second winter than in the first, then fell in the third and fourth winters, followed by rises in the fifth and sixth and fell again in the seventh (MacLeod *et al.* 2004). A positive relationship was detected between arthropod diversity, but not density, and the age of the beetle bank. This indicated that arthropods respond to increasingly complex plant communities that develop in the beetle banks (there was a positive relationship between plant diversity and arthropod diversity both in beetle banks and field margins; Thomas *et al.* 2001). In contrast, in another study (Pywell *et al.* 2005*a*), there was no effect of habitat age on the overall abundance or diversity of Araneae and Coleoptera in 'newly' sown tussock grass (3–4 years old) and mature field margins (*ca* 50 years old), or in newly planted (2–5 years old) and mature hedge bases (40–60 years old). This highlights the fact that many farmland arthropods have a good dispersal ability, which permits them readily to colonize different habitats regardless of age.

However, a large proportion of ecological restoration studies have been conducted only on newly created habitats in the early stages of succession (less than 3–5 years; Thomas *et al.* 1991; Grey *et al.* 1997; Bowie & Frampton 2004; Waltz & Covington 2004; Bowie *et al.* 2006; Prasad & Snyder 2006). In some cases, only very short-term changes of the order of days to a few weeks are of prime interest, such as in studies of movement (Hossain *et al.* 2002) and predation (Snyder *et al.* 2006), which support the theoretical foundations of ecological restoration. Conversely, Pywell *et al.* (2005*b*) examined only the later stages of succession. In some cases, the age of the habitat being sampled is not stated and is possibly unknown (Goulson *et al.* 2002; Pywell *et al.* 2005*a*).

The importance of the particular time of year in relation to seasonal lifecycles has also been the focus of several studies. Beetle banks provide essential over-wintering habitat for many species of arthropods and birds (Thomas *et al.* 1991; MacLeod *et al.* 2004). Natural and artificial nesting sites and supplementary food are reportedly helpful for breeding birds in the spring, as the density of arthropods (as chick food) tends to be lower in the spring than summer months in beetle banks and grassy field margins (Thomas *et al.* 2001). While Pywell *et al.* (2005*b*) examined only the value of different habitats in providing forage (pollen and nectar) for bumble-bee populations in late summer, they urged that the food requirement in spring and early summer, together with suitable nesting and hibernation habitat, be considered for effective bumble-bee conservation. Similarly, though another study lasted only four weeks during the main period of bumble-bee forage and nest growth in early summer, it was suggested that the nest establishment phase in early spring when the queen bee has to gather sufficient forage to provision her first batch of offspring is a critical time when differences in availability of floral resources between habitats is most vital (Goulson *et al.* 2002). Finally, it may be possible to minimize the effects of adverse husbandry practices, such as soil cultivation, planting and harvesting, on beneficial

arthropods and other animals under conservation by uncoupling their temporal (and spatial) synchrony at the farm scale (Holland *et al.* 2005). For instance, spring-sown cereals provide better quality habitat for farmland birds than do autumn-sown cereals (e.g. winter wheat; Brickle *et al.* 2000).

6. *WHERE* DOES ECOLOGICAL RESTORATION TAKE PLACE?

According to the IUCN Red List of Threatened Species (Baillie *et al.* 2004), there are more threatened species in terrestrial ecosystems (4427 species threatened of the 21 053 assessed; 21%) than freshwater (1388 of the 5574; 25%) or marine (187 species threatened of the 843 assessed; 22%) worldwide (species that live in more than one ecosystem are counted more than once). Although the proportion of threatened species appears to be little different between ecosystems, this is thought to be an artefact of fewer marine and freshwater assessments; preliminary indications suggest that freshwater species may actually be relatively more seriously threatened than the terrestrial species (Baillie *et al.* 2004). The absolute numbers of threatened species are unevenly distributed throughout the eight recognized terrestrial biogeographic realms of the world, with most species (including endemics) occurring in tropical areas in countries such as Australia, Brazil, China, Indonesia and Mexico (Baillie *et al.* 2004). As mentioned earlier, the threatened species that also colonize farmland are not explicitly listed in this or any other 'red-list'. Nevertheless, a review of 68 restoration studies published in *Restoration Ecology* between 1993 and 2003 reported that the majority of studies were carried out in North America (53%), but that there was also relevant work in Australia (19%), Europe (16%), Africa (4%), South America (4%) and Asia (3%) (Ruiz-Jaen & Aide 2005). Within Europe, just six countries had contributed a total of 62 relevant evaluation studies between 1994 and 2000. These countries were: the United Kingdom ($n = 29$ studies), the Netherlands ($n = 18$), Germany ($n = 6$), Switzerland ($n = 5$), Ireland ($n = 3$) and Portugal ($n = 1$; Kleijn & Sutherland 2003). Similarly, a mini review here identified farmland ecological restoration work in Australia, Denmark, Finland, France, New Zealand, United Kingdom and USA (table 1).

