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Exploring ecosystem service issues across diverse knowledge domains using Bayesian Belief Networks

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Abstract

The analysis of the relationships between people and nature is complex, because it involves bringing together insights from a range of disciplines, and, when stakeholders are involved, the perspectives and values of different interest groups. Although it has been suggested that analytical-deliberate approaches may be useful in dealing with some of this complexity, the development of methods is still at an early stage. This is particularly so in relation to debates around the concept of ecosystem services where biophysical, social and economic insights need to be integrated in ways that can be accessed by decision-makers. The paper draws on case studies to examine the use of Bayesian Belief Networks (BBNs) as a means of implementing analytical-deliberative approaches in relation to mapping ecosystem services and modelling scenario outcomes. It also explores their use as a tool for representing individual and group values. It is argued that when linked with GIS techniques BBNs allow mapping and modelling approaches rapidly to be developed and tested in an efficient and transparent way, and that they are a valuable scenario-building tool. The case-study materials also show that BBNs can support multicriteria forms of deliberative analysis that can be used to capture stakeholder opinions so that different perspectives can be compared and shared social values identified.

Keywords

analytical deliberative techniques, Bayesian Belief Networks, BBN, mapping ecosystem services, multicriteria methods, non-monetary valuation, scenarios

I Introduction

The assessment of ecosystem services poses a number of disciplinary challenges. Although the proposition that the outputs ecosystems can contribute to human well-being (MA, 2005) is a simple and attractive one, we face considerable difficulties in showing how these links operate, and in particular how we can use this understanding in decision-making. Sutherland et al. (2009) have sought to identify the most pressing problems that confront conservation biology, and, if we consider those that relate specifically to ecosystem services, it is clear that their solution will require inter- and transdisciplinary[1] approaches. The identification of thresholds in relation to the impact of biodiversity loss on

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service output and the consequences for human well-being, together with the problem of valuation and the use of valuation data in shaping management responses, are tasks that will demand the integration of understandings across different subject areas. Dialogue and knowledge exchange between the natural and social sciences is essential. Moreover, the need to apply those insights will mean that researchers will also often have to contend with the norms of unfamiliar knowledge cultures that approach questions about the nature of evidence, uncertainty and risk in unfamiliar ways. As a result, we must find new ways of supporting such dialogues.

The need for inter- and transdisciplinary approaches is not unique to questions about ecosystem services, nor as a general requirement in the environmental arena is it new. A number of commentators have reflected on the contemporary approaches to science that are required in the context of sustainability (e.g. Kates et al., 2000) and argued that traditional scientific rationalities must be in wider sets of social processes (Potschin and Haines-Young, 2006). Rather than explore these philosophical questions further, however, the aim of this paper is to accept the challenges they pose, and examine some methodologies that might help us negotiate these sometimes unfamiliar research and policy agendas. The particular approach that will be considered here is that of Bayesian Belief Networks (BBNs). Although they have been used widely to construct decision support tools, they also have much to offer in relation to questions of ecosystem services. They can help in the analysis of marginal changes in outputs and values associated with different policy options, and in examining the kinds of trade-off that might need to be considered when exploring alternative scenarios. It will be argued that they can help to link different types of knowledge and therefore help communicate insights across knowledge cultures and support participatory approaches to ecosystem assessment that conform to the kind of analytical-deliberative frameworks called for by Fish (2011).

II Bayesian Belief Networks and ecosystem service cascades

Cain (2001) has suggested that BBNs can be viewed as graphical tools for building decision support systems under uncertain conditions. Elsewhere Smith et al. (2007: 334) have suggested that they are a tool for ‘organising current thinking, and generating testable hypotheses that can be calibrated, validated and updated as new knowledge and data becomes available’. The ability to organize and make knowledge operational is a key part of understanding the utility of BBNs. On the one hand, the BBNs can be used to represent current knowledge by setting out the causal relationships between variables in an influence diagram. Such diagrams can be used as a focus of discussion between experts and/or stakeholders to elicit their views about how a system might work, what its key elements are, and hence what the boundaries of the system of interest might be. On the other hand, the networks can be used to operationalize knowledge by expressing the strength and certainty we have about the relationships as a set of probabilities. Their structure allows the implications of changes in the state of a given variable (known as a node) to be traced through the network so that the sensitivity of the outputs to different combinations of input variables can be examined. A particular advantage of BBNs is that they can combine both quantitative empirically determined relationships as well as more qualitative expert defined ones; the node states can be defined either as a set of distinct classes or as a series of quantitative ranges. A number of environment-related studies demonstrate the versatility of BBN methods.

In the study by Smith et al. (2007), for example, the aim was to use a BBN to construct a habitat suitability model for an endangered mammal, the Julia Creek dunnart (*Sminthopsis douglasi*). The probability that a given area was rated as ‘high’, ‘medium’ or ‘low’ suitability for the animal was determined by a set of quantitative and qualitative relationships, some of which were based on field measurements (e.g. the relationships between mapped variables and corresponding environmental conditions on the ground) and some on expert judgement (e.g. the relative strengths of the different environmental variables as determinants of habitat suitability). The network was then used to predict and map habitat suitability for different land areas based on their ecological and management status. Elsewhere, Newton et al. (2007) have described the use of BBNs in the context of evidence-based conservation management, and Stelzenmüller et al. (2010) have proposed a BBN that can serve as a decision support tool for spatial marine planning. In the latter study, a set of characteristics for the different marine spatial areas off England and Wales were used to predict the vulnerability of the 'marine landscape' to changes in human pressures such as aggregate extraction, fishing and the development of oil and gas infrastructure. The estimation of sensitivities was made on the basis of a literature review. Finally, Zorrilla et al. (2010) have gone on both to describe the development of a BBN for participatory water resources management and to evaluate stakeholders' responses to the use of the methods. They found that the ability of these approaches to incorporate stakeholder values into decision-making was identified as particularly important, along with the increased understanding of stakeholder concerns and system characteristics. These features could clearly be beneficial in studies dealing with ecosystem services.

If we are to understand how coupled socio-ecological systems operate, we need to unpack and model the 'production chains' that lead to the output of ecosystem services. Although the examples of BBNs identified above are clearly relevant to those making ecosystem assessments, an explicit link between the assessments and modelling of ecosystem services and BBN techniques has not been widely made. It is suggested that one way to begin this task is to represent these production chains in terms of a 'cascade' (Potschin and Haines-Young, 2011; see also De Groot et al., 2010; Haines-Young and Potschin, 2010), which itself is a graphical device designed to capture the key elements of the debates that have developed around the service concept; the key elements are the underlying ecological structures and processes, their functional characteristics or capacity to deliver a service, the service itself and the benefits and values that flow from them. The cascade model is clearly a simplification of 'real world ecosystems' in that most services depend on a number of functional properties and many more structural components and processes. Similarly, there is often no single one-to-one relationship between benefits and values. Nevertheless, it does provide something of an analytical template that can be used to identify the different elements that have to be taken into account when making an assessment or analysis of ecosystem services. In this paper, two case studies are therefore presented to explore how BBNs can be used to operationalize different components of the cascade model, and hence promote inter- and transdisciplinary dialogues. Both involve working with scenarios and assessing their impact on an aspect of ecosystem services; the first uses a network as part of an empirical, data-driven approach, while the latter is used for the elicitation of stakeholder beliefs and values. Smith et al. (2011) have argued that while some recent studies have used BBN techniques to look at issues relevant to ecosystem services, further work is needed in order to use them to assess complex spatial-temporal systems. The purpose of this paper is to stimulate such work so that more rapid progress can be made.

