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Cost–benefit analysis of ecological networks assessed through spatial analysis of ecosystem services

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Summary

1. The development of ecological networks could enhance the ability of species to disperse across fragmented landscapes and could mitigate against the negative impacts of climate change. The development of such networks will require widespread ecological restoration at the landscape scale, which is likely to be costly. However, little information is available regarding the cost-effectiveness of restoration approaches.

2. We address this knowledge gap by examining the potential impact of landscape-scale habitat restoration on the value of multiple ecosystem services across the catchment of the River Frome in Dorset, England. This was achieved by mapping the market value of four ecosystem services (carbon storage, crops, livestock and timber) under three different restoration scenarios, estimating restoration costs, and calculating net benefits.

3. The non-market value of additional services (cultural, aesthetic and recreational value) was elicited from local stakeholders using an online survey tool. Flood risk was assessed using a scoring approach. Spatial Multi-Criteria Analysis (MCA) was conducted, incorporating both market and non-market values, to evaluate the relative benefits of restoration scenarios. These were compared with impacts of restoration on biodiversity value.

4. Multi-Criteria Analysis results consistently ranked restoration scenarios above a non-restoration comparator, reflecting the increased provision of multiple ecosystem services. Restoration scenarios also provided benefits to biodiversity, in terms of increased species richness and habitat connectivity. However, restoration costs consistently exceeded the market value of ecosystem services.

5. *Synthesis and applications*. Establishment of ecological networks through ecological restoration is unlikely to deliver net economic benefits in landscapes dominated by agricultural land use. This reflects the high costs of ecological restoration in such landscapes. The cost-effectiveness of ecological networks will depend on how the benefits provided to people are valued, and on how the value of non-market benefits are weighted against the costs of reduced agricultural and timber production. Future plans for ecological restoration should incorporate local stakeholder values, to ensure that benefits to people are maximised.

Key-words: biodiversity, climate change, ecological restoration, ecosystem benefit, habitat connectivity

Introduction

Many countries have incorporated the concept of ecological networks into national environmental policies (Jongman & Pungetti 2004). For example in Europe, some 54 countries have formally endorsed strategies for a Pan-European Ecological Network (PEEN) (Jones-Walters 2007). At least 42 ecological network initiatives have been established across Europe, including seven at the national scale, with many other actions undertaken at more local scales (Boitani *et al.* 2007). Ecological networks may be broadly defined as networks of areas that are connected to enhance biodiversity conservation, typically through the establishment of corridors and buffer zones to facilitate the dispersal and migration of species

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(Boitani *et al.* 2007). According to Opdam, Steingröver & van Rooij (2006), an ecological network is a multi-species concept, linking ecosystems, and based on a consideration of ecological processes. The concept is founded on the principles of landscape ecology, metapopulation theory and metacommunity dynamics, which emphasise the importance of connectivity among patches of habitat to ensure the viability of both populations and communities of species (Jongman & Pungetti 2004; Boitani *et al.* 2007). The growth in interest in ecological networks reflects increasing concern regarding the reduction and fragmentation of natural and semi-natural habitats, which are recognised as major causes of biodiversity loss (Fahrig 2003).

In the UK, Lawton *et al.* (2010) recently examined whether England's wildlife sites comprise a 'coherent and resilient' ecological network. These authors recommended that species and habitats should be restored 'to levels that are sustainable in a changing climate' and highlighted the need for ecological restoration to be undertaken throughout the country to develop ecological networks. The incorporation of large-scale ecological restoration into land use policy would represent a new initiative for the UK, but is consistent with the recent restoration targets introduced by the Convention on Biological Diversity (Bullock *et al.* 2011).

The importance of ecological restoration for countering biodiversity loss is widely recognised (Bullock *et al.* 2011). However, the costs of ecological restoration can be substantial. Typical restoration costs range from 100s to 1000s US$ per ha, but values vary markedly with ecosystem type, the extent of degradation and the restoration methods used (TEEB 2009). Such high costs raise the question of whether ecological restoration actions are likely to be cost-effective (Bullock *et al.* 2011). Very few attempts have been made to conduct a cost-benefit analysis of restoration initiatives. In a review of over 2000 restoration case studies, TEEB (2009) found that $<5\%$ provided meaningful cost data, and of those that did, none provided detailed analysis of the achieved or projected benefits.

In the context of developing ecological networks in England, Lawton *et al.* (2010) suggested that the value of the ecosystem services provided will often far outweigh any costs incurred. However, this hypothesis remains untested. The aim of this investigation was therefore to examine both the costs and benefits of developing an ecological network through habitat restoration. Potentially, the cost-effectiveness of such interventions can be evaluated through the spatial analysis of ecosystem services. Balmford *et al.* (2011) have developed a conceptual framework that focuses on quantifying the costs and benefits associated with changes in ecosystem services as a result of a policy action, through comparison of counterfactual scenarios. This approach is consistent with a number of other studies that have emphasised the importance of comparing alternative policy actions rather than a static analysis of current service provision, for robust estimates of both values and costs (Fisher *et al.* 2008; Nelson *et al.* 2009). The approach is spatially explicit, reflecting the fact that both the production and value of ecosystem services vary spatially, and land management decisions are typically spatially oriented (Nelson *et al.* 2009; de Groot *et al.* 2010; Balmford *et al.* 2011).