In terms of habitats most frequently examined, wetlands were most frequently studied (19%), followed by grassland (16%) and montane forest (13%). Further, ecological restoration was most commonly conducted at sites wherein the previous land uses were mining (37%), agriculture (18%; presumably arable cropping) and pasture (10%; Ruiz-Jaen & Aide 2005). Recent studies on the response of different animals to various habitat types on farmland have considered taxa such as bats (Wickramasinghe *et al.* 2003), farmland birds (Brickle *et al.* 2000) and butterflies (Pywell *et al.* 2004).

Regional effects on biodiversity are also likely to occur within a given country. Pywell *et al.* (2005b) tested the hypothesis that the abundance and diversity of bumble-bees are greater in the more enclosed, mixed farming region of the West Midlands, compared with the open, intensive arable region of East Anglia in the United Kingdom. Despite the significantly higher abundance and plant species richness of dicotyledonous flowers on the lighter soils of the West Midlands, there were no significant differences in the abundance or richness of bumble-bee species recorded on the field margins between the two study areas (Pywell *et al.* 2005b). In Germany, the species richness of spiders (Araneae) was no different between the southern Lower Saxony region that had moderate to high amounts of arable land (25–85%) and the Central Hesse region, which had less arable land (7–61%) (Schmidt *et al.* 2005). Indeed, a regional scale approach is presently being used to 'green' or restore ecologically the entire Waipara vineyard region of New Zealand. Flowering herbs and native shrubs and trees have been planted both within and surrounding the vineyards to restore components of previous ecosystems in a productive landscape and provide habitat (shelter and food resources) for native birds and beneficial arthropods. Critically, this conservation effort brings tangible tourism and marketing benefits. A typical rear label on a bottle from the Canterbury House vineyard in that region is dominated by 'added value' restoration words: 'This wine comes from one of the vineyards taking part in the Greening Waipara project. Native vegetation is being restored for butterfly conservation and other benefits.' (McKinnon 2005b; S.D.W. 2005, personal observation).

Several other studies have urged the development of effective restoration practices at a regional spatial scale (Saunders *et al.* 1993; Hobbs & Norton 1996; Lindenmayer *et al.* 2002; Holland *et al.* 2005; Tscharntke *et al.* 2005; Samways 2007). Indeed, Anderson *et al.* (2002) and Cramer & Hobbs (2005) were highly critical of efforts to manage a certain habitat or a group of habitats within an area at 'human' scales that may in fact be inappropriate for the majority of the species present and the associated ecological processes. In this way, the 'ecological scales' to which various taxa respond and at which key processes occur should be adopted rather than human-perceived catchment or other scales. Hence, just as the negative effects of agricultural intensification can operate at different scales that transcend farm boundaries (discussed above), it is becoming apparent that animals such as bumble-bees and spiders may respond to spatial and temporal changes in resource supply at scales greater than that of a single farm, and that habitat restoration measures may need to be targeted at the regional, rather than the local level (Pywell *et al.* 2005b; Tscharntke *et al.* 2005). Intensive studies of landscape wide patterns of distribution and the tracking of animal movement are helping foster this appreciation of ecological conservation and restoration purposes (Goulson *et al.* 2002; Lavandero *et al.* 2004; Holland *et al.* 2005; Knight *et al.* 2005; Tscharntke *et al.* 2005). Some of this work has specifically examined the movement of animals between patches of arable land and non-crop areas like remnant vegetation (Sutcliffe *et al.* 2003; Silberbauer *et al.* 2004; Summerville *et al.* 2005).