III Case study: mapping ecosystem services and modelling scenario outcomes

1 Background

Ecosystem assessments commonly include an attempt to map services and the external pressures upon them (Chan et al., 2006; Imhoff et al., 2004; Naidoo et al., 2008; Schröter et al., 2005). Many also seek to examine the spatial impact of future scenarios on ecosystem services and human well-being (Swetnam et al., 2010). Karvera et al. (2011) have recently assembled a wide body of material which demonstrates that although a number of different approaches have been employed, a common feature is that they rely on the use of spatial proxies derived from land-cover data and depend on empirical or modelled relationships.

Tallis and Polasky (2011a), for example, provide an account of the InVEST tool, which uses land cover along with a range of other spatial ecosystem and socio-economic characteristics to build 'production' and 'value' functions. The modelling approaches are described in terms of a set of 'tiers' in which simple models based on readily available and sometimes generalized data are contrasted with analysis at tiers 2 and 3, where models are more complex and more demanding in their construction and data requirements. Because of their simplicity, models at the lowest tiers are generally more easy to understand but more prone to error. Tier 2 and 3 type models, on the other hand, are time-consuming and difficult to apply, but are often capable of describing real outcomes at specific places. Tallis and Polasky (2011b) argue that all types of mode have their role. The challenge is to understand the requirements of different decision-making contexts and what is gained in moving from simple to more complex approaches. If modelling tools are to be used as part of a stakeholder-driven process it may be difficult, for example, to lose the transparency of assumptions and understandability associated with simpler approaches.

The material for the first case study arose from work that contributed to the UK National Ecosystem Assessment (UK NEA, 2011). The UK NEA looked at the relationships between different land cover of habitat types and their capacity to supply a range of different ecosystem services, and as part of this a set of scenarios was constructed to examine how ecosystems and their services might change at national scales for a set of plausible but contrasting futures (Haines-Young et al., 2011). The scenarios were developed with the help of stakeholders (potential users of the UK NEA). This consultation process allowed the identification of a set of focal questions around which the analysis of alternative futures could be built. Altogether six storylines for 2060 were created (each with high and low climate impact variants): *Green and Pleasant Land*, *Nature@Work*, *World Markets, National Security*, *Local Stewardship* and *Go with the Flow*. The scenarios were based on the same direct and indirect drivers of change used to examine the current status of ecosystem services in the UK, and their impacts for each storyline were initially described qualitatively and tested by a second round of consultation.

In the UK NEA the qualitative scenarios proved useful as a focus for discussion with stakeholders, but it was clear that they did not provide the kind of information that was needed to make a valuation of the impacts of the different futures. Thus an effort was made to quantify the scenario outcomes by using a BBN to capture their key assumptions and map the patterns of land-cover change that they implied using a GIS.[2] The BBN thus helped to make the scenarios operational in the sense that from these projections of land-cover change the impacts of different services might be deduced or estimated quantitatively. It was also felt that the mapping exercise could provide an additional way to examine the plausibility of the scenario assumptions. The analysis of the economic impacts of the scenarios has been reported elsewhere (see Bateman et al., 2011). This case study focuses on the methodological issues involved in making the projections of land-cover change, and how production functions might be built using the BBN approach. The example of estimating vegetation carbon is used as the basis for

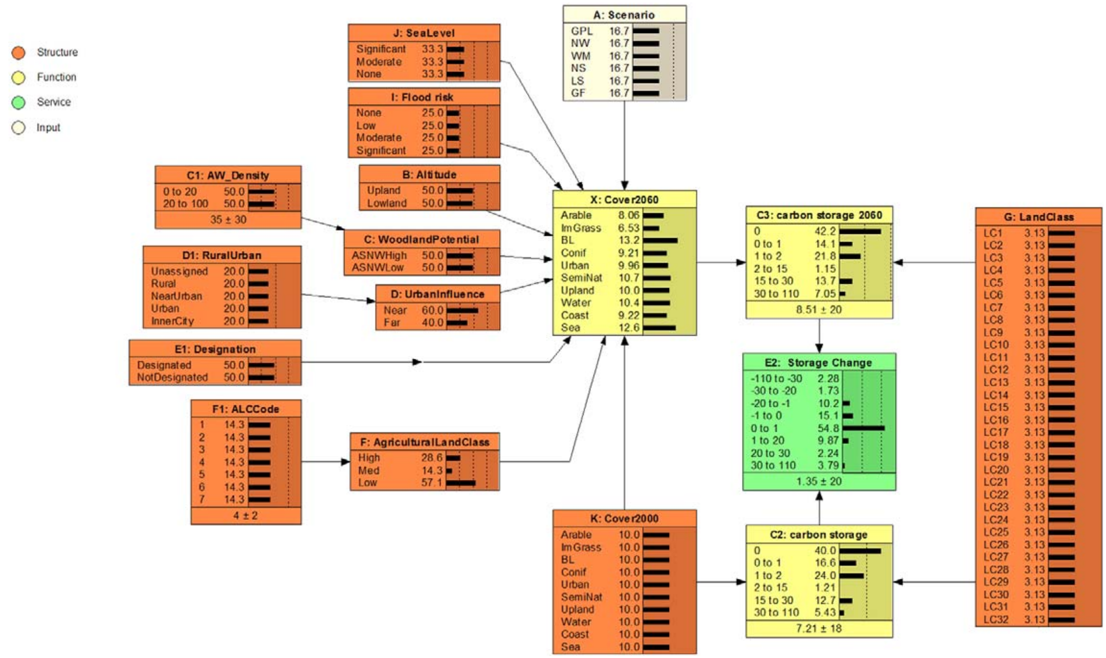

Figure 1. BBN used to estimate change in vegetation carbon for UK NEA scenarios

discussion. Although the model presented is 'tier 1' in character, it will be argued that the BBNs have the potential to develop more sophisticated yet transparent 'meta-models' that can address some of the complexity surrounding the links between land-cover and ecosystem services.

2 Case-study methods

The structure of the influence diagram used for the case study is described in Figure 1; it shows the particular setup used for estimating vegetation carbon. As a baseline, the model used the land cover reported in each of the 1 km × 1 km Ordnance Survey grid squares that make up Great Britain (GB), as reported in Land Cover Map 2000. A simplified, coarse-resolution version of these data has been made publically available via the *Countryside Information System V6* (CIS[3]); these data recorded land cover in terms of the proportions of 10 general categories in each grid square. The cover classes were: arable, improved grassland, broadleaved woodland, coniferous woodland, urban, semi-natural, upland, water, coast and sea.[4] The cover categories are highly aggregated ones; the classes labelled 'semi-natural' and 'upland', for example, are particularly heterogeneous. The 'semi-natural' class amalgamates the neutral grassland, calcareous grassland, acid grassland, bracken and fen, marsh and swamp broad habitats, while 'upland' merges dwarf shrub heath, bog, montane and inland rock. It should be noted, however, that the classes in the 'upland' category do not exclusively occur at higher altitudes in GB, and so the label is somewhat misleading. The terminology and grouping used in the CIS was retained, however, because it was used both to provide a general summary of major land-use patterns and to take account of some of the misclassification problems in the original data.

The BBN was designed to project how the proportions of the 10 general categories would change under each scenario depending on the geographical context of each 1 km × 1 km cell.

Thus key components are the nodes for land cover in 2000 (Node K) and the projection for land cover in 2060 (Node X). The bar charts for land cover at the start and end nodes show the proportions of the different cover types present in a given area (or the probability of finding such a cover type within a defined spatial unit). The change over time is implemented using a transition matrix that expresses the probability that a given cover type in a 1 km cell either remains the same or changes to one of the other types. A transition matrix is commonly used for describing and modelling land-cover change (see, for example, Kocabas and Dragicevic, 2007); the advantage of using a BBN to construct such a transition matrix is that the accounting aspects related to the shifts of land area between classes is done in a single step.