Although progress has been made in mapping the value of ecosystem services (Nelson *et al.* 2009; Kareiva *et al.* 2011), we are aware of only one previous study that has applied the spatial analysis of ecosystem services to evaluate the cost-effectiveness of ecological restoration, namely that conducted by Birch *et al.* (2010) in dryland areas of Latin America. This investigation mapped the values of five ecosystem services together with restoration costs, to examine the costs and benefits of restoring native forest communities. The results indicated that cost-effectiveness was dependent on the restoration methods used and on the specific location. Whereas passive restoration approaches employing natural regeneration were cost-effective for all study areas, the benefits from active restoration were generally outweighed by the relatively high costs involved (Birch *et al.* 2010).

As noted by de Groot *et al.* (2010), few landscape-scale assessments are available of the provision and value of multiple ecosystem services under alternative management regimes, despite their importance for informing policy. Here, we first present estimates of the monetary value of four ecosystem services (carbon storage and production of crops, livestock and timber) under three different habitat restoration scenarios, for the catchment of the River Frome, Dorset, UK. These scenarios were based on plans for large-scale habitat restoration that aim to mitigate against the negative impact of climate change on biodiversity. We then examine the marginal changes in the monetary value of both benefits and costs by comparing restoration scenarios with a 'pre-project' baseline, enabling a cost–benefit analysis to be performed. As noted by Rouquette *et al.* (2009), cost–benefit analysis should not be based purely on monetary values, but should ideally also consider the value of ecosystem services that are non-market public goods. We therefore incorporated local stakeholder values of three additional ecosystem services, namely recreational, aesthetic and cultural values. In addition, the impact of restoration scenarios on flood risk mitigation was examined using a scoring approach.

Materials and methods

SCENARIO DEVELOPMENT

This investigation was undertaken at the scale of a single river catchment, that of the River Frome, Dorset, southern England (Appendix S1, Supporting Information). A map of the current land cover of the catchment was derived from the UK Land Cover Map 2000 (LCM2000) (Fuller *et al.* 2002). This was used as the basis of a 'pre-project' baseline scenario (labelled PP), effectively representing no future land cover change. To explore the potential benefits and costs of establishing ecological networks, three habitat restoration scenarios were then developed by modifying this land cover map. These scenarios were based on a regional approach to landscape-scale planning for habitat restoration that has recently been developed by conservation organisations in South West England (Brenman 2005;

The Wildlife Trusts 2006). Proposed networks are referred to as strategic nature areas (SNAs), which have been identified in a map ('Nature Map', http://www.biodiversitysouthwest.org.uk/nm_dwd.html) designed to inform habitat restoration strategies to create functional habitat networks (Brenman 2005). It was assumed that each of the three restoration scenarios would be fully implemented by the year 2060 (i.e. a timeline of 50 years), with carbon build-up times of 50 years to account for woodland maturation, following Cantarello, Newton & Hill (2011).

To develop the restoration scenarios, the LCM2000 map (PP) was modified according to the SNAs illustrated in the 'Nature Map'. Three scenarios were developed (Appendix S1, Supporting information): (i) the priority habitat constituting 30% of the area of each SNA (LS 30), (ii) the priority habitat constituting 60% of the area of each SNA (LS 60), and (iii) the priority habitat constituting a combination of 30% and 60% based on the targets described by Brenman (2005, p. 43) (i.e. broad-leaved/mixed woodland, 30%; fen, marsh, swamp, 60%; neutral grassland, 30%; calcareous grassland, 60%; and dwarf shrub heath, 60%) (LS 30–60).

ASSESSMENT OF ECOSYSTEM SERVICES

An assessment was made of the economic value of four ecosystem services, namely arable crop production, livestock production, carbon storage and timber production. In addition, the non-market value of four ecosystem services was assessed, namely flood risk mitigation, aesthetic, recreational and cultural value. Maps were produced for each ecosystem service and habitat restoration scenario by estimating values according to land cover type (for details of methods see Appendix S1, Supporting Information).

ASSESSMENT OF COSTS

In the current analyses, production costs were subtracted from the estimated values of economic benefits (i.e. crops, livestock, timber); these values are therefore net. Rayment (2006) was used as a source of generalised cost estimates for restoration of each habitat type. These estimates include the capital cost of habitat establishment and a maintenance cost per hectare. Annual maintenance costs were applied over two time-scales (Option A: 50 years and Option B: 10 years), to explore the sensitivity of the results to variation in this variable. These estimates also include the opportunity costs of habitat restoration as illustrated by agri-environment scheme (AES) compensation payments.

Marginal costs were estimated by taking account of current income from AES and land purchase agreements in the PP scenario. Spatial data showing the location of current AES were obtained from Natural England (http://www.naturalengland.org.uk/, downloaded 6 September 2010) and clipped to fit the study area boundary. This provided information on costs associated with each land cover type, which were incorporated in the analysis. The total cost of AES in the study area for the current scheme (which spans 5–15 years depending on the scheme type) is £18 160 200, covering 37 170 ha, which represents the PP value. This gives a mean annual cost of £2 095 962 for the catchment, or £56·39 ha^{-1} year^{-1}. Estimates of marginal costs were produced by subtracting the PP values from those projected according to the restoration scenarios, which were calculated using the values presented by Rayment (2006). Costs were calculated based on the assumption that current AES income does not contribute to the projected increases in habitat area under the restoration scenarios.