Within a farming unit, the sites that have been the targets of ecological restoration include arable land for fallow or undersowing, areas adjacent to sensitive

Table 2. Details of generic ecological restoration methods cited in the literature

no.	method	reference
1	reduce pesticide and fertilizer use or substitute for less disruptive products in both the main crop (e.g. organics) and adjacent non-target areas like headlands	Pywell *et al.* (2005*b*)
2	increase the size of habitat patches and connectivity between them by creating corridors or, contrary to expectations, establishing small patches of vegetation to facilitate dispersal	Simberloff & Cox (1987); Tscharntke *et al.* (2002) and Steffan-Dewenter & Leschke (2003)
3	increase the availability of non-cultivated land adjacent to fields which provide natural nesting and over-wintering sites (physical shelter and shelter from predation) and primary or alternative food sources, either by deliberately sown non-crop plants (beetle banks, cover strips, floral mixtures and hedgerows), natural regeneration (set-aside land without pesticides or conservation headlands with selective pesticides, uncropped field margins and grassy margins) or repair of existing vegetation	Thomas *et al.* (1991; 2001); Bro *et al.* (2004) and Pywell *et al.* (2004, 2005*b*)
4	increasing the number of fields by reducing the size of each	Holland *et al.* (2005)
5	establish artificial nests or shelter and feeding stations	Bowie & Frampton (2004); Bro *et al.* (2004) and Bowie *et al.* (2006)
6	substitute or diversify the species of arable crops grown at any one time and over the year (arable reversion to pasture, crop rotation, retain over-wintered stubble in a spring fallow, spring not autumn sown cereals such as winter wheat, undersow crops with grasses)	Wakeham-Dawson & Aebischer (1998) and Brickle *et al.* (2000)
7	stagger the timing and location of adverse husbandry practices such as soil cultivation, planting and harvesting (strip harvesting)	Hossain *et al.* (2002) and Holland *et al.* (2005)
8	translocate animals despite the high cost over large areas	Hobbs & Norton (1996) and McKinnon (2005*a*)
9	remove unwanted animals and plants, including fencing to exclude stock from sensitive riparian areas	Byrom (2002); Bro *et al.* (2004); Evans (2004); Macleay (2004) and Waltz & Covington (2004)

urban areas such as schools and roads, farm woodland, field margins including uncultivated areas (set aside), headlands and hedges, grassland, land surrounding aquifers and land water catchments, mid-field strips (beetle banks and cover strips), roadside verges, scrub, waterways and riparian areas (Thomas *et al.* 1991; Fortin & Arnold 1997; Grey *et al.* 1997; Brickle *et al.* 2000; Wickramasinghe *et al.* 2003; Bowie & Frampton 2004; Bro *et al.* 2004; Carvell *et al.* 2004; MacLeod *et al.* 2004; Pywell *et al.* 2004; Pywell *et al.* 2005*a,b*; Bowie *et al.* 2006). The paper by Ryan *et al.* (2002) is a good example of how land attributes such as aspect, elevation, soil depth and slope can be used in farm-scale site selection for farm forestry.

7. *HOW* IS ECOLOGICAL RESTORATION PRACTISED?

This could easily be the largest section of the review if it were intended to provide a detailed description of every type of farmland restoration project. Yet the inherent complexities of ecological restoration and idiosyncrasies of each project means it is unwise to duplicate the same efforts elsewhere, though the principles should be transferable (Clewell *et al.* 2004). In any case, such a detailed synthesis is a task that lies beyond the scope of this review. Nevertheless, we present the generic principles of ecological restoration and draw on specific examples. The Society for Ecological Restoration

International (SERI) produced a document that listed 51 generic guidelines for conceiving, organizing, conducting and assessing ecological restoration projects that would be applicable to any habitat and land use type like farmland (Clewell *et al.* 2000). The guidelines are listed under the following stages: (i) conceptual planning, i.e. identify the project site location and its boundaries (guideline no. 1) and identify restoration goals, if any, pertaining to social and cultural values (no. 5), (ii) preliminary tasks, i.e. document existing project site conditions and describe the taxa (no. 20), prepare a list of objectives designed to achieve restoration goals (no. 27), (iii) installation planning, i.e. describe the interventions that will be implemented to attain each objective (no. 34), (iv) installation tasks, i.e. implement restoration objectives (no. 42), (v) post-installation tasks, e.g. protect the project site against vandals and herbivory (no. 43), and (vi) evaluation, i.e. assess monitoring data to determine if performance standards are being met (no. 48). These guidelines and other generic principles (Samways 2007) have been supported by conceptual frameworks and practical methods for developing ecological restoration programmes and evaluating ecological risk, which are generally not specific to farmland but nevertheless provide a useful starting point for this particular land use type (Hobbs & Norton 1996; Hobbs & Harris 2001; Parkes *et al.* 2003; Oliver 2004; Cramer & Hobbs 2005). It is generally accepted that

