# Cost-benefit analysis of ecological networks assessed through spatial analysis of ecosystem services

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4 Tables, 3 Figures, 6 Appendices

**Running title:** Cost-benefit analysis of ecological networks

## Summary

1. The development of ecological networks represents a potential approach for adaptation to climate change, by enhancing the dispersal of species across fragmented landscapes. Development of such networks will require widespread ecological restoration at the landscape scale, which is likely to incur significant costs. However, little evidence 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 selected ecosystem services (carbon storage, crops, livestock and timber) under three different restoration scenarios, estimating restoration costs, and calculating net benefits.

3. The non-market values of additional services (cultural, aesthetic and recreational value) were elicited from local stakeholders using an on-line survey tool, and 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. MCA 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 particularly 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 maximized.

**Keywords:** climate change, ecosystem benefit, biodiversity, ecological restoration, habitat connectivity

## Introduction

Many countries have now incorporated the concept of ecological networks into national policies (Jongman and 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 (Boitani *et al.* 2007). According to Opdam *et al.* (2006), the 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 emphasize the importance of connectivity among patches of habitat to ensure the viability of both populations and communities of species (Boitani *et al.* 2007; Jongman and Pungetti 2004). The growth in interest in ecological networks reflects increasing concern regarding the reduction and fragmentation of natural and semi-natural habitats, which are now recognised as major causes of biodiversity loss (Fahrig 2003).

In the context of 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 now widely recognised (Bullock *et al*. 2011). However, the costs of ecological restoration can be substantial. Typical restoration costs range from hundreds to thousands US$ $ha^{-1}$, 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 previous attempts have been made to perform a cost-benefit analysis of restoration initiatives. In a review of over 2,000 restoration case studies, TEEB (2009) found that less than 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) recently 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 emphasized 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 (Balmford *et al.* 2011; Groot et al. 2010; Nelson *et al.* 2009).

Although progress has recently been made in mapping the value of ecosystem services (Kareiva *et al.* 2011; Nelson *et al.* 2009), 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 involved mapping the values of five ecosystem services together with restoration costs, to examine the costs and benefits of restoring native forest communities. Results indicated that cost effectiveness was dependent on the restoration methods used and on 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 provide adaptation to climate change. 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.

## Methods

### SCENARIO DEVELOPMENT

This investigation was undertaken at the scale of a single river catchment, namely that of the River Frome, Dorset, southern England (Appendix S1 in 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 the 'South West Nature Map', a regional approach to landscape-scale planning for habitat restoration that has recently been developed by conservation organisations in South West England (Brenman 2005), as part of the national 'Living Landscapes' initiative (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 *et al.* (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):

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),

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 conducted 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. Details of the methods used are given in Appendix S1.

## 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 timescales (Option A: 50 years and Option B: 10 years), to explore sensitivity of 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 $ha^{-1}yr^{-1}$. 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% to 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 net present value (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):

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

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

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

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 (H.M. Treasury 2003) regarding discounting over this timescale. 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.

## 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 noneconomic values to be incorporated in the same analysis (Appendix S1). 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.

In order 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 Appendix S1, S2);
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).

In order 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., USA) (see Appendix S4).

## 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 broadleaved 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). 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. Broadleaved 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).

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 (Figure 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 (Figure 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 five-fold difference recorded under Option B and a more than two-fold 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. 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 benefit:cost ratios (BCR) similarly indicated that habitat restoration is not cost effective (BCR values <1, Appendix S6). Variation in discount rate between 1-10% had relatively little impact on BCR 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 broadleaved woodland (114 species), improved grassland (110) and dwarf shrub heath (92) (Appendix S4). 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 broadleaved woodland are concentrated (Figure 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). The number of independent networks decreased in each of the restoration scenarios compared to 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 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). Very similar results were obtained when the analyses were repeated with larger buffer distances (1000 m and 2000 m).

## MULTI-CRITERIA ANALYSIS (MCA)

MCA 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). 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 (Figure 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 flood risk mitigation and recreation, but negatively related to crop value ($P< 0.05$ in each case, Appendix S4). 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 ($P< 0.05$ in each case, Appendix S4), indicating the low value of agricultural land for these services. Carbon storage was positively correlated with timber value ($r = 0.55$, $P = 0.017$), and recreation and aesthetic values were also positively correlated ($r = 0.98$, $P<0.001$) (Appendix S4).