ANALYSIS OF NET BENEFITS

For each of the restoration scenarios (LS 30, LS 60, LS 30–60), the marginal changes in the value of benefits (MVB) and costs (MVC) were estimated by subtracting the total value of each benefit in the PP scenario from total value of each benefit in each of the restoration scenarios. The net present value (NPV) was calculated as the difference between MVB and MVC for each scenario under the two different cost options (A and B). The net social benefit (NSB) was calculated as the present total value of benefits minus current costs, providing an indication of current stocks. NSB was calculated for each of the three restoration scenarios as the PP value plus NPV under the two different cost options (Options A and B). NSB therefore represents the summed change in value of ecosystem services between the PP and restoration scenarios, minus the costs of each scenario.

A cost–benefit analysis of each restoration scenario was performed by calculating the benefit:cost ratio (BCR) for a range of discount rates (0–10%), and for the two cost options (A and B). The BCR was calculated by dividing MVB by MVC. A scenario is cost-effective if BCR > 1.

DISCOUNTING

Discount rates are widely used in economic analyses to assess the present value of future benefits based on assumptions such as positive rates of inflation, continual economic growth, and time preference. Typically, the value of any future amount of money is discounted at a chosen rate to estimate current NPV. Here, a range of discount rates was applied to benefits and costs, to explore the sensitivity of research findings to this variable. We used the following equations to calculate the net present value from the summed marginal values of each ecosystem service benefit (MVB) minus the costs (MVC) for $t = 0$ to $T = 50$ (the number of years considered for the restoration scenarios):

$$\mathrm{MVB} = \sum_{t=0}^{T} (Ft)/(1+r)^{t}$$

$$\mathrm{MVC} = \sum_{t=0}^{T} (Ct)/(1+r)^{t}$$

$$\mathrm{NPV} = \sum_{t=0}^{T} (\mathrm{MVB}) - (\mathrm{MVC})$$

where F_t is marginal ecosystem service flow (F) in £ sterling at time t and r is the discount rate.

All benefits with an economic value were discounted at a declining rate (3·5% for years 1–30, 3% thereafter, to a total of 50 years). This follows UK Government advice (Treasury 2003) regarding discounting over this time-scale. In addition, benefits were discounted at 0, 1, 3·5, 7 and 10% over 50 years, for sensitivity analysis.

BIODIVERSITY VALUE

Two approaches were used to examine the potential impacts of habitat restoration on biodiversity, involving calculation of: (i) a species richness index and (ii) a measure of habitat connectivity for species of conservation concern. Methods are described in Appendix S1 (Supporting information).

MULTI-CRITERIA ANALYSIS

Multi-Criteria Analysis (MCA) was used to explore the relative effectiveness of the different scenarios in providing ecosystem benefits, enabling both economic and non-economic values to be incorporated in the same analysis (Appendix S1, Supporting Information). A 10-m grid cell raster map was generated for each criterion (ecosystem service) for each of the scenarios, and all criterion maps were combined in a spatial MCA performed in ILWIS 3.6 (© 2009, ITC, University of Twente, the Netherlands), using a weighted-sum method. The MCA was repeated as a non-spatial analysis using DEFINITE 3.1.1.7 (Institute for Environmental Studies, Amsterdam, the Netherlands) to permit further sensitivity analysis.

To assess the influence of different criterion weights on the different restoration alternatives, three cases were explored involving application of different weights to each criterion: (i) *equal weighting*, where each criterion was weighted equally; (ii) *stakeholder weighting*, where each criterion was weighted according to the mean score assigned to each ecosystem by stakeholders in a workshop (see Appendices S1 and S2, Supporting information); and (iii) *economic value weighting*, where each criterion was weighted according to the to their current economic value; the four non-market ecosystem services were each accorded a zero weight (see Appendix S3, Supporting information). To identify the preferred scenario for the provision of ecosystem services, the scenarios were ranked based on the results of the MCA. These results were then compared with ranking of the scenarios in terms of the two measures of biodiversity value that were employed (species richness index and habitat connectivity).

To examine the potential trade-offs between biodiversity and ecosystem services, and between different ecosystem services, a Spearman rank multiple correlation was performed on the normalised ecosystem service values for each land cover type that were used as input to the MCA, using SPSS 16.0 for Windows (1989–2007, SPSS Inc., Chicago, Illinois, USA) (see Appendix S4, Supporting Information).

Results

VALUE OF ECOSYSTEM SERVICES

Currently, the landscape of the Frome catchment is dominated by agricultural land use, with arable land accounting for 42·1% of land cover and improved grassland accounting for a further 32·3%. While a further 14 land cover types are differentiated on LCM2000, each of these accounted for < 8% of total land cover, with broad-leaved woodland (7·93%) being the most extensive type after agricultural land.

Each of the restoration scenarios resulted in changes to all of the land cover types included within the scenarios (Appendix S5, Supporting Information). The land cover type that increased most in terms of total area was calcareous grassland, which increased by more than a factor of two in LS 30, and by more than a factor of five in the other restoration scenarios. Other land cover types that increased in all restoration scenarios were dwarf shrub heath, fen/marsh/swamp and neutral grassland. Broad-leaved woodland displayed contrasting trends in the different scenarios, decreasing in both LS 30 and LS 30–60, but increasing in LS 60. The expansion in area of these habitats was associated with losses of acid grassland, coniferous woodland, improved grassland and arable land cover types, which consistently declined in all restoration scenarios (Appendix S5, Supporting information).