In constructing the scenarios for the UK NEA it was assumed that not only would the storylines differ in the pattern and magnitude of the land-cover changes associated with them, but also the spatial distribution of these changes would differ. The other nodes of the influence diagram define how the context variables operate. It was assumed, for example, that the type of land-cover changes would differ between upland and lowlands, and may be affected by such factors as proximity to urban areas, the density of ancient woodlands and agricultural land quality. Using nine input variables for Node X and 10 land-cover types, however, the matrix that lies behind the transition for land cover was very large (10 × 34,560). Thus it was constructed using a spreadsheet that allowed a basic 10 × 10 transition matrix for land cover for each scenario to be modified semi-automatically according to the assumed influence of the context variables in each scenario.[5] Thus, in the case of woodland, for example, under the *Green and Pleasant Land* scenario it was assumed that new planting would be targeted partly where existing ancient woodland density was highest; this enters the analysis via nodes C and C1. By contrast, in *Nature@Work* these and areas close to population centres would be prioritized, while in *World Markets* limited or no overall woodland expansion is envisaged.

The BBN was used to produce mapped output for each scenario by creating a spreadsheet that recorded the characteristics of each of the 235,980 1 km × 1 km for GB. The characteristics were stored either as numerical (ancient woodland density in 5 km radius) or categorical (designation) variables that could then be easily be recoded, as required, using the BBN. In the case of ancient woodland density, for example, Node C was used to relabel the cell as being of low or high potential for new plating based on the numerical score, using 20% as the threshold between the classes; this allowed for more rapid prototyping. For land cover, the proportions of each type were stored as part of the record for each cell, alongside a unique identifier. To generate output from these data, the Netica 'process cases' tool was used. This allows cases to be read one by one, and corresponding output created according to the combination of input variables. These output data can then be mapped using a unique identifier that links to a vector file for the cells of the basic 1 km × 1 km grid.

3 Case-study results

The BBN shown in Figure 1 was designed to estimate the standing crop of carbon in vegetation, based on estimates of carbon densities for different cover types provided by Milne and Brown (1997). The general approach is a standard one that employs converting land-cover stock to estimate of carbon standing crop by means of a set of rating factors (see, for example, Cantarello et al., 2011; Conte et al., 2010; Eaton et al., 2008; Fisher et al., 2011; Swetnam et al., 2010). The BBN differs from the version used to make estimates in the UK NEA by including 'ITE Land Class' as a parameter controlling the carbon densities of woodlands in different types of biophysical environment. As Milne and Brown (1997) note, the bulk of the vegetation carbon in GB is stored in woodlands, and so the mapping is particularly sensitive to estimates of carbon densities for the different forest types. Land Classes (Bunce et al., 1996) have been used to describe the range of environmental conditions across GB and are therefore a useful way of capturing the variations in woodland character; they were also used by Milne and Brown (1997) to refine estimates of vegetation carbon for woodlands.[6] Inclusion of the influence of Land Class on the network has therefore been used to illustrate how a wider range of environmental factors might be included in such models, compared to the original UK NEA version.

Using the BBN, the total vegetation carbon for GB in 2000 is estimated at approximately 121 Mtonnes.[7] Milne and Brown (1997) make their estimates of total vegetation carbon for 1990 to be around 114 Mtonnes. Given that their estimate was made using Land Cover Map 1990 and that their calculation method was different in detail to the one used here, the two were considered sufficiently close to justify the further use of the BBN tool in the scenario exercise. Thus, using the same carbon densities and the same patterns of influence for Land Class, equivalent estimates for vegetation carbon were made for each storyline, and their associated spatial distributions of vegetation carbon and their marginal changes compared to the 2000 baseline mapped. Figure 2 illustrates the mapped output that can be obtained from the BBN; the pattern of vegetation densities for the 2000 baseline are shown, together with the marginal change for two contrasting scenarios, *Green and Pleasant Land* and *World Markets*. By virtue of the significant expansion of woodland cover (from around 6% in 2000 to 12% by 2060) *Green and Pleasant Land* results in an increase of around 50% in the amount of carbon stored in vegetation, whereas *World Markets* appears to show a small decline compared to the baseline.

4 Evaluation of case-study outcomes

The BBN model for vegetation carbon is simple in that it does not take account of changes in soil carbon, nor does it use the wider range of estimates of vegetation carbon densities that are available (see, for example, Cantarello et al., 2011). In terms of judging the plausibility of these projections for 2060, the BBN result is possibly an overestimate because the model assumes that all the additional woodland has the same carbon densities as the mature stock, which is unlikely. Since planting will occur progressively over the 50-year time period being considered, the new woodland is likely to have a lower average carbon density than the older stock, although this may be offset partly as the age of the initial stock increases. The carbon density values derived from Milne and Brown (1997) for broadleaves and conifers were the average for all the main species in the groups, weighted by the age distribution at that time. Thus modification to the conversion factors to take account of forest age for the model is therefore likely to be small and unlikely to change the broad spatial patterns that are projected. The plausibility of the mapping probably depends more on the assumptions of the land-cover changes implied by the scenarios than the carbon calculation itself.

The experience in the UK NEA suggested that the graphical representation of the BBN and the ability to code into the network both quantitative and qualitative types of data enabled prototype models to be developed rapidly in a transparent way. They were also found to have the advantage that they can be used to integrate data at different spatial scales. In the current model, for example, the influence of the neighbourhood characteristics of grid cells was taken into account by including ancient woodland densities in a 5 km radius, and differences in carbon densities in different environments were included using the influence of the Land Class node. Moreover, the simple way in which the network input and outputs could be linked to mapping software meant that the consequences of the modelling assumptions could be examined. In addition to the agreement between the current estimate for total stock of vegetation