practical guidelines for restoration must be based on sound theoretical and empirical foundations, yet be easily integrated into agricultural and horticultural practice (Summerville *et al.* 2005). Yet in this context, Hobbs & Harris (2001) made two salient remarks: firstly, practitioners now desire a stronger ecological foundation for developing and implementing restoration projects; and secondly, it has become apparent that the assumptions underlying many restoration projects are based on outdated concepts of how ecological systems function, such as on the stability of ecosystems and the ability of them to return to equilibrium states following disturbance. Indeed, it appears that some of the earlier methods applied in farmland ecological restoration, which were promoted through agri-environmental schemes (discussed in more detail below), were in fact based on scant ecological empirical data, but now this situation is changing (reviewed by Kleijn & Sutherland 2003).

Before specific restoration options can be applied to a degraded area, it is necessary to consider the identity of the causal agent, the extent of present damage (i.e. condition) and the rate of change (i.e. trend). The causal agents are either biotic or abiotic factors. The appropriate response to biotic factors (e.g. grazing-induced changes in vegetation composition) should be to eliminate or reduce the degrading factor (e.g. grazing animals) and adjust the biotic composition (e.g. replant or allow to regenerate the desired species). Alternatively, the appropriate response to abiotic factors (e.g. through soil erosion or contamination) should be to initially remove the degrading factor and then repair the physical and/or chemical environment. In the latter case, there is little point in manipulating biotic composition if the underlying abiotic problems that impair the expected ecosystem functioning are still present (Hobbs & Harris 2001).

Notwithstanding the above caveat in relation to the causal agent, ecological restoration of farmland has generally involved attempts to moderate the effects of fragmentation and habitat loss and decrease the intensity of management. A list of nine generic ecological restoration methods is shown in table 2. Some of these methods have been supported by good empirical evidence. However, as stated above, the 'ecological' component of ecological restoration was frequently lacking from some earlier programmes such as set-aside, whose foundations were instead based on untested assumptions and anecdotal observations or which restoration filled an objective subsidiary to a greater goal such as to reduce excess agricultural production (Buckingham *et al.* 1999; Kleijn & Sutherland 2003). Expert opinion can be used to prioritize actions in ecological restoration (Cipollini *et al.* 2005). Predictive modelling approaches, including 'alternative futures' analysis, can also be used to test a variety of scenarios and are particularly useful when management planning must balance conflicting land uses (Parkes *et al.* 2003; Stephens *et al.* 2003; Sutcliffe *et al.* 2003; Berger & Bolte 2004; Oliver 2004). Specifically, Stephens *et al.* (2003) concluded that daily ration models were the most useful type of model for simulating the response of farmland birds to changing food supply; the two broad classes of models they considered were phenomenological models (aggregative and population approaches) and behavioural depletion models (daily ration and functional response approaches).

A few recent examples of empirical restoration research designed to support the above farmland ecological restoration methods are provided below. Recent work in New Zealand is using untreated discs of pine wood to accelerate ecological succession; these discs provide many of the ecological functions of natural fallen logs and can harbour late-succession invertebrate communities which usually would not be present in highly modified farming landscapes (Bowie & Frampton 2004). A similar example from New Zealand, which again involves designing end of succession habitats, is the use of 'Weta Hotels'. Wetas (Orthoptera: Anostostomatidae and Rhaphidophoridae) are large, iconic native insects that are usually associated with undisturbed forest landscapes. Weta hotels, artificial shelters which mimic coarse woody debris with cavities, harbour this specialist fauna on farmland on which this insect disappeared in association with forest clearance (Bowie *et al.* 2006). Populations of pollinating bees and predatory wasps in agroecosystems can also be enhanced with the introduction of suitable nesting sites (Gathmann *et al.* 1994; Tscharntke *et al.* 1998; Barron *et al.* 2000).

