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Operationalizing an ecosystem services-based approach using Bayesian Belief Networks: an application to riparian buffer strips

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Abstract

The interface between terrestrial and aquatic ecosystems contributes to the provision of key ecosystem services including improved water quality and reduced flood risk. We develop an ecological-economic model using a Bayesian Belief Network (BBN) to assess and value the delivery of ecosystem services from riparian buffer strips. By capturing the interactions underlying ecosystem processes and the delivery of services we aim to further the operationalization of ecosystem service approaches. The model is developed through outlining the underlying ecological processes which deliver ecosystem services. Alternative management options and regional locations are used for sensitivity analysis.

We identify optimal management options but reveal relatively small differences between impacts of different management options. We discuss key issues raised as a result of the probabilistic nature of the BBN model. Uncertainty over outcomes has implications for the approach to valuation particularly where preferences might exhibit non-linearities or thresholds. The interaction between probabilistic outcomes and the statistical nature of valuation estimates suggests the need for further exploration of sensitivity in such models. Although the BBN is a promising participatory decision support tool, there remains a need to understand the trade-off between realism, precision and the benefits of developing joint understanding of the decision context.

Keywords: Bayesian Networks; Ecosystem services; Interdisciplinary research; Valuation

1 Introduction

Recent years have seen the growing adoption of ecosystem services-based approaches for analysis and decision-making with respect to the environment. This approach has also encouraged the development of a common language across natural and social science disciplines that in turn has led to joint analysis and assessments. Notable examples of the latter include the Millennium Ecosystem Assessment (MA, 2005) and the UK's National Ecosystem Assessment (UK NEA, 2011). However, the increasing prevalence of interdisciplinary analysis has highlighted the need to further develop common models and tools to explore our joint understanding of ecosystem services that might better inform management and policy (Martin-Ortega et al., 2015). This is the key issue in the operationalization of ecosystem services as an analytical and decision making approach. To this end there have been some targeted attempts to foster interdisciplinary working, such as the UK's Valuing Nature Network[1], which specifically seeks to promote research capacity on the integration of approaches to the valuation of ecosystem services to support policy and practice.

The complexities and interdependencies among components within and between ecosystems make describing and quantifying interactions within and across ecosystems a considerable challenge (Heal et al., 2001; Pereira et al. 2005; Carpenter et al., 2009; Maskell et al., 2013). Multiple ecological mechanisms interact within ecosystems resulting in the delivery of single or multiple services; or a single mechanism may contribute to multiple ecosystem services. The provision of ecosystem services may also be dependent on the contributions of many different ecosystems (Defra, 2007), for example good water quality arises from both terrestrial and aquatic ecosystems. Hence, policy decisions affecting any part of those interactions can cause changes across multiple services and ecosystems. Given this complexity, from an economic perspective the value of any ecosystem service may then be determined by its relationship with other services (UK NEA, 2011).

NRC (2005) reviewed studies attempting to integrate ecological and economic knowledge to value either single or multiple ecosystem services, concluding that our inability to estimate the '*true*' value of ecosystem

[1] The VNN is a UK Natural Environment Research Council funded initiative aimed at bringing together natural and social scientists, economists, policy-makers and business interests. http://www.valuing-nature.net/

services is mainly associated with three factors: i) lack of ecological understanding of how ecosystem services are being affected by alternative management practices, ii) inadequacy of the existing economic techniques to quantify the '*true*' value of multiple ecosystem services, and iii) inability to integrate ecological and economic knowledge. In order to tackle the methodological challenges of valuing ecosystem services, there is a growing consensus that integrated studies should be undertaken, which account for the interactions and non-linear relationships among ecosystem components (Carpenter et al., 2009; Kremen and Ostfeld, 2005; Tallis and Kareiva, 2005; Turner et al., 2003). Many authors suggest that it is necessary to develop a more holistic (Turner and Daily, 2008), interdisciplinary valuation approach that integrates economic and ecological knowledge (Brauman et al., 2007; Hein et al., 2006; O'Riordan et al., 2002; Pagiola et al., 2004). In other words, there is need for an approach that could quantify the economic value of the 'ecosystem service cascade' proposed by Haines-Young and Potschin (2009), integrating the underlying linkages between services and processes to provide a more accurate estimate of the ecosystem value.

A common problem with developing interdisciplinary models and tools has been to integrate different scientific and social science disciplines that operate at varying degrees of complexity. Biophysical science approaches to ecosystems operate over a wide range of scales and complexities including very context specific field studies (Norton et al., 2012a). Socio-economic approaches, such as non-market valuation, are often broad-brushed to avoid overburdening survey respondents, whose values we seek, with complex information. Relevant economic data are also often only available at large scales (e.g. national or regional). Neither of these scales may match policy or decision-making. Consequently, there is a potential mismatch of complexity and scales in the use of extant models and data. In order to operationalize an ecosystem services-based approach researchers and decision makers may need to develop joint models where we explicitly *sacrifice* precision in disciplinary approaches to achieve outcomes that are still of use to decision making.