Estimates of the monetary value of current stocks (Table 1) suggest that the total value of carbon is substantially higher than that of the other services considered here. This partly reflects the fact that carbon storage is associated with all land cover types, whereas production of other services is limited to only a subset of land cover types. These results also highlight the sensitivity of value estimates to the price of carbon that is used, total value differing by more than a factor of two between low and high carbon prices. Regardless of the price adopted, the total value of stored carbon was projected to increase in all scenarios, by up to 8% in LS 60. In contrast, timber value declined markedly in each of the restoration scenarios, by more than 40% in LS 30 and LS 30–60 (Table 1). This primarily reflects the conversion of conifer plantations to other land cover types. Crop and livestock value similarly declined in each restoration scenario, associated with the conversion of agricultural land, with greatest losses recorded in LS 60 (Table 1).

The contrasting responses of different ecosystem services under the three restoration scenarios are similarly reflected in the marginal values (Table 2), which represent the changes resulting from restoration actions. While marginal carbon values were consistently higher than the values of other benefits, these differences are less pronounced than for estimates of total stocks. In general, habitat restoration was associated with an increase in marginal carbon value; estimates for LS 60 were more than double those for LS 30. In contrast, all of the other services were characterised by a decline in marginal value with increasing land cover conversion, reflecting the loss of agricultural land and conifer plantations (Table 2). Declines in the value of livestock production were slightly larger than those associated with crop production, but both were most pronounced in the LS 60 scenario. Declines in the value of timber were consistently greater than the declines in crop and livestock production. Overall, at zero discount rate, these declines were more than compensated for by the increase in carbon value, even when the lowest carbon price was used. However, this finding was sensitive to discount rate (Table 2).

Restoration scenarios were also associated with increases in the value of non-market ecosystem services. Under all three restoration scenarios, an increase in the value of flood risk mitigation was evident throughout the catchment, particularly in western areas (Fig. 1). The other non-market services similarly displayed spatial heterogeneity across the study area, with areas of higher value tending to be concentrated in eastern parts of the catchment. Areas of relatively high value increased for all three services in each of the restoration scenarios, but in a more restricted manner than was evident for flood risk mitigation. Increases were primarily restricted to the eastern part of the catchment, with only localised areas increasing in western areas (Fig. 1).

The three restoration scenarios differed markedly in cost, as expected given the contrasting areas of land undergoing conversion. The LS 60 scenario consistently incurred substantially higher costs than the LS 30 scenario, with a more than fivefold difference recorded under Option B and a more than twofold

Table 1. Present value of benefits (PVB) represent monetary values of ecosystem service stocks using (i) 0% and (ii) 3·5% declining discount rates over 50 years (t = 50)

	Present value of benefits (PVB) (£)			
Ecosystem services	PP	LS 30	LS 30–60	LS 60
Carbon (low)	219 549 352	226 913 036	231 436 156	237 389 454
Carbon (mid)	*402 507 146*	*416 007 232*	*424 299 619*	*435 213 999*
Carbon (high)	*493 986 043*	*510 554 330*	*520 731 351*	*534 126 271*
Timber	18 936 361	16 262 663	10 767 134	11 288 738
Crops	10 046 545	8 908 980	7 488 206	7 217 134
Livestock	9 061 469	7 560 639	6 131 706	5 681 442
Combined PVB (zero discount rate)	257 593 727	259 645 318	255 823 202	261 576 768
Combined PVB (3·5% declining discount rate)	257 593 727	258 565 876	256 754 763	259 523 817

PP: pre-project baseline; LS 30: habitat restoration scenario with 30% target; LS 30–60: restoration scenario with combined 30% and 60% targets; LS 60: restoration scenario with 60% target (see text). PP represents the current value of stocks at the present day (t = 0) and so discounting is not applicable. The combined PVB of services uses the low traded carbon value for all scenarios (see text). Alternative (mid and high) carbon values are also presented in italics (see Appendix S1 for details). The italicised numbers therefore do not contribute to the total values presented.

Table 2. Marginal value (£) of economically valued ecosystem benefits (MVB) at different discount rates

		Discount rate (%)				
Scenario	Ecosystem service	0	1	$3{\cdot}5^{D}$	7	10
LS 30	Carbon (low)	7 363 683	5 772 556	3 489 291	2 032 487	1 460 191
	Carbon (mid)	*13 500 086*	*10 583 019*	*6 397 034*	*3 726 225*	*2 677 017*
	Carbon (high)	*16 568 288*	*12 988 251*	*7 850 906*	*4 573 095*	*3 285 430*
	Crops	−1 137 565	−891 763	−539 037	−313 985	−225 575
	Livestock	−1 500 830	−1 176 534	−711 170	−414 251	−297 609
	Timber	−2 673 698	−2 095 971	−1 266 935	−737 980	−530 184
	Total MVB	1 958 550	1 608 288	972 149	566 271	406 823
LS 30–60	Carbon (low)	11 886 804	9 318 331	5 632 578	3 280 935	2 357 109
	Carbon (mid)	*21 792 473*	*17 083 607*	*10 326 393*	*6 015 048*	*4 321 367*
	Carbon (high)	*26 745 308*	*20 966 245*	*12 673 301*	*7 382 104*	*5 303 495*
	Crops	−2 558 339	−2 218 039	−1 212 272	−780 960	−561 062
	Livestock	−2 929 763	−2 296 706	−1 388 272	−808 658	−580 961
	Timber	−8 169 227	−6 404 040	−3 871 000	−2 254 829	−1 619 927
	Total MVB	−1 958 688	−1 600 454	−838 966	−563 512	−404 841
LS 60	Carbon (low)	17 840 101	13 985 254	8 453 557	4 924 134	3 537 626
	Carbon (mid)	*32 706 853*	*25 639 633*	*15 498 187*	*9 027 580*	*6 485 648*
	Carbon (high)	*40 140 228*	*31 466 822*	*19 020 502*	*11 079 302*	*7 959 658*
	Crops	−2 829 411	−2 005 539	−1 340 721	−706 140	−507 309
	Livestock	−3 380 027	−2 649 679	−1 601 630	−932 938	−670 247
	Timber	−7 647 623	−5 995 143	−3 581 116	−2 110 858	−1 516 495
	Total MVB	3 772 137	3 334 893	1 930 090	1 174 198	843 575