The provision of non-crop vegetation as a food source for immature and/or adult insect natural enemies and birds requires a good understanding of food-web theory. The 'shotgun' approach of planting a species-rich wildflower/wildlife seed mixture (Carreck & Williams 2002; Pfiffner & Wyss 2004) or by allowing natural regeneration of species from the seed bank (Pywell *et al.* 2005b) may result in inadvertent proliferation (and possible dominance) of a single or few species in the target area (MacLeod *et al.* 2004; Pywell *et al.* 2005b). Ill-conceived selection of flowering plants could unintentionally enhance the incidence of herbivorous pests, higher-order predators/hyperparasitoids or plant diseases (Stephens *et al.* 1998). This could lead to disillusionment among agricultural and horticultural practitioners with habitat restoration methods. In order for these methods to be effective, their implementation must be guided by empirical and theoretical research, such as the initial screening of individual tussock grass (Thomas *et al.* 1991; MacLeod *et al.* 2004) or flower species alone (Baggen *et al.* 1999; Lavandero *et al.* 2006) and then in mixtures (Pontin *et al.* 2006; i.e. a directed approach *sensu* Gurr *et al.* 2004).

The monitoring of restoration projects to judge their success generally involves consideration of three major ecosystem attributes: (i) diversity (species richness, evenness and abundance), (ii) vegetation structure (cover, density or biomass), and (iii) ecological processes (e.g. nutrient cycling, rates of herbivory and predation; Ruiz-Jaen & Aide 2005). Importantly, these three attributes are not only evaluated in the restored site, but also in unrestored ('impacted') and intact ('unimpacted') reference sites to evaluate the level of restoration success against the nominal objectives. Further, the restoration 'treatments' should be replicated across different sites. The SERI has produced a 'primer' that provides a list of nine ecosystem attributes as a guideline for measuring restoration success (SERI 2004). They suggested that a
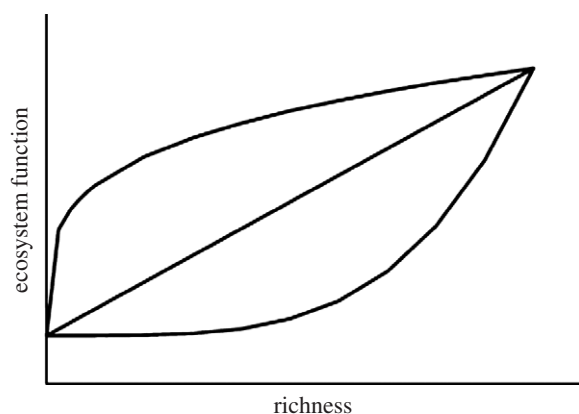
Figure 2. Theoretical relationships between species richness and ecosystem function. The top line shows saturation in which the majority of species contribute little, middle line shows a linear relationship in which each additional species adds a consistent unit of function and bottom line shows an accelerating function whereby interactions between species enhance their efficiencies. The latter is least well supported by empirical data. Adapted from Kremen (2005).

restored ecosystem should have the following attributes: (i) similar diversity and community structure in comparison with reference sites, (ii) presence of indigenous species, (iii) presence of functional groups necessary for long-term stability, (iv) capacity of the physical environment to sustain reproducing populations, (v) normal functioning, (vi) integration with the landscape, (vii) elimination or reduction of potential threats, (viii) resilience to natural disturbances, and (ix) self-sustaining. Modelling approaches can be used to evaluate restoration success (Stephens *et al.* 2003; Anand & Desrochers 2004). Thus, ecological restoration of farmland involves adapting guidelines, applying methods and assessing the success against the objectives, and if possible, reference sites.

(a) *Agri-environmental schemes*
Public concern over the environmental impacts of agriculture has been a significant driver for the introduction of various types of agri-environmental scheme. A defining characteristic is that they provide compensation to farmers for any loss of income that results from their implementation of prescribed measures designed to benefit the environment: 'payment for habitat and wildlife gain' (Sherrott 2001).