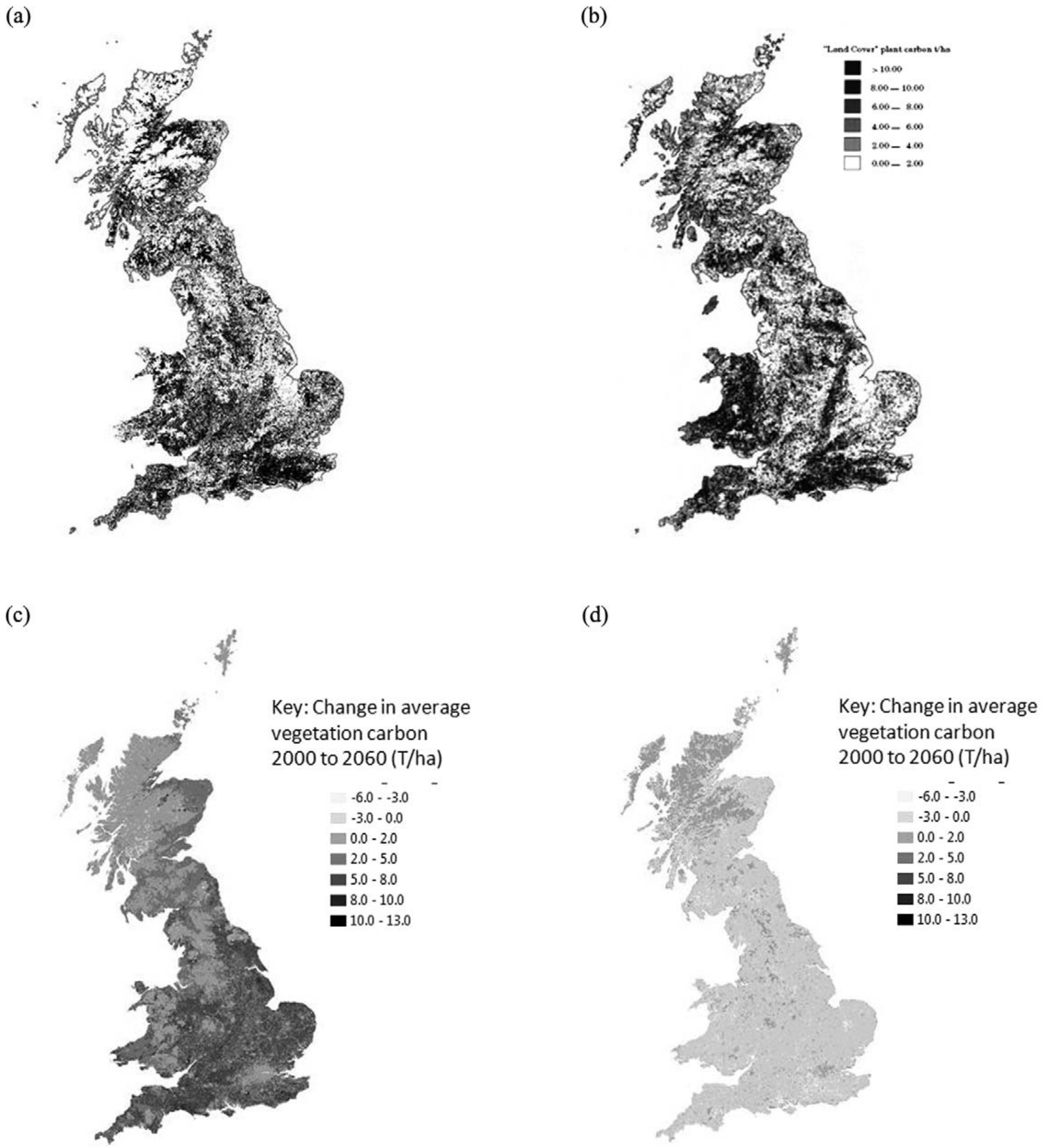

Figure 2. Mapping of vegetation carbon. (a) Variation of vegetation carbon in 2000 predicted by BBN model. (b) Variation of vegetation carbon in 1990 estimated by Milne and Brown (1997), both (a) and (b) have a common key. (c) Marginal changes in vegetation carbon between 2000 and 2060 for *Green and Pleasant Land* scenario. (d) Marginal changes in vegetation carbon between 2000 and 2060 for *World Markets* scenario.

carbon and that of Milne and Brown (1997), for example, the general correspondence between the baseline mapping for 1990 and 2000 is satisfactory. Most importantly, the associated mapping of land cover for the scenarios was also found to provide an acceptable basis on which estimates of economic change could be made (see Bateman et al., 2011).

The case study suggests that BBNs can be a useful tool in constructing the kind of production functions that are needed for making ecosystem assessments. Elsewhere, mapping studies by Burkhard et al. (2009) and Kienast et al. (2009) have used deterministic look-up tables to model the capacity of land to generate more general suites of ecosystem services; the conclusion derived from this case study was that BBNs could provide a way in which more nuanced, probabilistic approaches can be developed. Unfortunately, the time available in the UK NEA did not allow the rating information used for the economic analysis of services, such as agricultural production, recreation and greenhouse gas emissions to be built into an extended set of BBNs, so that the approach could be explored further.

Although the model presented in this first case study is clearly 'tier 1' in character, in principle BBNs can be used to develop more complex (tier 2 and 3) models. In so doing, it is likely that there will be a need to loosely couple them with other software tools, and in particular other process specific models. However, while greater refinement is a laudable aim, the merits of using this tier 1 approach should not be overlooked. They can serve as useful communication and learning devices. Moreover, the experience gained in the UK NEA suggests that BBN approaches may have the capacity to bridge the divide between tiers 1, 2 and 3, by integrating different kinds of knowledge in a meta-model type structure. While some nodes might be based on expert judgement, because appropriate empirical information may not be available, other nodes can be measured and based on empirical data (as for carbon in the current case study) or be used to summarize modelled process-response patterns (say, in terms of the speed to reach equilibrium in the soil carbon store). By linking a range of relationships together in an integrated framework these meta-models can therefore be used to explore the potential trade-offs between different types of service. The interactive character of these BBNs also enables the sensitivity of outcomes to different combinations of input parameters to be explored. In this way estimates of the marginal changes in output can be made and assessments made in physical or monetary terms.

IV Case study: identifying shared social values

1 Background

Fish (2011) has argued that analytical deliberative techniques must bring 'facts and values in to more open play'. This second case study focuses on how BBNs can be used to work in a deliberative way with stakeholders to achieve this aim. The issue described here concerns identifying the impacts of different futures on a set of landscape characteristics and how this affects landscape as a cultural entity. The study was designed primarily to elicit the factual understandings of a set of countryside issues held by stakeholders drawn from a range of natural and social science backgrounds, and explore the collective values they apply to different dimensions of landscape. For the purposes of the present discussion we regard landscape as a 'cultural service'.

The work also aimed to examine the contribution that BBN methods could make to participatory impact assessment techniques. It did this by adapting and extending methods that were developed as part of the EU-funded SENSOR project, which aimed to develop a suite of sustainability impact assessment tools (Helming et al., 2011). One output of the SENSOR work was the so-called 'Framework for Participatory Impact Assessment' (FoPIA) (Morris et al., 2011), which is a set of methods designed to enable stakeholders to make assessment of national, regional and local sustainability priorities (Figure 3). FoPIA was used within the SENSOR Project to explore sustainability issues at local scales in Europe; the application reported by Morris et al. (2011) concerns biodiversity policy in Malta. Other applications include those of König (2010) and König et al. (2010).

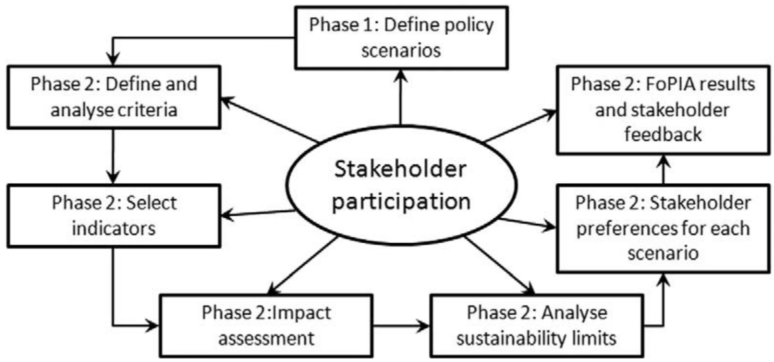

Figure 3. The Framework for Participatory Impact Assessment (FoPIA)
Source: After Morris et al. (2011)

FoPIA comprises two key phases: the definition of a set of target scenarios and the impact assessment itself. For the study in Malta reported by Morris et al. (2011), the scenarios were developed through semi-structured interviews with national- and regional-level policy-makers and experts and the examination of relevant national documentation; the scenario impacts were then examined through a stakeholder workshop. The methods used in these workshops are valuable to consider here, because they can also be implemented, captured and potentially extended using BBN techniques.

In the workshop phase of FoPIA, discussion between participants began by asking them to select criteria that can be used to assess sustainability impacts of policy scenarios using a set of indicators (known within SENSOR as land-use functions). To facilitate their choice, participants are asked to select from a predetermined set those that they consider most relevant, and then refine them so that they reflect local issues and conditions. In a subsequent, moderated discussion, the stakeholders are then asked to score the importance they assign to each of the criteria, score the magnitude of the impact (both positive and negative) that each scenario is likely to have on them, and specify what they would consider to be the limits of acceptable change in relation to each indicator. Specification of limits involved defining either a minimum standard that should be achieved for a given indicator, or a state below which stakeholders would not want the indicator to fall. During the facilitated discussions, individuals could modify their views and scores to incorporate a deliberative element into the process. Finally, stakeholders are asked to comment and provide feedback on the insights they gained from the exercise.