In this paper we present an interdisciplinary approach based on Bayesian Belief Networks (BBN) in the hope of provoking discussion and debate about the virtues and limitations of BBNs as a tool to address some of the integration challenges. The benefit of using BBNs in natural resource management is their usefulness for predicting the links between management practices and ecosystem reactions (Clark et al., 2001; Borsuk et al., 2004), whilst they can also deal with a large number of interconnected data and integrate different types of variables (e.g. environmental, economic, social and physical variables) or knowledge from diverse sources (Bromley et al., 2005). In fact, BBNs have been widely applied in environmental studies including fisheries assessment (Kuikka et al., 1999; Lee and Rieman, 1997; Pollino et al., 2007); forest restoration (Haas et al., 1994); climate change problems (Gu et al., 1996; Kuikka and Varis, 1997); habitat restoration (Rieman et al., 2001); watershed management (Hamilton et al. 2007; Ames et al., 2005; Borsuk et al., 2004; Bromley et al., 2005; Henriksen et al., 2004) and nitrogen pollution impacts on wetland ecosystem services (Spence and Jordan, 2013). The review by Landuyt et al. (2013) indicates the excellent conceptual fit between the structure of BBN's and the ecosystem service production cascade (Haines-Young and Potschin, 2009), but alludes to limited attempts in the literature to exploit the potential of BBN's for elucidating the cascade in particular cases of ecosystem service delivery. Haines-Young (2011) uses two case studies from the UK NEA to explore how BBNs could be used to operationalize different components of the cascade model. This paper seeks to develop this approach by explicitly analysing the effects of one management mechanism (riparian buffer strips) on the delivery of ecosystem services (in the UK NEA example used by Haines-Young, different land cover scenarios are explored but not linked to management mechanisms). Landuyt et al. (2013) note, that BBNs have particular value because of the capacity for using them to consider the delivery of multiple ecosystem services whilst allowing the integration of multidisciplinary knowledge. However, they conclude that the integration of decision nodes and valuation into Bayesian networks remains an important challenge; this paper attempts to address that challenge.

The BBN was developed through a series of workshops under the Valuing Nature Network involving natural and economic scientists interested in identifying approaches for valuing the provision of ecosystem services across agricultural and aquatic ecosystems. The choice to focus on water quality and flood risk was based on workshop discussions around these two high profile services which are a focus of policy with respect to the European Water Framework Directive and Floods Directive. Buffer strips were identified as a relevant management instrument, widely employed through various agri-environment schemes for precisely the delivery of those services (Doody et al., 2012; Haygarth et al., 2009), and used here as a test case. We recognise that buffer strips offer a far wider range of services (Stutter et al., 2012) but in recognition of the potential complexity of valuing all these services, we have focused on the water services only. In the following section we discuss the issue of complexity and interactions in ecosystem service analysis and subsequent economic valuation in the context of the approach adopted. We then outline our approach before describing its specific application to riparian buffer strips. Finally we discuss outputs from this model and its further potential development.

2 Ecosystem service valuation – complexity, interactions and scale

As Boyd and Banzhaf (2007) argue, there should be a clear distinction between the 'final ecosystem services' that are directly consumed by individuals and the 'intermediate ecosystem functions' or processes that contribute to their delivery. Ecological processes are considered the intermediate biological, physical and chemical interactions between ecosystem services, rather than end-products. For instance, nutrient cycling and water flow are ecological functions which interact to deliver the service of water quality alongside other ecosystem services. Haines-Young and Potschin (2009) use the idea of a 'service cascade' to illustrate the mechanisms that underpin the connections between ecological assets and welfare, and the series of intermediate stages in which they are linked (Figure 1). This service cascade serves as the basic template for building the BBN in this study.

FIGURE 1 HERE

In the context of environmental valuation, the classification of ecosystem services into 'intermediate processes', 'final services' and 'benefits' addresses the problem of 'double counting' the values of ecosystem services (Boyd and Banzhaf, 2007; Fisher and Turner, 2008; Fisher et al., 2009; Fu et al., 2011; Ojea et al., 2012). For instance, in the case of a wetland, the intermediate functions of nutrient cycling and water regulation interact to deliver clean water. The actual benefit that humans derive from water provision may include recreation (e.g. angling, swimming, seeing water in the context of a landscape (Norton et al., 2012b)) or potable water (Fisher et al., 2009). Although it seems sensible to value the consumed products (tangible or intangible), the ability to acknowledge and measure the extent to which the processes underlying their delivery contribute to the final value of benefits is vital. Only in this way, can policy decisions affecting environmental management be valued for their impact on ecosystem services and ultimately the delivery of ecosystem benefits. It is therefore important that integrated models reflect relationships between final services, underlying processes and generated benefits.