These values represent the discounted difference in value between each habitat restoration scenario and the PP baseline. The total marginal value of benefits (Total MVB) uses the low value for traded carbon (see text). The 3.5^{D} column is the MVC under a 3.5% declining discount rate (see text). Alternative (mid and high) carbon values are also presented in italics (see Appendix S1 for details).

difference recorded under Option A (Table 3). Costs of LS 30–60 were consistently slightly lower than those of the LS 60 scenario. Cost estimates of all three scenarios were sensitive to discount rate, with values decreasing as the discount rate increased (Table 4).

Values of NPV represent the difference between the total marginal value of benefits (MVB) (Table 2) and the total marginal value of costs (MVC) (Table 3). NPV values therefore provide an indication of the cost-effectiveness of the restoration options. Our results indicated that NPV values were negative for all scenarios irrespective of the discount rate used (Table 4). Cost–benefit analysis based on calculation of BCR similarly indicated that habitat restoration is not cost-effective (BCR values < 1, Appendix S6, Supporting Information). Variation in discount rate between 1% and 10% had relatively little impact on BCR

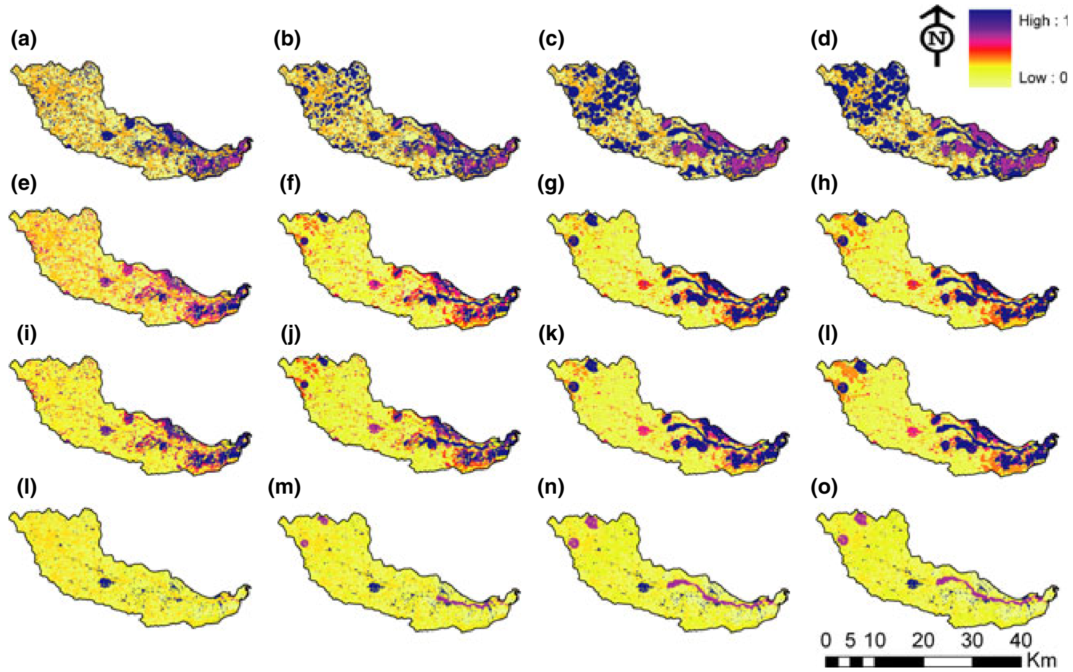

Fig. 1. Spatial variation in the non-market value of different ecosystem services across the Frome catchment. The four columns illustrate the four scenarios: (a), (e), (i), (l), the pre-project baseline (PP); (b), (f), (j), (m), LS 30; (c), (g), (k), (n), LS 30–60; and (d), (h), (l), (o), LS 60. The rows represent different ecosystem services, namely: (a), (b), (c), (d), flood risk mitigation; (e), (f), (g), (h), aesthetic value; (i), (j), (k), (l), recreational value; and (l), (m), (n), (o), cultural value.