2 Case-study methods

The case study described here followed the main steps in the FoPIA approach. The exercise arose as part of the scoping work (CQuEL[8]) commissioned by Natural England to follow up *Countryside Quality Counts*. Two separate stakeholder events were organized, involving 12 and 15 people, respectively; the participants were a mixture of landscape professionals, policy advisors and academics. The workshops were not specifically designed to test methods of stakeholder deliberation based on BBNs, but they did provide an opportunity to explore their application and, in particular, how they could extend the Phase 2 component of the FoPIA methodology (Figure 3). The substantive focus of the meetings was to explore whether it was possible to distinguish between a set of national scenario outcomes using a small set of landscape indicators, and whether indicators defined at national scales can be used to explore and assess local trends. Use of the BBNs for the group exercise allowed a test of whether they could help to capture the diversity of opinions within the group and whether the graphical methods could help people review their initial thoughts on levels of acceptable change.

In the first part of each workshop, participants were asked to split into groups of 3–5 people and consider a set of published scenarios that had been developed by Natural England (Natural England, 2009a, 2009b) using an ethnographic futures framework. Four contrasting narratives (*Connect for Life*, *Go for Growth*, *Keep it Local* and *Succeed through Science*; see Table 1 for summary) were used by Natural England to explore interaction of people and society with the environment, and to investigate how changes in people's values, culture and behaviours might shape different futures (Creedy et al., 2009). In the workshops these narratives were used to examine how different futures might be tracked using indicators. The groups were asked to define indicators covering the themes 'people's engagement with the environment', 'landscape' and 'biodiversity'. Each group worked on one indicator, and at the end of their initial discussions they were asked to present the indicator to all the participants in the workshop. After each indicator was discussed, the FoPIA methodology was followed and participants were asked to: score each of the indicators for importance; estimate the magnitude and direction of change that they would expect compared to the present under each scenario; and specify limits of acceptable change. The impacts and limits were scored on a six-point scale, from –3 (large negative impacts) to +3 (large positive impacts), with zero being the mid-point; the importance assigned to each indicator was scored on a three-point scale: low importance, important, very important. The scores were collected on posters during the workshop, then captured on a spreadsheet and transferred to Netica for analysis.

3 Case-study results

The individual scores for importance, limits and impacts collected during the workshop were used to calibrate a BBN shown in Figure 4. The outcomes of the two workshops were similar, and so only one is described here. During the initial discussions in this workshop about the potential indicators, the measure suggested for people's engagement was the 'number of people affiliating with environmental causes'. The proposal for the landscape indicator was 'the legibility of place', measured in terms of the pattern and form of different landscape elements and their fragmentation. The suggestion for biodiversity was the abundance of an appropriate indicator species.

The BBN diagrams shown in Figures 4a, 4b and 4c should be read from the top downwards. The frequency histograms for the individual scores given to the measures for the indicators, their importance and their limits were used to assign the probabilities that each of the corresponding nodes were in a given state, according to stakeholder opinion. The three impact nodes then estimate automatically the probability that the indicator is above or below the defined limit according to the views of the participants about the likely impact, and their views on the levels of acceptable change.

The hexagonal nodes shown in the diagram calculate the 'utility' of the different outcomes for the indicators. These nodes measure how far the indicator is above or below the defined limit, weighted by the importance that the participants assign to that measure. In the network the utility value for each indicator is stored separately and then summed and presented in the node labelled overall value (see below). The inclusion of these nodes in the BBN extends the framework beyond that used in the manual FoPIA approach.

For the BBN shown in Figure 4a, all the scenarios are considered together (i.e. they are considered equally likely) and as a result the other nodes show the average scores from

Table 1. Summary of scenarios used as input into FoPIA-BBN workshop (after Creedy et al., 2009)

	Connect for Life	Go for Growth	Keep it Local	Succeed through Science
Scenario features	Initial focus on information and communication technologies, improve productivity, little attention to the capabilities of social networking. *Through extensive social networks traditional ways of doing things become outdated.*	Current trends continue, economic growth is a priority. *Lifestyles remain focused on consumption driven by accelerating innovation.*	Initial focus on consumption, little attention paid to resource and environmental limits. *Environmental and financial crises drive protectionism reducing globalization.*	Focus on short-term global productivity, little attention paid to long-term consequences for society and environment. *New market entrants gain competitive advantage through focus on innovation to safeguard long-term social and environmental capital.*
People's engagement with the natural environment	Large numbers of people engaging frequently with the environment, often enabled through, or enhanced with, high-definition virtual reality and immersive presence.	Decreasing active engagement. Few have the leisure time but, more generally, the natural environment is regarded as a resource for economic growth. Increasing view of the natural environment as a source of threat.	Local pride in biodiversity and iconic landscapes. Increased awareness of the direct benefits of the natural environment particularly for food, energy and water.	The natural environment is valued for the tangible benefits it can bring. Indirect benefits, including cultural and aesthetic considerations, are recognized, especially when a financial benefit can be obtained.
Biodiversity	Area of semi-natural habitat increased. High species abundance and functioning land and marine ecosystems.	The speed of the long-term decline in terrestrial and marine biodiversity is increasing. Islands of biodiversity in private estates. Increased pressure from invasive species and biotechnology.	Reduced area of semi-natural habitat. Iconic species and habitats protected in specific locations.	Biodiversity supporting ecosystem services is protected and enhanced. Technology used to avoid and reduce negative impacts. Natural systems increasingly managed but increased risk of unintended ecological consequences.
Landscape	Significant change to accommodate development. Sense of place important with emphasis on the protection of ecosystem services. A blurring of urban and rural landscapes.	Significant change due to large-scale energy, transport and adaptation infrastructure projects and more industrialized agricultural practice.	Significant change relating to land use and infrastructure for the production of energy, food and fibre. Iconic landscapes protected in specific locations.	Significant and rapid changes in land use as new technologies are adopted and replaced; in some locations, postindustrial legacies increase in number and extent.

participants across all four storylines. The probability histograms show that in general there was good agreement between workshop participants on the impact of scenarios on people's engagement; the mean and standard deviation of the scores are shown at the bottom of each node. Opinion was more widely spread, however, for landscape and biodiversity. Indeed for biodiversity the scores seem to indicate a bimodal distribution.

Figures 4b and 4c show the outcomes for the two most contrasting scenarios, *Connect for Life* and *Go for Growth*. Each is selected by setting the appropriate state of the scenario node to 100%. With *Connect for Life* (Figure 4b), stakeholders clearly felt that this would have very positive outcomes for people's engagement and biodiversity (more positive than the average for all scenarios taken together), but were more mixed in their views