The first agri-environmental scheme was introduced in the United Kingdom in 1986 (Dobbs & Pretty 2004) and a series of schemes has since been introduced: the Environmentally Sensitive Area Scheme, the Countryside Stewardship Scheme, the Arable Stewardship Scheme (Sherrott 2001) and most recently the England Rural Development Program (DEFRA 2005). Support from the European Union (EU) Common Agricultural Policy (CAP) has led to agri-environmental schemes becoming popular within and outside the EU with schemes in place in at least 26 of the 44 European countries (Kleijn & Sutherland 2003) including Belgium (Dupraz *et al.* 2003), Finland (Yliskyla-Peuralahti 2003), Scotland (Egdell 2000) and Sweden (Ottvall & Smith 2006). The Conservation Reserve Program occupies a similar place in the USA (Lovell & Sullivan 2006).

(b) *Ecological background*
Quantitative estimates available of the economic value of biodiversity (Costanza *et al.* 1997; Pimentel *et al.* 1997; Sandhu *et al.* in press) are supported by ecological theory (Kremen 2005). Though the science of valuing and mechanistically understanding ecosystem services is in its infancy, empirical studies tend to support the notion that increasing biodiversity favours ecosystem functions of value such as biological control of crop pests, pollination and nutrient cycling. However, the ecosystem function–biodiversity debate is highly topical among ecologists and the circumstances under which ecosystem function actually is enhanced can be complex, and sometimes counter-intuitive (Finke & Denno 2004). Typically, the response function to biodiversity increase takes the form of a linear or saturation curve (figure 2; Kremen 2005). Though the addition of a single species to an already complex ecosystem is likely to have only a small effect (the saturation curve), this is less likely to apply in simple agroecosystems than in more complex natural systems. The accelerating form of response (figure 2) is not well supported by empirical data and this has important practical implications (Schwartz *et al.* 2000). If the response were of this type then harnessing the ecosystem benefits of biodiversity would be far less tractable owing to the high numbers of species and microhabitats, as well as land area and genetic diversity that would need to be conserved or reinstated. Evidence that relatively large increases in ecosystem function may be achieved with low or modest levels of biodiversity enhancement supports the potential use of agri-environmental schemes for the provision of ecosystem services as well as for altruistic reasons (e.g. conservation of endangered but not directly useful species) and/or political reasons (e.g. reducing agricultural overproduction). Indeed, using agri-environmental schemes to promote numerous ecosystem services from restored farmland is encapsulated in the blueprint proposal for a 'multifunctional' countryside (Sutherland 2004).

An increasingly important aspect of biodiversity in agricultural systems is that many species are exotic, especially in Australasia. Quite rightly, a great deal of attention has been given to the negative effects of such introduced species on endemics but relatively little attention has been given to the potentially positive effects on ecosystem services. Ecological fitting (Janzen 1985) occurs when a species expands its geographical range and enters into novel ecological associations; for example, hummingbird use of exotic plants (Gill 1987). The most remarkable example is the development of an apparently fully functional cloud rainforest of exotic plant species on the mid-Atlantic island of Ascension, a location reported to be destitute of trees when Darwin visited in 1836 (Wilkinson 2004). Ecological fitting is apparent also in agricultural systems in cases where exotic plants are recognized and used by native natural enemies as in the case of the Australian endemic egg parasitoid *Trichogramma carverae* using the Mediterranean plant *Lobularia maritima* (L.) Desv. (sweet alyssum) as a nectar source with consequent increases in their parasitism of vineyard pests (Begum *et al.* 2006). Other similar

examples are known from the conservation biological control literature (Landis *et al.* 2000) suggesting that closer examination may reveal cases of services other than biological control involving ecological fitting. This concept further supports the value of biodiversity being enhanced within agri-environmental schemes despite the apparently unavoidable presence of exotics.

(c) Costs and benefits

In the EU where agri-environmental schemes are most widely used, only a small fraction of total government expenditure on the CAP was historically allocated to the support of the schemes—4% in 1999 (Wilson & Hart 2001). Of course this is a small part of a large overall budget; accordingly, total expenditure on EU agri-environmental schemes since 1994 has been estimated at €24.3 billion (Kleijn & Sutherland 2003). Recently, the EU's new Rural Development Regulation made possible the allocation of up to 20% of CAP funding to agri-environmental and rural development schemes (Dobbs & Pretty 2004). Such levels of expenditure have made farmers in some countries like Austria heavily dependent on this funding (Schmitzberger *et al.* 2005). The conditions placed upon farmers in order to qualify for payments may be rigorous. In Switzerland, it is a requirement that at least 7% of every farm is allocated to ecological compensation areas (Jeannerett *et al.* 2003).