Table 3. Marginal value (£) of restoration costs (MVC) under varying discount rates

	Discount rate (%)				
Scenario	0	1	$3{\cdot}5^{D}$	7	10
Option A (50 years)					
LS 30	74 415 678	58 336 113	35 261 971	20 539 838	14 756 353
LS 30–60	198 960 840	155 969 849	94 277 867	54 916 162	39 453 196
LS 60	208 074 601	163 114 330	98 596 435	57 431 696	41 260 421
Option B (10 years)					
LS 30	6 749 278	5 290 910	3 198 155	1 862 901	1 338 357
LS 30–60	39 199 640	30 729 474	18 574 803	10 819 686	7 773 143
LS 60	41 389 801	32 446 390	19 612 614	11 424 203	8 207 444

These values are the discounted difference in cost between each restoration scenario (LS 30, LS30–60 and LS 60) and the pre-project baseline (PP). Two options are presented: Option A, initial capital investment plus 50 years of maintenance; Option B, initial capital investment plus 10 years of maintenance. The $3{\cdot}5^{D}$ column is the MVC under a 3·5% declining discount rate (see text).

values, but lower ratios were recorded when discount rates were increased above zero.

BIODIVERSITY VALUE

Different land cover types contrasted markedly in terms of their biodiversity value. The number of species of conservation concern varied by more than two orders of magnitude between land cover types, with highest numbers associated with broad-leaved woodland (114 species), improved grassland (110) and dwarf shrub heath (92) (Appendix S4, Supporting information). When presented as species density values, highest values were associated with neutral grassland, fen/marsh/swamp and acid grassland. Species richness index varied spatially, with areas of relatively high value tending to be more extensive in the eastern part of the catchment, where semi-natural habitats such as heathland and broad-leaved woodland are concentrated (Fig. 2).

With respect to habitat connectivity, the least-cost buffer approach that was used enabled a total of 759 woodland habitat networks to be identified in the current landscape (PP). Respective values for heathland and grassland were 110 and 434 (Appendix S4, Supporting information). The number of independent networks decreased in each of the restoration scenarios compared with the current situation, in each of the three land cover types. This provides evidence of increasing habitat connectivity as a result of restoration, which is further illustrated by the increase in mean and maximum network area

Table 4. Net present value (£) of all monetary ecosystem benefits combined at different discount rates

Scenario	Discount rate (%) 0	1	$3{\cdot}5^D$	7	10
Option A					
LS 30	−72 457 128	−56 727 825	−34 289 822	−19 973 567	−14 349 530
LS 30–60	−200 919 528	−157 570 303	−95 116 833	−55 479 674	−39 858 037
LS 60	−204 302 464	−159 779 437	−96 666 345	−56 257 498	−40 416 846
Option B					
LS 30	−4 790 728	−3 682 622	−2 226 006	−1 296 630	−931 534
LS 30–60	−41 158 328	−32 329 928	−19 413 769	−11 383 198	−8 177 984
LS 60	−37 617 664	−29 111 497	−17 682 524	−10 250 005	−7 363 869

These values are the difference in values between the total marginal value of benefits (MVB) presented in Table 2 and the total marginal value of costs (MVC) presented in Table 3 for each scenario under the two different cost options: Option A, initial capital investment plus 50 years of maintenance; Option B, initial capital investment plus 10 years of maintenance. The total MVB is calculated using the low value of traded carbon. The $3{\cdot}5^D$ column is the net present value under a 3·5% declining discount rate (see text).

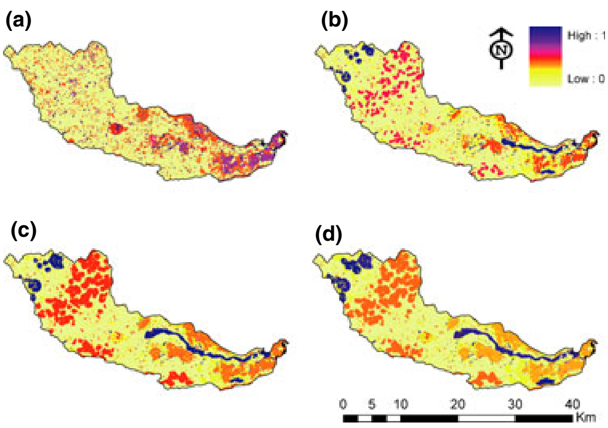

Fig. 2. Spatial variation in species richness index (standardised number of BAP species per ha) across the Frome catchment under (a) the pre-project baseline (PP), (b) LS 30, (c) LS 30–60 and (d) LS 60. Maps classes range between 0 and 1 where 0 = lowest biodiversity value, 1 = highest biodiversity value.

in all three land cover types in the restoration scenarios. However, the relative impact of the different scenarios varied between land cover types; whereas LS 60 consistently indicated greater habitat connectivity than LS 30, the LS 30–60 scenario was not always intermediate between the other two (Appendix S4, Supporting information). Very similar results were obtained when the analyses were repeated with larger buffer distances (1000 and 2000 m).

MULTI-CRITERIA ANALYSIS

Multi-Criteria Analysis was used to rank the alternative scenarios based on the weighted sum of the criteria scores, which provides a relative measure of combined ecosystem service provision. These results were then compared with ranking of scenarios based on measures of biodiversity value. As expected, the restoration scenarios ranked more highly than the current situation (PP) in terms of both species richness index and measures of habitat connectivity, for each of the three land cover types considered (Appendix S4, Supporting information). These results therefore suggest that landscape-scale habitat restoration would provide significant benefits to biodiversity.

In terms of ecosystem benefits, the scenarios ranked differently depending on how the ecosystem services were weighted. When weightings were based on market value, which excluded the non-market services, PP was found to rank more highly than LS 30–60. In contrast, when services were weighted equally or using weight values elicited from stakeholders, LS 30–60 ranked more highly than PP (Fig. 3). However, LS 60 consistently ranked first regardless of the weighting used, and LS 30 was also consistently ranked higher than PP. This indicates that habitat restoration is associated with increased provision of ecosystem services, regardless of the weightings of different services explored here.