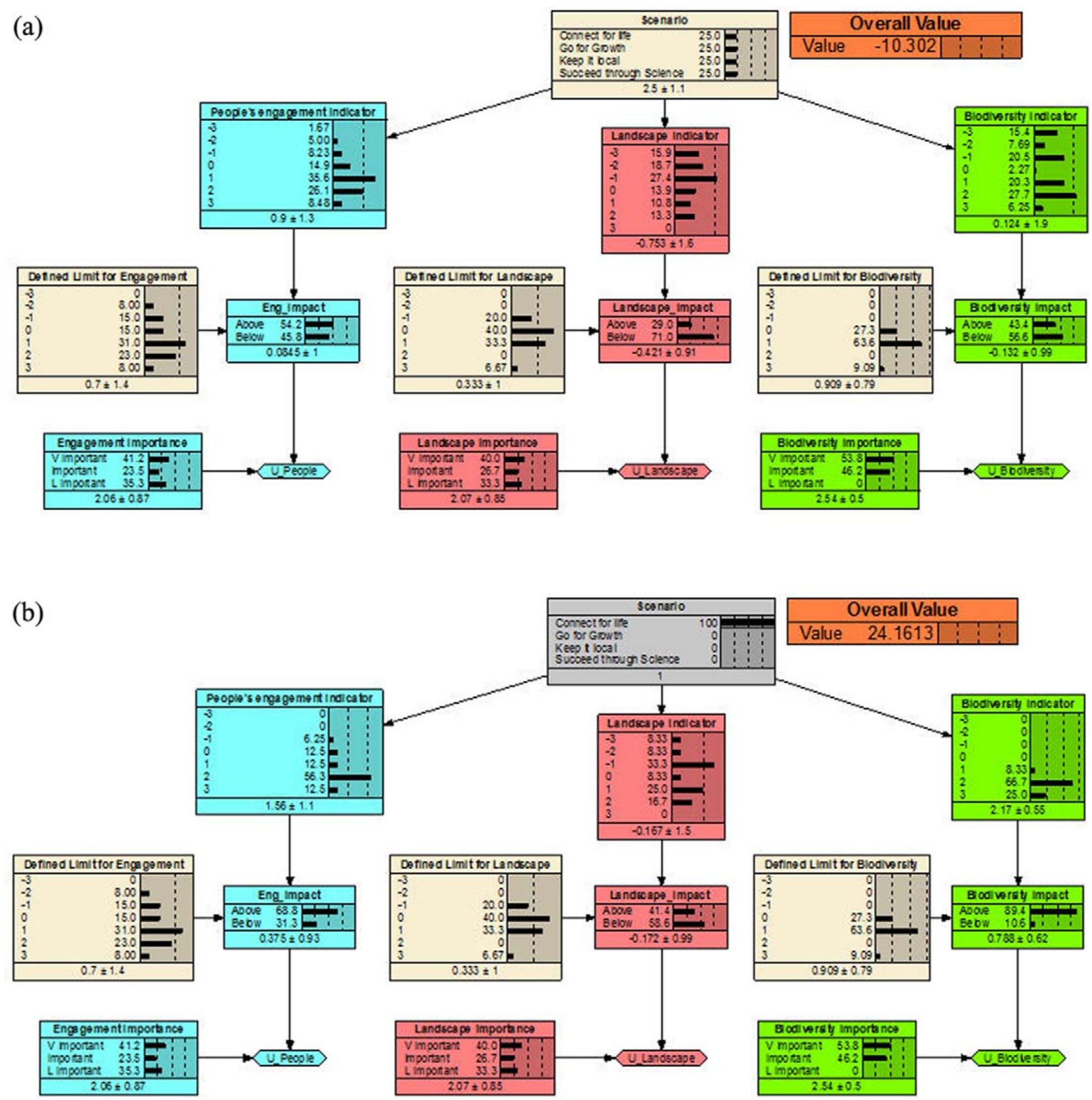

Figure 4. BBN used to implement Framework for Participatory Impact Assessment (FoPIA): (a) when all scenarios are equally likely; (b) when *Connect for Life* scenario is selected;

(c)

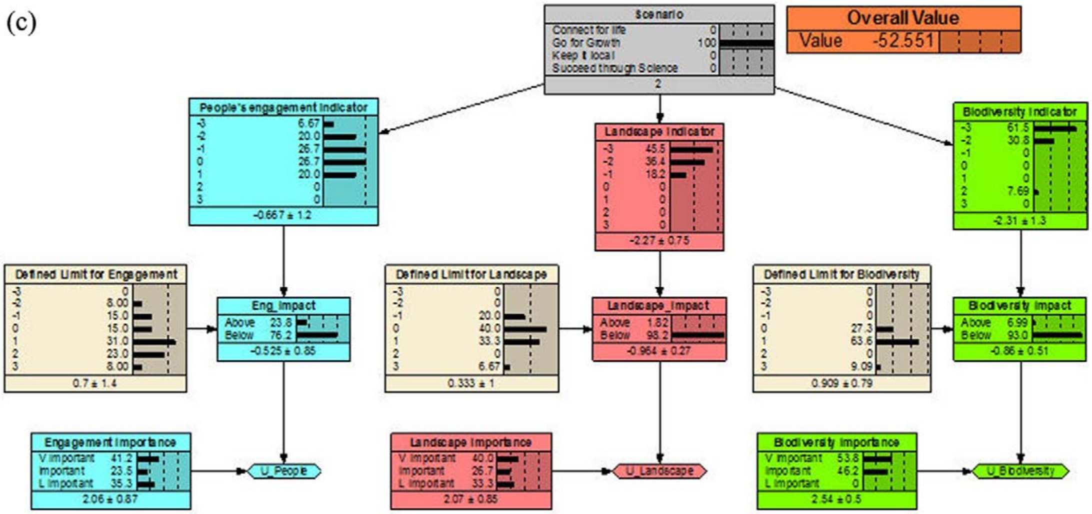

Figure 4. (Continued). (c) when *Go for Growth* scenario is selected

about its impact on landscape. In contrast, when the scenario *Go for Growth* is selected (Figure 4c) participants clearly felt that this would have detrimental effects on landscape and biodiversity and neutral or negative impacts on people's engagement.

For the networks shown in Figure 4, the act of selecting a particular scenario shows how the stakeholder opinions are differentiated across the four storylines, and how sensitive the outcomes are to the particular set of assumptions that define each scenario. Thus, when all the scenarios are taken together (Figure 4a), the chances of being above or below the limit of acceptable change for people's engagement and biodiversity are roughly equal, whereas landscape indicator is most likely to be below the limit set by the group. When *Connect for Life* is selected (Figure 4b), people's engagement and biodiversity are much more likely to be above the limits of acceptable change as defined by the stakeholders, and the landscape indicator is more evenly balanced than in Figure 4a. In contrast, when the *Go for Growth* scenario is selected (Figure 4c), there is a high probability that all the indicators will be below the acceptable minimum.

In the FoPIA methodology the analysis of stakeholder views and values stops at the point when the impact of scenarios is identified. The BBN exercise goes one step further and illustrates how utility scores can be assigned to the outcomes for each indicator and an overall valuation of the scenario outcomes made. The procedure for calculating the utility scores is shown in Table 2. For each combination of the defined importance and impact categories an estimate of utility is given. For the purposes of exploration using this network, a set of arbitrary numerical scores have been assigned; the measure ranks the beneficial outcomes for indicators that are deemed to be important to have larger positive scores than those that are of lower importance or which show negative impacts. In this example, the same scoring system was used for each of the utility nodes, and the values assigned were not modified by workshop participants, but the approach was discussed. Although the utility scores used in the example are arbitrary they could, however, be monetized and designed to express people's willingness to accept or pay for particular outcomes. In this way the utility functions could be different for each of the indicators and have

Table 2. Structure of utility function used to calculate impacts of scenarios on the indicator for people's engagement with nature. The score is a function of the magnitude of the impact and the importance assigned to the indicatory by stakeholders. All the utility functions shown in Figure 4 have the same structure. The utility scores are arbitrary, but could be replaced with monetary values for willingness to avoid or achieve a particular outcome.

States for input nodes		Assigned utility scores for output node
Engagement impact	Engagement importance	Utility value for engagement
Above	Very important	30
Above	Important	20
Above	Low importance	10
Below	Very important	–30
Below	Important	–20
Below	Low importance	–10

a different influence on the overall utility value. The utility scores thus provide the basis for a multicriteria assessment, that potentially can take account of any trade-offs between the indicator elements.

In Figures 4a, 4b and 4c the three utility scores are summed and the result is expressed using the node at the top of each network diagram. The score expresses the overall change in utility compared to the baseline (present conditions). Taking all the scenarios as a set, then the stakeholders felt that there was likely to be some loss of utility. With *Connect for Life*, however, the outcomes were likely to lead to a marked gain in utility, while *Go for Growth* showed the largest loss of the four scenarios.