## 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 meta-communities (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 and 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). 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-£2100 $ha^{-1}$ for habitat establishment, depending on the habitat concerned, with annual maintenance costs varying from £200-450 $ha^{-1}$ $yr^{-1}$. 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 (Birch *et al.* 2010; Naidoo & Ricketts 2006; Nelson *et al*. 2009). 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 £1,279 $ha^{-1}$ $yr^{-1}$, which would suggest an increase in value of £1.5 – 2.9 million $yr^{-1}$ (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, 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).

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 on-line 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 largest area of restored habitat (LS 60) ranked more highly than the non-restored comparator (PP).

Crossman and Bryan (2009) highlighted the need to identify locations (or ‘hotspots’) that provide multiple ecosystem services in order 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.

This research suggested 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 likely 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 on-line 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

**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). 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). The italicised numbers therefore do not contribute to the total values presented.

| **Ecosystem services** | **Present Value of Benefits (PVB) (£)** | | | |
|---|---|---|---|---|
| | **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** |

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

| **Scenario** | **Ecosystem service** | | **Discount rate (%)** | | | | |
|---|---|---|---|---|---|---|---|
| | | | **0** | **1** | **3.5**[D] | **7** | **10** |
| **LS 30** | Carbon | $C_{low}$ | 7,363,683 | 5,772,556 | 3,489,291 | 2,032,487 | 1,460,191 |
| | | $C_{mid}$ | 13,500,086 | *10,583,019* | *6,397,034* | *3,726,225* | *2,677,017* |
| | | $C_{high}$ | 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 | $C_{low}$ | 11,886,804 | 9,318,331 | 5,632,578 | 3,280,935 | 2,357,109 |
| | | $C_{mid}$ | 21,792,473 | *17,083,607* | *10,326,393* | *6,015,048* | *4,321,367* |
| | | $C_{high}$ | 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 | $C_{low}$ | 17,840,101 | 13,985,254 | 8,453,557 | 4,924,134 | 3,537,626 |
| | | $C_{mid}$ | 32,706,853 | *25,639,633* | *15,498,187* | *9,027,580* | *6,485,648* |
| | | $C_{high}$ | 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** |

**Table 3**. Marginal value (£) of restoration costs (MVC) under varying discount rates. 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 $3.5^{D}$ column is the MVC under a 3.5% declining discount rate (see text).

| **Scenario** | | **Discount rate (%)** | | | | |
|---|---|---|---|---|---|---|
| | | **0** | **1** | **$3.5^{D}$** | **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 |

**Table 4**. Net present value (£) of all monetary ecosystem benefits combined at different discount rates. 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 $3.5^{D}$ column is the net present value under a 3.5% declining discount rate (see text).

| **Scenario** | **Discount rate (%)** | | | | |
|---|---|---|---|---|---|
| **Option A** | **0** | **1** | **$3.5^{D}$** | **7** | **10** |
| 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 |

**Figure 1**. Spatial variation in the non-market value of different ecosystem services across the Frome catchment. The four columns illustrate the four scenarios; (A) the pre-project baseline (PP), (B) LS 30, (C) LS 30-60 and (D) LS 60. The rows represent different ecosystem services, namely (A) food risk mitigation, (E) aesthetic value, (I) recreational value, and (L) cultural value.