In general, ecosystem service valuation tends to focus on one service at a time (Turner et al., 2003), disregarding interactions between ecosystem functioning and services. This is in part influenced by the difficulties faced by ecosystem science in considering multiple ecosystem service delivery, although it is acknowledged that such an approach is essential for the sustainable management of natural systems (NRC, 2005; Diaz and Rosenberg 2008; Gordon et al., 2008). In addition, the available approaches to undertake economic valuation of ecosystem services may themselves be inadequate for encompassing the complexities of natural systems. Valuation approaches vary in the extent to which they directly value individual or combinations of ecosystem services. Stated preference studies, either by virtue of the constructed valuation scenario or the good being valued (e.g. public goods and/or cultural services such as landscape), can be more closely linked to final ecosystem services than revealed preference, market value or cost based approaches (Barkmann et al., 2008). Marketed goods, such as food, require inputs of man-made and human capital (e.g. manufactured inputs, labour and knowledge) so the contribution of final ecosystem services to the goods that generate human welfare is less clearly identifiable (Bateman et al., 2011). These issues require care in the interpretation and use of estimated values. Therefore, benefit estimates derived via stated preference valuations are likely to be of use in the context of developing integrated models mirroring the ecosystem service cascade.

Müller et al. (2010) stress the need for an approach which integrates multiple ecosystem services (i.e. does not focus only on a single service or a limited set of services). Ecosystem services-based approaches would incorporate the interrelationships between ecological processes across the components of the ecosystem service cascade; the different spatial and temporal scales; and incorporate stakeholders into the decision making process (Hein et al., 2006; Martin-Ortega et al., 2015). Conceptually, BBNs seem to be particularly well fitted to address these challenges; they can be designed to fit particular study contexts and hence consider spatial and temporal scales (albeit with difficulty), and can be participatory through including stakeholders in the BBN development. Alternatively, BBNs may be constructed to investigate alternative management scenarios for generic ecosystems as opposed to ecosystem conditions at a particular location, i.e. they may be used as a tool to investigate the general effectiveness of policy interventions. This study considers the latter.

3 Developing an integrated ecological-economic model

Our interdisciplinary team of terrestrial and aquatic ecologists, soil scientists and economists held three workshops. Figure 2 shows the sequence of interdisciplinary workshops that took place during the development of the BBN model. The first workshop included a broader group of science and policy stakeholders, who together with the research team produced very complex mappings of ecosystem process and service linkages for services in agricultural and freshwater systems. This served to highlight the complexity of the issues rather than provide a potential approach.

We therefore held a smaller second workshop which focused on the specific management intervention of riparian buffer strips on agricultural land. Buffer strips provide an excellent subject for study in this context because they play an important role in interactions between agricultural land and freshwater ecosystems and whilst they are used as a policy instrument, many of the policies that directly affect buffer strips are conceived of and applied independently (Stutter et al., 2012). The second workshop specifically explored the use of a BBN approach to model the interactions between improving water quality and mitigating flood risk as two ecosystem services produced by riparian buffer strips, leading to benefits that might be valued. The aim of the BBN was to explore the effectiveness of different types of riparian buffer strip management at a regional scale with alternative scenarios relevant to the East and West of England offering contrasting climatic, topographic and land use conditions. A final workshop was held to review the BBN model and explore how it could be further developed to integrate the valuation component and to include a wider range of socio-economic drivers.

FIGURE 2 HERE

Bayesian Belief Networks (BBNs) represent interactions between a range of variables, which may include uncertain quantities as a directed acyclic graph which is formed by a series of interconnected nodes that link actions to outcomes (Barton et al., 2008; Pollino et al., 2007; Borsuk et al., 2004). The nodes represent the variables of the system, whilst the linkages among them indicate direct causal dependencies (Pollino et al., 2007); as they are acyclic these cannot form a closed loop (Bromley et al., 2005). Those nodes that do not have any conditional dependencies are called 'parent' nodes and represent input variables, whilst those that are conditionally dependent on at least one other are called 'child' nodes. Nodes without child nodes constitute the output of the system.

The strengths of the causal relationships among the system variables are quantified by conditional probabilities. These are defined by a set of conditional probability tables (CPTs) that specify the probability of each variable having a particular 'state' considering every possible combination of states of the parent nodes linked to it (Kjærulff and Madsen, 2005; Kragt, 2009; Pollino et al., 2007; Bromley et al., 2005). The state of the parent nodes is determined by a marginal (or unconditional) distribution of probabilities (Pollino et al., 2007; Borsuk et al., 2004) set by the operator. Variables can be determined either as discrete or continuous (Cain, 2001); with the state of each described by either a numerical value, a verbal description, or even a true or false statement (Bromley et al., 2005). The probability values can be either observed data, information elicited from experts or a combination of sources (Pollino et al., 2007).