4 Evaluation of case-study outcomes

The experience of using the BBN in the two stakeholder workshops showed that it was possible to employ the approach to capture and display the kinds of stakeholder-derived information collected in FoPIA. In both workshops the results were 'played back' to participants during the final feedback session and used to discuss the plausibility of the scenarios, their impacts and the values people assigned to them. Unfortunately, the time available in the workshops did not allow individuals to modify their predictions and values. Nevertheless, it is clear that the use of the BBN to capture their views and values was helpful in the context of a more general participatory method like FoPIA. Although both workshops used the same format and materials, the discussions and deliberations of the two groups were independent. Although the interpretation of the indicators by which impacts would be judged differed between groups, and while their scoring also differed, both came up with the same ranking of the most and least beneficial scenario projections. *Connect for Life* was considered significantly less damaging than *Go for Growth*; the remaining two were assessed to be quite similar and of intermediate impact. Moreover, both groups assigned greater importance to the impact of their biodiversity indicator than the metrics for people's engagement or landscape structure.

In terms of the primary aims of the CQuEL study, it was concluded that participatory techniques like FoPIA could be used to examine the impact of scenarios and how stakeholders valued different components of the cultural service of 'landscape'. In relation to exploring the use of a BBN to assist this deliberative process, it was concluded that it might support the process in a number of ways. For example, apart from capturing the data from the workshops so that the strengths of opinion within and between groups could be examined interactively by the researchers, it also had the virtue of preserving the variability of opinions within and between the groups. This was not easily done in the conventional application of the FoPIA methods where average scores were used in the assessment. Finally, in the construction of a utility function for each indicator, it is clear that BBNs can be used as a device that can capture something about the shared social values in the group; that is, both the average value and the variation about that average. Depending on the aims of the participatory exercise, the utility scores could be expressed on an arbitrary scale, simply to compare the scenarios and what are considered to be the marginal differences between them. Alternatively, they could be turned into willingness to pay for avoiding or achieving a particular impact score. In this particular exercise the impacts of the scenarios were assessed qualitatively according to the beliefs and facts held by stakeholders. The scenario impacts shown through the BBN could equally well have been model based, and more emphasis given to elicitation of the importance stakeholders gave to particular indicators and the limits of acceptable change. It is in this sense that BBN could help natural and social scientists bring 'facts and values in to more open play'.

V Conclusions: building analytical-deliberative frameworks

Kuhnert et al. (2010) have proposed a guide for obtaining and using expert knowledge in building Bayesian ecological models. They argue that expert knowledge can be employed with the same level of rigour as that applied in the collection of empirical data, providing a number of conditions are met in the design of the elicitation process. These include clearly articulating the research question and expertise available in the expert group, through to the methods used to structure the elicited information in such a way that it can be use to model the situation given the data and expertise available. They also recommend that feedback mechanisms and sensitivity tests be built into the elicitation and modelling process. Finally, the elicitation process has to capture the uncertainties that experts have about the system. Such guidelines could be extended and applied to work with non-expert groups. The value of their input into any participatory modelling process based on Bayesian methods would also clearly be enhanced by close attention to problem framing and knowledge representation methods, and the need to review outputs critically.

If the standards of practice suggested by Kuhnert et al. (2010) could be met, then the use of BBNs as a modelling tool would go some way to meeting the challenges identified by Fish (2011) and provide the basis for the analytical-deliberative approach he feels is needed to address the trans- and interdisciplinary analysis of ecosystem services. BBNs seem, for example, to offer the possibility of combining analytical rigour with interpretive complexity (Fish, 2011: challenge 6). They also provide opportunities for exploratory forms of investigation and representation that can help link across different knowledge domains (Fish, 2011: challenge 3). In this paper we have therefore argued that BBNs can be used as one way of unpacking the ecosystem service cascade that links biophysical structures and processes on the one hand, and human well-being and values on the other.

It must be acknowledged that the case studies presented here are still a long way from fully representing the structure and dynamics of a real social-ecological system. However, the examples demonstrate that by linking BBN approaches to other tools, particularly GIS, there is some scope for making progress in addressing the spatial and temporal complexities associated with ecosystem assessments. In reviewing the case studies, there is clearly a temptation to suggest that they need to be refined and deepened, and move them from tier 1 to tiers 2 and 3 by capturing greater degrees of realism and precision. While this may well be the case, it could also be argued that the characteristic of approximation embodied in these simple models is a virtue. An equally valuable goal might be to broaden such models rather than deepen them. In so doing we might then better exploit the ability of the 'language of ecosystem services' to generate new ideas and perspectives by engaging new communities of interest (Fish, 2011: challenge 4). Indeed, it could be argued that the 'limitation' of approximation associated with tier 1 models can be offset by an ambition to extract and combine detailed understandings from sector- or process-specific models and build an overarching meta-model. Bayesian Belief Networks are a useful framework in which these broader understandings can be built and the uncertainties surrounding the construction of production functions clearly laid out for decision-makers. These networks can therefore help us re-explore the traditional perspectives of Physical Geography and connect them to insights from other knowledge domains, so that we can better understand and manage the links between ecosystems services and human well-being.

Notes

1. In this paper 'interdisciplinarity' is used to be an integrated attempt at problem solving that involves experts from a number of research domains; 'transdisciplinarity' also provides integrated perspectives but includes lay-knowledge to frame questions and evaluate outcomes.
2. The BBN software used for the work was Netica V4.16 and ArcMap 10 was used for the spatial analysis.
3. See http://www.countrysidesurvey.org.uk/archiveCS2000/CIS_files.htm.
4. Note the proportions of 'sea' are low because only coastal cells were included in the data set.
5. Only the models for the low climate change impact versions of the scenarios are considered here; in the models discussed the influence of flooding (Node I) and sea-level rise (Node J) were set to 'none', due to problems of partial data coverage, so that effectively these operated as constant nodes.
6. To follow Milne and Brown (1997) the original definition of GB Land Classes is used here rather than the refined set used following Countryside Survey 2000.
7. The area of GB used in this analysis was larger than that used by Milne and Brown (1997) and so the estimate has been adjusted so that a comparison can be made on an equal area basis. The total vegetation carbon estimated for 2000, for an area of 235,980 km^2 using the BBN method was 128.3 Mtonnes; the area used by Milne and Brown (1997) was 224,001 km^2.
8. Character and Quality of England's Landscapes.

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References

Bateman I, Abson D, Andrews B, Crowe A, Darnell A, Dugdale, et al. (2011) Valuing changes in ecosystem services: Scenario analyses. In: *UK National Ecosystem Assessment, Technical Report*. Cambridge: UNEP-WCMC, Chapter 26.

Bunce RGH, Barr CJ, Clarke RT, Howard DC and Lane AMJ (1996) ITE Merlewood land classification of Great Britain. *Journal of Biogeography* 23(5): 625–634.

Burkhard B, Kroll F, Müller F and Windhorst W (2009) Landscapes' capacities to provide ecosystem services – a concept for land-cover based assessments. *Landscape Online* 15: 1–22.

Cain J (2001) *Planning Improvements in Natural Resources Management: Guidelines for Using Bayesian Networks* *to Support the Planning and Management of Development Programmes in the Water Sector and Beyond.* Wallingford: Centre for Ecology and Hydrology.

Cantarello E, Newton AC and Hill RA (2011) Potential effects of future land-use change on regional carbon stocks in the UK. *Environmental Science and Policy* 14(1): 40–52.

Chan KMA, Shaw MR, Cameron DR, Underwood EC and Daily GC (2006) Conservation planning for ecosystem services. *PLoS Biology* 4(11): 2138–2152.