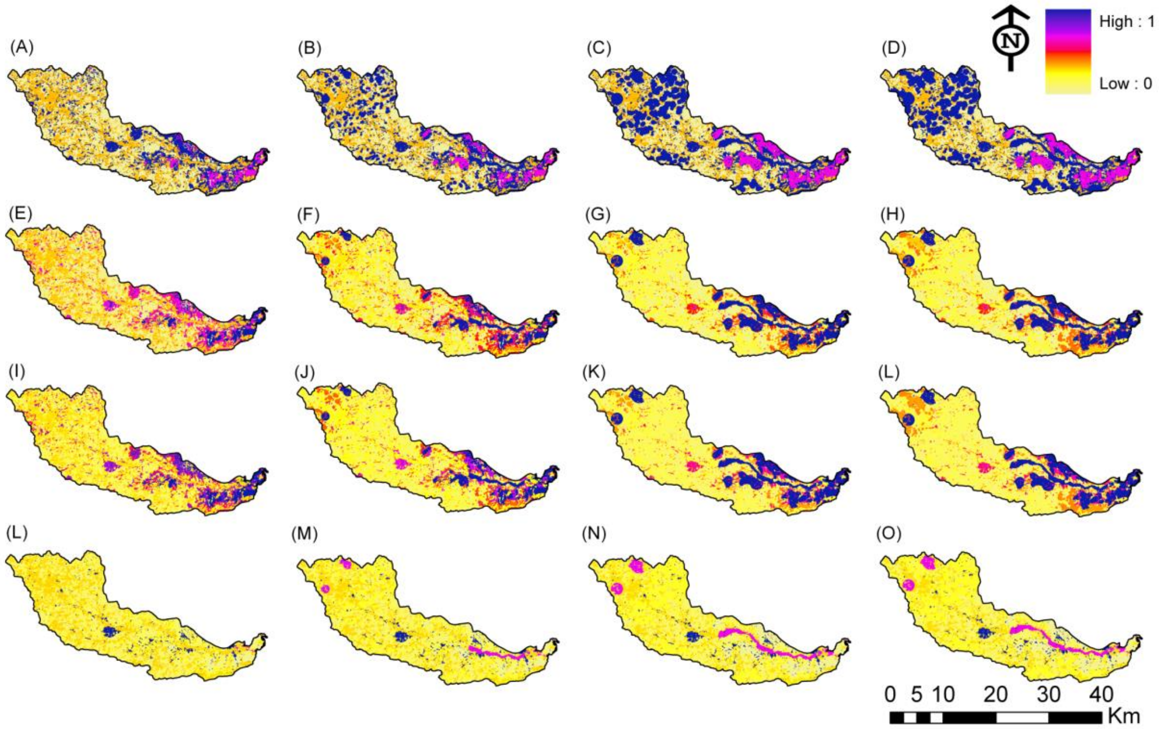

**Figure 2.** Spatial variation in species richness index (standardized number of BAP species per ha) across the Frome catchment under (A) the pre-project baseline (PP), (B) LS 30, (C) LS 30-60 and (D) LS 60. Maps classes range between 0 and 1 where 0 = lowest biodiversity value, 1 = highest biodiversity value.

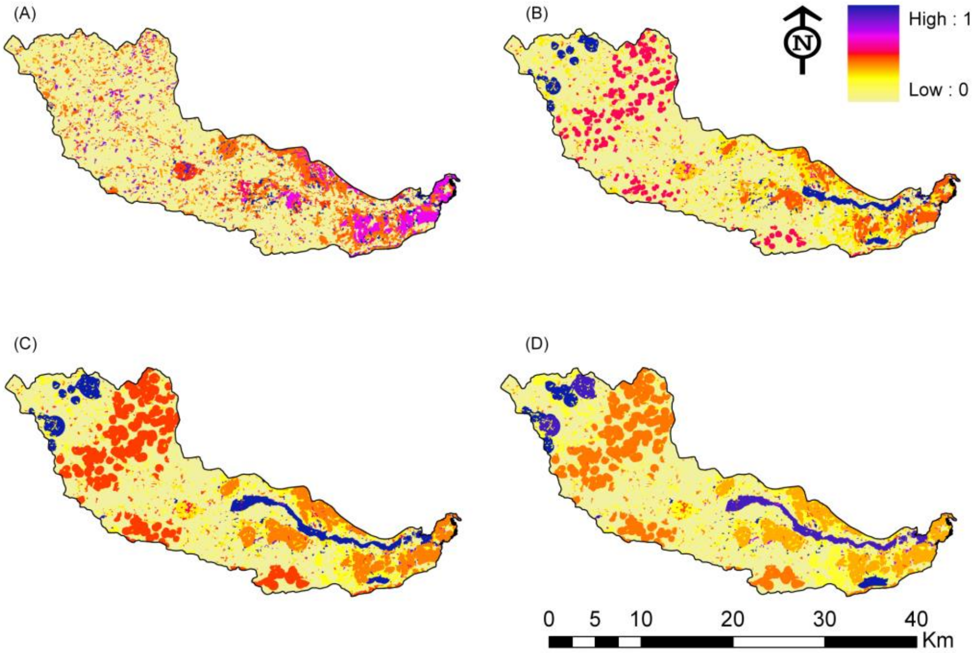

**Figure 3**. Ranking of scenarios based on MCA results according to three different weighting methods: (1) based on market value, with non-market services (flood mitigation risk, cultural, aesthetic, recreation) given zero weight; (2) equal weighting of all services, including those with market and 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.