3.1 Riparian buffer strips

Riparian buffer strips are vegetated strips of land that extend along the side of a watercourse which are set aside from production by farmers, often under agri-environment agreement (Stutter et al., 2012). Buffer strips are primarily encouraged in order to exclude nutrients, sediment and other organic matter from the watercourse (Ramilan et al., 2010), but may also play important roles in flood control, water retention and infiltration, climate regulation, habitat provision, recreation and amenity (Tabachi et al., 2000; NRC, 2002; Dwire and Lowrance, 2006; Soman et al., 2007). It is recognised that there is a range of interdependencies associated with the provision of the ecosystem services outlined above. For instance, decreases in the infiltration capacity of any riparian area will affect both productive capacity and water quality through decreasing nutrient uptake by plant roots, decreasing water storage and increasing surface runoff, thereby impacting on flood risk, recreational activities, water supply, etc.

The use of riparian vegetation as buffer strips was examined from a perspective of alternative management practices, i.e. a) grassland; b) natural vegetation; c) mixed (i.e. a and b); or d) no buffer strip. The impacts of these characteristics of buffer strips are documented in the literature (Siameti, 2012); further characteristics such as width and vegetation height will modify impacts but we assume these are implicit in the management of each buffer strip type. The functions provided by riparian buffer strips were incorporated into their effects on a) runoff rate, b) sedimentation load and c) water temperature. Effects of alternative land uses (i.e. arable or pasture), soil type, slope, as well as seasonal effects on water temperature and aquatic vegetation were also taken into consideration.

3.2 BBN construction

The initial stage in the development of a BBN was to construct a conceptual model specifying the cause-and-effect relationships among the system components. This process began during our second workshop. The conceptual model formed the basis for the directed acyclic graph. Firstly, the objectives (output nodes) of the model were defined; in this case: flood risk and water quality. The output nodes represent the 'physical' outcomes of the model (services) and are distinct from 'value' outcomes (benefits) which are captured in further utility nodes. We define the output nodes for the BBN as follows:

Flood risk: riparian buffer strips contribute to moderating flood risk either by delaying the passage of floodwater downstream or reducing surface runoff through infiltration or interception of precipitation..

Water quality: riparian buffer strips may enhance water quality through a number of processes. These include; direct interception of nutrient containing sediments, interception and infiltration of water, shading of the watercourse and nutrient cycling within the vegetation. The net effect of such processes is to reduce the nutrients reaching the associated water and reduce temperatures.

Once the output nodes and the policy tool (node 'buffer strips") were defined, development of the BBN drew on system variables and their interrelationships, as identified in our first and second workshops exploring the ecological processes involved in provision of water quality and flood risk specifically relating to farmland (summarised in Table 1). Given that the lower number of nodes a model has, the more easily understood it will be by the involved parties (Cain, 2001; Marcot et al., 2006) the challenge was to select the variables which would provide a realistic representation of terrestrial and aquatic ecosystems whilst at the same time keeping the model as simple as possible. The variables that were agreed during the second and third workshops for use in the model can broadly be divided into four groups: states of nature, terrestrial processes, management intervention and aquatic processes. The states of nature variables represent the local conditions which determine the variables of the terrestrial and aquatic processes, which together with the 'management intervention' variables indirectly or directly determine the final ecosystem services, flood risk and water quality. The individual variables have been defined and assessed for their dependencies in the scope of this study. The definitions and the results of the assessments are summarised in Table 1. In addition the table includes the assumptions that are used in the parameterization process.

TABLE 1 HERE

Flood risk was modelled as a variable determined by the level of river flow. It is affected indirectly by the surface runoff rate, the rainfall rate and aquatic vegetation. This is a simplification of a complex system where river flow is not the sole determinant of flood risk but it reflects our focus on a small number of key processes. Water quality can be defined by a range of biological, chemical, hydrological and morphological characteristics, such as levels of dissolved oxygen, pH, temperature, soluble nutrient content, and fish populations (UK NEA, 2011). In this study, Biological Oxygen Demand (BOD) was selected as the water quality indicator because of its importance as an indicator of biological quality and the availability of evidence related to factors impacting upon it. Water temperature, water nutrient concentration and aquatic vegetation coverage are considered to have an indirect impact on water quality through their effect on BOD, although these factors in themselves can also directly impact on water quality.

The BBN was created using Netica software (Norsys Software Corp, 2003) and was further developed to include decision, nature and utility nodes. Decision nodes are associated with factors controlled by decision makers, whilst utility nodes represent those variables that need to be optimised (i.e. system outputs). Thus, 'riparian buffer strips' was depicted as a decision node, whilst the end-points of the system were connected to a utility node, 'satisfaction'. We use the term 'satisfaction' due to its link to the economic concept of utility and also because it is not linked to any specific unit or estimate of value within the current model. The values for all the other variables were dependent on probability relationships with other variables, expressed as conditional probability distributions, and were drawn as nature nodes. Our BBN model is illustrated in Figure 3.