Conte M, Nelson E, Carney K, Fissore C, Olwero N, Plantinga AJ, et al. (2010) Terrestrial carbon sequestration and storage. In: Karvera P, Tallis H, Ricketts TH, Daily GC and Polasky S (eds) *Natural Capital. Theory and Practice of Mapping Ecosystem Services*. Oxford: Oxford University Press, 111–128.

Creedy J, Doran H, Duffield S, George N and Kass G (2009) *England's Natural Environment in 2060 – Issues, Implications and Scenarios*. Sheffield: Natural England.

De Groot R, Fisher B, Christie M, Aronson J, Braat L, Gowdy J, et al. (2010) Integrating the ecological and economic dimensions in biodiversity and ecosystem service valuation. In Kumar P (ed.) *The Economics of Ecosystems and Biodiversity: Ecological and Economic Foundations.* London: Earthscan, 9–40.

Eaton JM, McGoff NM, Byrne KA, Leahy P and Kiely G (2008) Land cover change and soil organic carbon stocks in the Republic of Ireland 1851–2000. *Climatic Change* 91(3–4): 317–334.

Fish R (2011) Environmental decision making and an ecosystems approach: Some challenges from the perspective of social science. *Progress in Physical Geography* 35(5): 671–680.

Fisher B, Turner RK, Burgess ND, Swetnam RD, Green J, Green R, et al. (2011) Measuring, modeling and mapping ecosystem services in the Eastern Arc Mountains of Tanzania. *Progress in Physical Geography* 35(5): 595–611.

Haines-Young RH and Potschin M (2010) The links between biodiversity, ecosystem services and human well-being. In Raffaelli D and Frid C (eds) *Ecosystem Ecology: A New Synthesis*. BES Ecological Reviews Series. Cambridge: Cambridge University Press, 110–139.

Haines-Young RH, Paterson J and Potschin M (2011) The UK NEA scenarios: Development of storylines and analysis of outcomes. In: *UK National Ecosystem Assessment, Technical Report*. Cambridge: UNEP-WCMC, Chapter 25.

Helming K, Diehl K, Bach H, Dilly O, König B, Kuhlman T, et al. (2011) Ex ante impact assessment of policies affecting land use, Part B: Application of the analytical framework. *Ecology and Society* 16(1). Available at: http://www.ecologyandsociety.org/articles/3839.html.

Imhoff ML, Bounoua L, Ricketts T, Loucks C, Harriss R and Lawrence WT (2004) Global patterns in human consumption of net primary production. *Nature* 429(6994): 870–873.

Karvera P, Tallis H, Ricketts TH, Daily GC and Polasky S (eds) (2011) *Natural Capital: Theory and Practice of Mapping Ecosystem Services*. Oxford: Oxford University Press.

Kates RW, Clark WC, Corell R, Hall JM, Jaeger CC, Lowe I, et al. (2000) Sustainability science. *Science* 292(5517): 641–642.

Kienast F, Bolliger J, Potschin M, de Groot RS, Verburg PH, Heller I, et al. (2009) Assessing landscape functions with broad-scale environmental data: Insights gained from a prototype development for Europe. *Environmental Management* 44: 1099–1120.

Kocabas V and Dragicevic S (2007) Enhancing a GIS cellular automata model of land use change: Bayesian networks, influence diagrams and causality. *Transactions in GIS* 11(5): 681–702.

König HJ (2010) Multifunctional forest management in Guyuan: Potentials, challenges and trade-offs. *Journal of Resources and Ecology* 1(4): 300–310.

König HJ, Schuler J, Suarma U, McNeill D, Imbernon J, Damayanti F, et al. (2010) Assessing the impact of land use policy on urban-rural sustainability using the FoPIA approach in Yogyakarta, Indonesia. *Sustainability* 2: 1991–2009.

Kuhnert PM, Martin TG and Griffiths SP (2010) A guide to eliciting and using expert knowledge in Bayesian ecological models. *Ecology Letters* 13(7): 900–914.

Millennium Ecosystem Assessment (MA) (2005) *Ecosystems and Human Well-Being*: *Synthesis*. Washington, DC: Island Press.

Milne R and Brown TA (1997) Carbon in the vegetation and soils of Great Britain. *Journal of Environmental Management* 49(4): 413–433.

Morris JB, Tassone V, de Groot R, Camilleri M and Moncada S (2011) A framework for participatory impact assessment: Involving stakeholders in European policy making, A case study of land use change in Malta. *Ecology and Society* 16(1). Available at: http://www.ecologyandsociety.org/vol16/iss1/art12.

Naidoo R, Balmford A, Costanza R, Fisher B, Green RE, Lehner B, et al. (2008) Global mapping of ecosystem services and conservation priorities. *PNAS* 105(28): 9495–9500.

Natural England (2009a) Global drivers of change to 2060. Natural England Commissioned Report, NECR030.

Natural England (2009b) *Scenarios compendium.* Natural England Commissioned Report, NECR031.

Newton AC, Stewart GB, Diaz A, Golicher D and Pullin AS (2007) Bayesian Belief Networks as a tool for evidence-based conservation management. *Journal for Nature Conservation* 15(2): 144–160.

Potschin MB and Haines-Young RH (2006) Rio+10, sustainability science and Landscape Ecology. *Landscape and Urban Planning* 75(3–4): 162–174.

Potschin MB and Haines-Young RH (2011): Ecosystem services: Exploring a geographical perspective. *Progress in Physical Geography* 35(5): 575–594.

Schröter D, Cramer W, Leemans R, Prentice IC, Araújo MB, Amell NW, et al. (2005) Ecology: Ecosystem service supply and vulnerability to global change in Europe. *Science* 310(5752): 1333–1337.

Smith CS, Howes AL, Price B and McAlpine CA (2007) Using a Bayesian belief network to predict suitable habitat of an endangered mammal – The Julia Creek dunnart (*Sminthopsis douglasi*). *Biological Conservation* 139(3–4): 333–347.

Smith RI, Dick JM and Scott EM (2011) The role of statistics in the analysis of ecosystem services. *Environmetrics* 22(5): 608–617.

Stelzenmüller V, Lee J, Garnacho E and Rogers SI (2010) Assessment of a Bayesian Belief Network-GIS framework as a practical tool to support marine planning. *Marine Pollution Bulletin* 60(10): 1743–1754.

Sutherland WJ, Adams WM, Aronson, RB, Aveling R, Blackburn TM, Broad S, et al. (2009) One hundred questions of importance to the conservation of global biological diversity. *Conservation Biology* 23(3): 557–567.

Swetnam RD, Fisher B, Mbilinyi BP, Munishi PKT, Willcock S, Ricketts T, et al. (2010) Mapping socio-economic scenarios of land cover change: A GIS method to enable ecosystem service modelling. *Journal of Environmental Management* 92(3): 563–574.

Tallis H and Polasky S (2011a) Assessing multiple ecosystem services: An integrated tool for the real world. In: Karvera P, Tallis H, Ricketts TH, Daily GC, and Polasky S (eds) *Natural Capital: Theory and Practice of Mapping Ecosystem Services*. Oxford: Oxford University Press, 34–52.

Tallis H and Polasky S (2011b) How much information do we need? The sensitivity of ecosystem service decisions to model complexity. In: Karvera P, Tallis H, Ricketts TH, Daily GC and Polasky S (eds) *Natural Capital: Theory and Practice of Mapping Ecosystem Services*. Oxford: Oxford University Press, 264–277.

UK National Ecosystem Assessment (NEA) (2011) *The UK National Ecosystem Assessment: Synthesis of the Key Findings*. Cambridge: UNEP-WCMC.

Zorrilla P, Carmona G and De la Hera, A (2010) Evaluation of Bayesian networks in participatory water resources management, upper Guadiana Basin, Spain. *Ecology and Society* 15(3). Available at: http://www.ecologyandsociety.org/vol15/iss3/art12.