When analysed by correlation, biodiversity value, as indicated by the species richness index, was positively related to

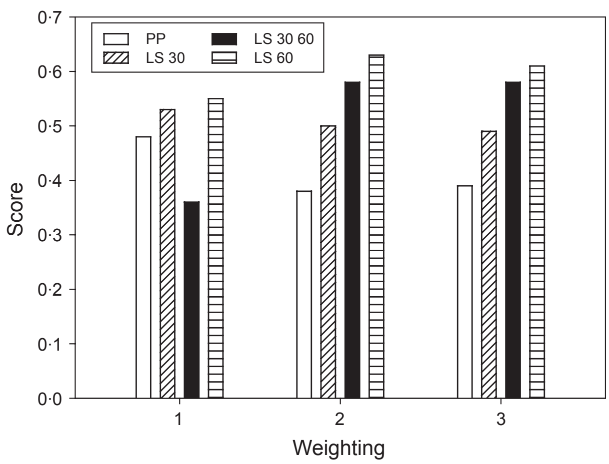

Fig. 3. Ranking of scenarios based on Multi-Criteria Analysis (MCA) results according to three different weighting methods: (1) based on market value, with non-market services (flood mitigation risk, cultural, aesthetic, recreation) given zero weight; (2) equal weighting of all services, including those with market and non-market values; (3) weighting of all services based on values elicited from stakeholders within the study area. The scores represent the outputs of the MCA, based on the weighted sum of the criteria scores.

flood risk mitigation and recreation, but negatively related to crop value ($P < 0{\cdot}05$ in each case, Appendix S4, Supporting information). This reflects the generally low value of agricultural land for biodiversity and indicates a potential trade-off between biodiversity conservation and agricultural production. Potential trade-offs between ecosystem services were also indicated by significant negative correlations observed between crop value and each of flood risk mitigation, aesthetic value and cultural value ($P < 0{\cdot}05$ in each case, Appendix S4, Supporting information), indicating the low value of agricultural land for these services. Carbon storage was positively correlated with timber value ($r = 0{\cdot}55$, $P = 0{\cdot}017$), and recreation and aesthetic values were also positively correlated ($r = 0{\cdot}98$, $P < 0{\cdot}001$) (Appendix S4, Supporting information).

Discussion

Although ecological networks have been widely incorporated into environmental policy, and scientific advice continues to recommend their implementation (Lawton *et al.* 2010), a number of reservations have been expressed regarding their value and effectiveness (Boitani *et al.* 2007). To date, no ecological network has been validated in practice, in terms of increasing the viability of multiple species or metacommunities (Boitani *et al.* 2007). Although the need for an evidence-based approach to conservation management is now widely recognised (Sutherland *et al.* 2004), few systematic reviews have been conducted that examine the effectiveness of ecological networks. In considering the specific role of hedgerows in increasing woodland connectivity, Davies & Pullin (2007) found little robust evidence of positive impacts on populations of individual species. In a more extensive systematic review of the impact of landscape features on species movement, Eycott *et al.* (2008) concluded that while some evidence exists that corridors do facilitate the movement of individual animal species, the evidence is based on a limited range of studies, restricting the ability to generalise across species and landscapes.

Little information is available on the impact of developing ecological networks on the provision of ecosystem services, although this has recently been identified as a research priority (Jones-Walters 2007). The current results illustrate how provision of selected ecosystem services could potentially be enhanced by ecological restoration actions, but also highlight a number of trade-offs between different ecosystem services, and with biodiversity. For example, areas of high value for production of crops were of relatively low aesthetic and recreational value (Appendix S4, Supporting information). The trade-off recorded here between production of agricultural crops and biodiversity value has been reported by a number of other studies (e.g. Nelson *et al.* 2009), although converse relationships have also been reported (Naidoo *et al.* 2008).

As noted by Balmford *et al.* (2011), the costs associated with different management interventions are difficult to estimate with precision, and the estimates presented here should therefore be viewed with caution. The values employed are primarily based on current agri-environment and woodland grant schemes, which are assumed to reflect the costs of meeting habitat restoration targets incorporated in national Habitat Action Plans (Rayment 2006). Overall, these estimates are likely to be conservative; in practice, the amounts provided under payment schemes may fail to fully cover the costs of restoration, which are often highly variable and site-specific (Rayment 2006). Furthermore, it is conceivable that the unit cost of habitat restoration might increase over time as it becomes necessary to include land that is more difficult or costly to manage. Another important caveat is that while agri-environment payment rates are intended to compensate landowners for loss of agricultural production, they may not fully cover such opportunity costs. The cost estimates used here varied from £350 to £2100 ha^{-1} for habitat establishment, depending on the habitat concerned, with annual maintenance costs varying from £200 to 450 ha^{-1} year^{-1}. These values fall within the range reported for restoration projects in a global review (TEEB 2009).