FIGURE 3 HERE

4 Model parameterisation

Once the conceptual network was designed, the next step was to populate each CPT with probability values. Since the model is generic rather than site-based, the parameterisation process was based on evaluations of the general patterns of riparian ecosystem functioning relevant to buffer strips, drawn from the literature and from expert knowledge (see Table 1 assumptions).

All the system components were identified as discrete variables; these were chosen to simplify parameterisation in the absence of data to populate continuous variables. Decision and parent nodes are deterministic with their states provided by decision makers (Castelletti and Soncini-Sessa, 2007; Cain, 2001); hence, these nodes did not need to be populated in the same way. . The generic probabilities used in this model were intended to reflect contrasts between the different states of the variables (e.g. low, medium, high) rather than absolute values. The use of observed data might lead to more robust results, but as emphasised previously would limit the potential to derive general policy recommendations for alternative scenarios. We argue that the benefit of the BBN approach in this context lies in developing an understanding of processes and their interactions as part of a decision support tool. The CPT for Overland flow is presented in Table 2 as an example of our approach.

TABLE 2 HERE

As we were unaware of any joint valuations of flood risk and water quality, the values used to parameterise satisfaction were developed by the authors. This was treated as a continuous variable ranging from 0 to

100; effectively this was an index of the benefits associated with different combinations of states for the flood risk and water quality outcomes: low flood risk and high water quality = 100; high flood risk and poor water quality = 0, other combinations were assigned values in between; these are presented in Table 3. Although the utility values presented in Table 3 appear to be discrete values, the utility node itself must be defined as continuous to allow compilation of the network and subsequent estimation of the probability weighted utilities associated with different management actions in the decision node. Between the upper and lower bounds of high water quality/low flood risk and poor water quality/high flood risk there is an inherent trade-off between water quality and flood risk where the benefit of improving one of these can potentially result in a worse outcome for the other. In determining the values for 'satisfaction' we made the assumption that regardless of water quality status the overall score could not exceed 50 if flood risk was high; utility lies between 35 and 65 for medium flood risk; and where flood risk is low utility will always be greater than 50.

To parameterise the CPT states for water quality, we drew on the water quality ladder first introduced by Carson and Mitchell (1993) that describes water quality on an ascending scale of water-use possibilities. The worst quality category is associated with severe limitations on use, whilst improving water quality allows for a range of activities, such as, for example, boating and swimming. Different forms of the water quality ladder inspired by this original one have been extensively used in the water valuation literature (see Baker et al., 2007; del-Saz-Salazar et al., 2009; Brouwer et al., 2010; Glenk et al., 2011; Ramajo-Hernandez and del-Saz-Salazar, 2012; Metcalfe et al., 2012; Schaafsma et al., 2012). Maybe the most advanced of these, is that by Hime et al. (2009), who produced a generic water quality ladder built on various indicators of water quality levels, including; fish life, aquatic vegetation, river bank vegetation, substrate composition and water clarity. This relatively sophisticated ladder has been tested in several European countries (Bateman et. al 2011) and is the one used in this study. Each of the ecological categories is associated with different water quality levels, which Hime et al. (2009) define as blue, green, yellow, and red respectively (from the highest to the lowest quality). Each level of water quality was further linked to the defined states of BOD as described in Table 1.

We assume that there is less sensitivity to water quality state with no distinction made between the utility for the blue and green levels (this reflect the role of inherent characteristics such as substrate type in differentiating these levels which might not be affected by riparian management); so the BBN will in effect only reflect the utility associated with changes in the probability of water quality being either poor (red), moderate (yellow) or good (green and blue).

Once all CPTs were populated with probability values the model was compiled and the decision network 'solved'. That means that the software performed standard belief updating and calculated the 'marginal posterior probability' for each variable (Marcot et al., 2006), showing the 'optimal solution' of the problem. The inclusion of both decision (management actions) and utility nodes means that when the model is 'solved' the utility values associated with each management action are obtained thus allowing the optimal action to be identified.

TABLE 3 HERE

For each combination of land use and buffer strip management a utility score is calculated as the sum of the utility values associated with each combination of flood risk and water quality outcome (i.e. Table 3) multiplied by the probabilities of those outcomes occurring:

$$U_m = \sum_{s=1}^{S} PrFR_{ms} \times PrWQ_{ms} \times U_s \qquad (1)$$

Where U_m is the utility associated with management option m; $PrFR_{ms}$ is the probability of flood risk outcome s occurring under management option m; $PrWQ_{ms}$ is the probability of water quality outcome s occurring under option m; and U_s is the utility associated with combined flood risk and water quality outcomes s.