Though there have been many attempts to measure the effectiveness of agri-environmental schemes, Kleijn & Sutherland (2003) concluded that the research design was inadequate in the majority and almost one-third did not include a statistical analysis of data. The most common fault was that most studies compared the areas subject to agri-environmental schemes with control areas, despite the fact that areas chosen for schemes often were superior from the outset, that being part of the basis for their protection and enhancement. Some studies avoided this problem by collecting baseline data and comparing trends or changes or more careful selection of comparison sites.

In a meta-analysis of agri-environmental schemes, evaluations that covered birds, arthropods and plants, just over half of all taxa exhibited increases in richness or abundance and only 6% declined in comparison with control areas (Kleijn & Sutherland 2003). Though these overall statistics appear encouraging, there is a degree of dissatisfaction that agri-environmental schemes are failing to achieve their conservation aims. In Ireland, there appeared to be a need for better farmer education and training (Aughney & Gormally 2002) and in the Netherlands the high overall intensity of land use was thought to constrain improvements (Kleijn *et al.* 2004). Other studies, however, show encouraging signs of reversal of long established trends in wildlife decline in the UK (Swetnam *et al.* 2004) or increases in Swiss biodiversity (Knop *et al.* 2005).

(d) Improving agri-environmental schemes

Given the criticism directed to the mixed success of agri-environmental schemes and of the methods used to evaluate these impacts (Kleijn & Sutherland 2003; Kleijn *et al.* 2006), there is a need to improve not only our understanding of the indicators to be used in assessment of schemes (Onate *et al.* 2000) but also to involve ecologists to a greater extent in the development and measurement of schemes (Ormerod *et al.* 2003). An example of how theory may support the design of schemes is provided by the 'habitat creation model' to identify those vegetation communities that can be established on ex-arable land (Gilbert *et al.* 2000). A further ecological consideration is the scale over which processes occur. Increasingly, there is empirical evidence that landscape-level factors affect ecosystem service providers such as pollinators (Tscharntke *et al.* 2005) and natural enemies (Thies *et al.* 2003). Such effects may be especially important at large temporal scales, i.e. effects over more than one cropping phase (Schmidt *et al.* 2005). It is therefore critical that agri-environmental schemes address the importance of landscape-scale effects (Roschewitz *et al.* 2005), including the relationship between local and regional diversity (Kleijn *et al.* 2006). In practical terms, this could take the form of better orchestration above the level of individual farms to ensure that the implementation of agri-environmental schemes within a landscape is optimal in relation to each other and the existence of other significant features like woodland. Doing this will, however, demand better knowledge of how ecosystem service providers respond to habitat fragmentation (and measures for its amelioration such as wildlife corridors) and habitat loss (and measures for its amelioration such as taking parcels of land out of production for afforestation and other habitat). This may not be as simple as it may appear first. In the case of natural enemies of pests, for example, research has shown that beneficial species may respond at spatial scales different from those applying to the pests (Thies *et al.* 2005). Though this finding signals tantalizing scope to 'engineer' landscapes to favour beneficials over pests, important differences may occur even within beneficial groups. For example, parasitoids responded to fragmentation of forest elements at differing scales, an effect apparently linked to the body size of the insects (Roland & Taylor 1997).

An emerging trend for agri-environmental schemes is 'multifunctionality' (Dobbs & Pretty 2004; Sutherland 2004) rather than narrowly defined environmental objectives. Accordingly, the multiple objectives must be reflected in a multidisciplinary approach to evaluation as advocated by Carey *et al.* (2003). An important consideration for future schemes is to shift financial support from agricultural production to 'stewardship' activities that will achieve desired environmental outcomes (Dobbs & Pretty 2004), a 'post-productionist' paradigm in the terms of Wilson (2004). There is little research available on how production and stewardship imperatives may be balanced in countries outside the EU. In Australia, for example, there has been a history of very little State or Commonwealth production support. Accordingly, the rather limited agri-environmental scheme-type measures that have been used, the 'Landcare' movement for example, have been largely independent of production incentives (Abensperg-Traun *et al.* 2004).