The cost-effectiveness of ecological networks is dependent on the increase in value of the benefits provided in relation to the costs incurred. The current results suggest that the monetary value of habitat restoration is highly dependent on the value of carbon stored. The other marketable services considered, namely crop, livestock and timber production, all declined in all restoration scenarios, reflecting the conversion of agricultural land and conifer plantations to other land cover types. A number of other investigations have similarly reported high monetary values of carbon storage relative to those of other ecosystem services (Naidoo & Ricketts 2006; Nelson *et al.* 2009; Birch *et al.* 2010). As a consequence, these results are sensitive to carbon price. The prices employed here (£44·04, £80·74 and £99·09 per tonne of carbon for traded values) were higher than those used in some previous studies (e.g. Naidoo & Ricketts 2006; Nelson *et al.* 2009), although they followed the current approach of the UK Government (DECC 2009). There is currently great uncertainty associated with forecasting carbon prices as these depend on the future commitments of major emitters and on the frameworks adopted to achieve these commitments. Currently, the market price of carbon is highly volatile; for example, during early 2011, it underwent a marked decline, reflecting the current global economic crisis. There is also great uncertainty regarding the potential income that landowners might receive in return for carbon storage, or for provision of other ecosystem services.

Such variation in market price highlights the uncertainty that is consistently associated with the valuation of ecosystem services (TEEB 2010); consequently, the results presented here should be viewed as tentative. This is illustrated further by the influence of discount rate on the results obtained. Discount rates are widely used in economic analyses to assess the present value of future benefits; typically, the value of any future amount of money is discounted at a chosen rate to estimate current net present value. However, there is no consensus between economists as to what rate should be applied to environmental management projects, if any (Newell & Pizer 2003).

Despite such caveats, the current results suggest that based on the services considered, development of ecological networks through habitat restoration is unlikely to provide net economic benefits. However, it is possible that inclusion of market values of additional services, such as flood risk mitigation and recreation, might enable net economic benefits to be achieved. Based on the results of a global meta-analysis, Woodward & Wui (2001) estimated the value of maintaining wetlands for their flood defence function at £1279 ha^{-1} $year^{-1}$, which would suggest an increase in value of £1·5–2·9 million $year^{-1}$ (in LS 30 and LS 60, respectively) resulting from wetland restoration in the current study. However, Woodward & Wui (2001) concluded that the prediction of a wetland's value based on previous studies remains highly uncertain and recommend that a site-specific valuation be performed, which was beyond the scope of the current investigation. With respect to recreation, the available statistics suggest that around two million tourists visit the study area each year (South West Regional Research Group 2003), suggesting significant market value, although again a comprehensive analysis was beyond the scope of the current investigation (Appendix S1, Supporting information).

Consequently, the non-market values of recreation and flood risk were examined here, together with aesthetic and cultural values. It is widely recognised that valuing ecosystem services that are non-market public goods is difficult, but highly important, as most services fall into this category (Fisher *et al.* 2008; Rouquette *et al.* 2009). The online mapping tool employed here demonstrates how the non-market values of different benefits held by local stakeholders can be elicited, in a spatially explicit manner. In addition, the current study demonstrates the value of MCA techniques as a decision-support tool (de Groot *et al.* 2010), enabling integration of both market and non-market values. The results of the MCA analysis indicate that despite the uncertainties involved, the development of ecological networks was consistently associated with increased overall provision of ecosystem services, regardless of the different weights explored, and whether or not non-market benefits were included in the analysis. In each case, the scenario with the largest area of restored habitat (LS 60) ranked more highly than the non-restored comparator (PP).

Crossman & Bryan (2009) highlighted the need to identify locations (or 'hotspots') that provide multiple ecosystem services to effectively target habitat restoration actions; such locations were identified using the approaches employed here. Although congruence between ecosystem services was here found to be generally low, as reported previously in other areas (Egoh *et al.* 2008), the spatial MCA enabled localised areas to be identified within the study catchment that displayed relatively high provision of multiple services. Such analyses could potentially be used to identify priority areas for restoration within a landscape based on a range of criteria, including the values held by different stakeholders.

Conclusions and recommendations

These results suggest that establishment of ecological networks through ecological restoration is unlikely to deliver net economic benefits, at least in intensively used landscapes such as that examined here. This reflects the high costs of ecological restoration in landscapes currently dominated by agricultural land use. Whether or not the increased provision of ecosystem services will outweigh the costs incurred, as suggested by Lawton *et al.* (2010), will depend critically on how the benefits provided to people are valued. At present, relatively few ecosystem services have a readily quantifiable market value, limiting the scope for cost–benefit analyses. However, the current results suggest that the overall market value of the increase in provision of ecosystem services arising from the development of ecological networks is highly dependent on carbon price.

These results of this research suggest that the conservation benefits of developing habitat networks will need to be traded off against reduced agricultural productivity. The overall cost–benefit analysis of ecological networks will probably depend on how the value of non-market benefits and the needs of biodiversity are weighted against the opportunity costs of reduced crop and timber production. Payment schemes such as AES will need to provide sufficient income to landowners to compensate for the opportunity costs incurred. This might be achieved by providing payments for a range of different ecosystem services, including flood risk mitigation and carbon storage.

The research also demonstrated how local stakeholder values of non-market ecosystem services, such as cultural and aesthetic value, can be elicited using interactive online tools. The future development of habitat restoration plans could usefully incorporate such values, to ensure that local people benefit from restoration actions.

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Supporting Information

Additional Supporting Information may be found in the online version of this article.

Appendix S1. Additional details of methods.

Appendix S2. Results of on-line survey.

Appendix S3. Details of MCA.

Appendix S4. Biodiversity value.

Appendix S5. Extent of land cover types in the study area.

Appendix S6. Benefit:cost ratios.

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