4.1 Model scenarios

The BBN was used to explore the effectiveness of the management intervention at regional scales. The model was able to explore all possible combinations of our 'states of nature' based on the parent nodes: region (2 states), slope (3 states), season (4 states), land cover (3 states) and soil type (3 states); this would give $2^1 \times 3^3 \times 4^1 = 72$ possible combinations, although some may be unlikely given the general geographical characteristics of the two regions. For brevity in this paper we evaluate a sub-set of three scenarios defined using typical combinations of region, land-use, soil type and slope (Table 4). These three scenarios were examined under alternative buffer strip management practices with ‘no buffer strips’ being referred as the ‘status quo’, in which it is assumed that vegetation in the riparian zone is managed for agricultural production whether grassland or arable such that the ecosystem processes associated with buffer strips are diminished. In particular the runoff rate and sedimentation load associated with these land uses are unmodified in the absence of buffer strips. The different buffer strip options ‘no buffer strips’, ‘grassland’, ‘natural vegetation’ and ‘mixed’ can be simultaneously evaluated, i.e. the BBN returns the utility values for all four. For each given ‘state of nature’ scenario, our aim was to: (i) identify the optimal buffer strip management practice; and (ii) compare how the system objectives changed between the ‘status quo’ and the ‘optimal solution’. The BBN can also take seasonal changes (associated with the rainfall rate, vegetation coverage and temperature) into account, however for the examples we present in the results specific seasons are not selected which means they represent year-round or average seasonal conditions. From a decision support perspective this signifies an evaluation of buffer strip performance throughout the year.

TABLE 4 HERE

5 Results

Table 5 presents the utility or satisfaction values associated with each of the scenarios for the different buffer strip management options and Table 6 shows the changes in the probabilities of the management objectives occurring under each of these options. In scenario A, where there is a low level of overland flow (i.e. East England: low rainfall; light soils with high infiltration capacity; low slope), natural vegetation proved to be the optimal buffer zone management practice (satisfaction score: 59.37) on arable land (Table 5). The model showed that a moderate level of flood risk was most probable, together with a moderate (yellow) level of water quality. The results indicate that the optimal solution would affect both system objectives positively, i.e. the probabilities of low flood risk level and high (blue) level of water quality were both improved (Table 6).

TABLE 5 HERE

TABLE 6 HERE

In contrast to Scenario A, the conditions of Scenario B (Table 5) are associated with a higher level of overland flow (i.e. West of England: high rainfall; heavy soil with low infiltration capacity; medium slope). Under this scenario, a moderate level of flood risk and a good (green) level of water quality were most likely to occur. This result arises because on average there is a higher density of vegetation coverage under scenario B due to the selected land use, i.e. grassland (see assumptions in Table 1). In this scenario, natural vegetation also proved to be the optimal buffer strip management practice (satisfaction value: 59.91 – Table 5). Table 6 shows the changes in the probabilities of the management objectives occurring when this solution was applied. Again both flood risk and water quality are positively affected with patterns and magnitudes similar to scenario A.

The conditions of Scenario C are similar to Scenario B, but with steeper slopes. Again Natural vegetation was the optimal buffer strip solution, but with less overall utility (score: 59.25 – Table 5) than in scenario B (score: 59.91 – Table 5). Regardless of the steeper slope, in this scenario the optimal solution led to a greater improvement in flood control (Table 6) than in the previous scenario. This is because under the status quo, flood risk is likely to be higher as steeper slopes increase surface flow rates. As a result, riparian buffer strips have a greater impact on flood control and are hence more effective in areas with steeper slopes.

For each of the scenario results in Table 5 we also present the percentage change in utility relative to the status quo situation. This reveals that the application of buffer strips in scenario C has the largest relative impact on utility, although this scenario is associated with the lowest absolute levels of utility. Given the underlying assumptions of the BBN parameterisation it is not surprising that 'natural vegetation' is the optimal buffer strip solution in each scenario. However, our model does not consider the costs or opportunity costs of the buffer strip options; these would be needed to fully evaluate whether the gains in utility or changes in the probabilities of water quality and flood risk are sufficient to justify particular buffer strip options. The changes in utility in Table 5 as represented in percentage terms suggest that each of the buffer strip options performs relatively better in scenarios B and C compared to A. This is particularly the case with grassland buffer strips, but less so with natural vegetation or mixed buffer strips. From a policy perspective this can affect recommendations for both regional targeting of buffer strips and the types being promoted.

In Table 6 we can observe that the changes in the probabilities of preferred outcomes are higher for flood risk than for water quality. The increase in the probabilities of low flood risk and reduction in probability of high flood risk are much larger than changes in probabilities for either high (blue) or poor (red) water quality status.