Farmer attitudes towards agri-environmental schemes clearly play a major role in uptake, the extent to which guidelines are followed and the degree to which outcomes

are achieved. A wide spectrum of degree of commitment and sympathy was evident in one survey of British farmers and these fell into two approximately equal sized categories (Morris & Potter 1995). 'Passive adopters' were driven primarily by financial rewards and appeared adept at maximizing payments while minimizing effort or impact on normal farming activities by, for example, 'resting' poor quality arable land. One interviewee commented '…Basically it is attractive because you are getting paid for what you are already doing'. Fortunately, 49% of subjects were 'active adopters' with an appreciation of the objectives of agri-environmental schemes and willing to make changes to pursue environmental objectives. Attracting farmers with such attitudes into schemes was considered an important indicator of success (Wilson & Hart 2001) but the fact that participants in schemes are self-selecting reinforces the need for careful design of schemes in terms of eligibility requirements and implementation and compliance regulations. A related problem associated with agri-environmental schemes is the confusion caused by multiple schemes in a given jurisdiction and the sheer complexity of the regulations associated with each. This factor led Falconer (2000) to propose a 'one-stop shop' for farmers to access information on the schemes for which they may be eligible.

Issues such as those dealt with above illustrate the need for social, as well as ecological, science inputs for the design, implementation and evaluation of agri-environmental schemes. Success is influenced by human as well as technical factors.

8. CONCLUSIONS

It is clear that more sustainable agricultural practices in conjunction with ecological restoration methods on farmland can contribute to reduce the unacceptably high external costs of agriculture (Costanza *et al.* 1997; Pretty *et al.* 2000). A wide range of bird, plant and insect species that are primarily rare or threatened with extinction have been targeted in ecological restoration (Kleijn & Sutherland 2003). However, there is no worldwide compendium of species that have been identified explicitly in farmland ecological restoration contexts. The process of ecological restoration can be long-lasting and determinants of endpoints can be difficult to ascertain (Ormerod 2003; Ruiz-Jaen & Aide 2005). Nonetheless, the synchrony of restoration efforts with plant and animal lifecycles has been highlighted (Goulson *et al.* 2002; Pywell *et al.* 2005*b*).

The SERI has produced generic guidelines for conceiving, organizing, conducting and assessing ecological restoration projects (SERI 2004). Additionally, there are now good conceptual frameworks and practical methods for developing ecological restoration programmes that are based on sound ecological principles and supported by empirical evidence and modelling approaches (Hobbs & Norton 1996; Hobbs & Harris 2001; Parkes *et al.* 2003; Stephens *et al.* 2003). Restoration methods must also be technically achievable and socially acceptable and extend over a range of locations (Lindenmayer *et al.* 2002). A directed approach for managing plant biodiversity is preferred to a shotgun approach that could inadvertently pose negative side effects (Gurr *et al.* 2004). The monitoring of restoration projects should include the restored sites as well as unrestored and intact reference sites, to evaluate, albeit problematic, the level of restoration success against the nominal objectives and accepted attributes of a restored ecosystem (Ruiz-Jaen & Aide 2005). In this context, it is important to reconcile differences between those methods that favour conservation and those that favour economic returns, to ensure that conservation efforts are mutually beneficial for the landowners and for biodiversity (Banks 2004). The multifunctionality direction of agri-environmental schemes referred to above may be seen as a response to this need. Much work remains to be done to define and test the effectiveness of these agri-environmental schemes for various taxa and ecosystem services (Kleijn & Sutherland 2003; Pywell *et al.* 2005*b*). Further, the conservation status and ecology of many taxa remain poorly studied (Baillie *et al.* 2004) making it challenging to identify both focal sites and landscapes within which restoration efforts are most likely to succeed (Summerville *et al.* 2005) and accurately gauge restoration success (Purcell *et al.* 2004; Ruiz-Jaen & Aide 2005). The potential for ecological restoration of farmland to improve the sustainability of agriculture production is clearly high but there is still much to learn as the field of restoration ecology develops over the course of this century.

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