6 Discussion

Our analysis explored a BBN using a framework that is suited to the integration of ecological and economic knowledge. The model was based on a review of the biophysical relationships between the ecosystem processes that lead to final ecosystem services and ultimately benefits that can be valued. Essentially we have unpacked and operationalized the ecosystem service cascade developed by Haines-Young and Potschin (2009). An important step in this operationalization was the introduction of specific management actions to which we can attribute utility values. The utility values used were determined for the specific purpose of this study, and serve to demonstrate the way final services and underlying processes can be related to an outcome that may be defined either in economic terms or that could be informed from non-monetary approaches such as identifying weights or scores using multicriteria analysis. Specifically, the BBN demonstrates that the utility associated with buffer strips is dependent on the supporting ecosystem processes and functions (e.g. soil, vegetation, organisms) and wider geographical and climactic contexts. It is in principle possible within the BBN to select specific levels of underpinning natural capital or ecosystem processes (e.g. infiltration, overland flow) and to evaluate their impact on the utility of buffer strip options in the decision node; in effect this potentially allows us to value those processes and states. There are a number of interesting consequences of the BBN approach that warrant further investigation.

As noted by Landuyt et al. (2013), the parameterisation of utility nodes can be informed by monetary valuation with stated preference methods being described as producing values that are compatible with BBNs. At first glance, choice experiments may appear to be most suitable for investigations of changes in multiple ecosystem service delivery because they allow valuation of multiple attributes. However, the attributes should not be causally related, i.e. benefits associated with a change in one ecosystem service (attribute) must be assumed to vary independently from other benefits. In cases where benefits are generated jointly as a result of a management intervention, contingent valuation will be more appropriate. The BBN model is also open to non-monetary valuation, for example through participatory ranking or weighting exercises. This approach would be of use where cultural and shared social values are of interest (UK NEA, 2011).

The nature of the outcomes produced by the BBN highlight an important consideration for valuation. The water quality and flood risk outcomes of the ecosystem processes represented in the model are probabilities for different states. This has the advantage of reflecting the inherent uncertainty of such outcomes in natural systems; however this may be problematic from an economic valuation perspective. The probabilistic nature of the outcomes raises questions with respect to the formation of values where those values themselves might also be uncertain (see for example Hanley et al., 2009). For example, if we were to develop a stated preference study of water quality states, would the willingness to pay for 'high' water quality be reduced where the probability of that outcome is low? And, could that value be lower than that stated for 'good' water quality where that outcome has a higher probability? The combined effects of outcome and value uncertainty might mean we are unable to differentiate between the values of outcomes.

The utility values, as currently expressed, refer to particular combinations of outcomes. But the model omits a necessary step in valuation which is to determine the value associated with moving between those outcomes, i.e. the management options are not evaluated with reference to a counterfactual. For example, to determine economic value we might elicit willingness to pay to move from a situation of no buffer strips to one with natural vegetation buffer strips; under scenario A we would be seeking the value of moving from a satisfaction value of 55.4 to one of 59.4. As it stands the BBN does not tell us how the status quo utility value of 55.4 was determined. Essentially, the BBN approach allows us to ascribe values to states of the world without consideration of how those states relate to alternative outcomes under different management or policy interventions (e.g. grass buffer strips versus no buffer strips). However, determining weights or 'values' for outcomes without reference to a counterfactual may be acceptable in a decision support context; such weights could be determined through participatory research, multicriteria analysis or expert judgement. If the aim of the model is to quantify monetary or non-monetary values this indicates a limitation of a fully integrated BBN. It would be necessary to make assumptions about how outcomes shift across categories. For example, would flood risk status be more likely to move between adjacent categories, medium to low rather than from high to low? Valuation counterfactuals would need to reflect the movement of outcomes between categories.

The probabilistic outcomes of the model suggest that there is a need to explore thresholds or other non-linearities that might influence preferences and values. For instance, in Scenario C, the optimal management action (grassland with natural vegetation buffer strips) sees an increase in probability of a low flood risk state from 21.3% to 27.7% with a concurrent decline in a high flood risk state from 32.5% to 24.2% (see Table 5). The question is whether there is some threshold level of reduction in high flood risk that must be crossed to allow the benefits of the increased probability of low flood risk to be realised, i.e. is there an acceptable maximum probability of flood risk being high? For example, the value of an increase in the probability of achieving a low flood risk state may be contingent on the probability of being in a high flood risk state falling below some specific level (e.g. 20%). Conversely, there may be thresholds above which the most desirable outcomes are sufficient to compensate for continuing risks of undesirable outcomes, e.g. low flood risk at the expense of 'medium' water quality levels. Valuation methods generally assume that ecosystem services are provided at a steady rate (i.e. linearly). However, there are many instances where interrelationships among the ecosystem services are remarkably non-linear (Farber et al., 2002; Koch et al., 2009; van Jaarsveld et al., 2005). Further, across multiple ecosystem services, there may be complex and interrelated non-linearities in preferences. Such non-linearities might reflect lexicographic preferences where there is no acceptable trade-off between probabilities of desirable and undesirable outcomes.

The model as formulated shows little apparent variation in utility values (Table 5) and probabilities of outcomes (Table 6), this reflects our choice of parameterisation for generic scenarios (i.e. two regions across multiple soil types, slopes and land uses). A more context specific parameterisation of values in the conditional probability tables may be necessary for studies investigating particular places. This may only be accommodated through either splitting the model into separate regions or land uses or by considerably increasing its complexity. The question then becomes one of whether we want to understand the processes involved or accurately model the outcomes.

Understanding the potential for extending the original BBN to more accurately represent both the biophysical and socio-economic elements of the system and place raises an important further issue. The attraction of the BBN approach is its relative simplicity and flexible data requirements. As models increase in complexity and realism the development task and data requirements become more exacting. Hence, there is ultimately a further trade-off between precision and usefulness which will depend on the needs of decision makers. But in situations where it is necessary to develop a joint understanding of ecosystem functioning, perhaps across multiple stakeholders, the relative simplicity of the BBN approach may be sufficient to make optimal decisions.

Our BBN model does not explicitly consider the socio-economic determinants of the values in the utility node. It is well recognised in the valuation literature (e.g. Garrod et al., 2012) that there is heterogeneity of preferences and that it is determined partly by a number of contextual factors. We propose a possible extension to the BBN (Figure 4) that incorporates socio-economic factors that might influence 'satisfaction' values for both water quality (income, type of recreational use, availability of substitutes, site amenities) and flood control (income, proximity). We have not evaluated this model as the additional socio-economic factors would need to be parameterised through further research (e.g. public workshops or surveys) that were beyond out project resources. In this extension the utility associated with water quality and flood control is separated, i.e. both provide benefits independently of one another. Although there are compelling reasons for joint consideration of utility, the benefiting populations may be different. The utility values in the decision node (‘buffer strips’) would still reflect the ‘joint’ value of the outcomes but without any implicit information on trade-offs between flood risk and water quality.

This extension is not intended to be comprehensive, but would allow us to explore the sensitivity of the BBN to both bio-physical and socio-economic assumptions. Further extensions could include additional terrestrial ecosystem services (landscape, biodiversity, recreation etc.) and the socio-economic factors influencing land manager decision making (Yu and Belcher 2011; Curtis and Robertson 2003). The latter would be important particularly if considering multiple measures or the relative value of public and private benefits (e.g. farm incomes) in policy making. This supply-side element of management remains a gap in ecosystem service evaluation and could add considerably greater complexity to an integrated model as willingness to adopt buffer strips has been shown to be dependent on a mix of economic, attitudinal and farm structural factors, in particular where there is interference with production (Buckley et al. 2012).

FIGURE 4 HERE

7 Conclusions

This research has proposed a novel way of operationalizing an ecosystem services-based approach following the ecosystem service cascade proposed by Haines-Young and Potschin (2009) for the identification and assessment of benefits of environmental interventions (in this case, riparian buffer strips). For that we have tested the potential of BBN as a tool for integrating knowledge across disciplines and dealing with information gaps and uncertainty. Our research represents a step further in the development and unpacking of (so far) theoretical frameworks for the conceptualization of ecosystem service delivery.

Interesting issues arise from the use of a BBN approach due to its probabilistic nature, as this both captures the uncertainty inherent in natural systems and raises questions over their incorporation in valuation and wider decision making where uncertainties over preferences are pervasive. The way these probabilities interact with non-linearities, thresholds, uncertainty in valuation and the statistical properties of valuation estimates (e.g. distributions and confidence intervals) will be key research areas if these approaches are to be used in interdisciplinary modelling and integrated decision support. Users of such models will also need to understand the trade-off between realism, precision and the benefits of developing joint understanding of the decision context.

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Table 1 Description of BBN nodes and states

Type of node	Variable	Definition	States	Dependencies	Assumptions
Decision	Buffer strip	Type of buffer strip installed in riparian areas	• Grassland • Natural vegetation • Mixed • No buffer strip		• Grassland buffer strips are uncultivated where land cover is arable and ungrazed or uncut where land cover is grassland • Natural vegetation would involve planting of trees or shrubs (offering shading of water)
Parent	Region		• East England • West England		• Generic regions which are interacted with season, land cover, soil type and slope
	Land cover		• Grassland • Arable • Natural vegetation		• Predominant type of land cover
	Seasons		• Autumn • Winter • Spring • Summer		
	Soil type		• Sandy (light) • Loamy (moderate) • Clay (heavy)		• Generic soil type reflecting drainage characteristics
	Slope		• Low • Medium • High		
Child	Riparian management	The vegetation type and level of coverage determined by the management intervention.	• Grassland • Natural vegetation • No riparian management	• Buffer strips	• This node allows buffer strips comprised of a mixture of grassland and natural vegetation
	Rainfall		• Low • Medium • High	• Region • Seasons	• West England is assumed to have higher rainfall rates than East England.
	Vegetation coverage	The proportion of ground surface covered by vegetation.	• Zero • Low • Medium • High	• Land cover • Seasons	• Grassland: grows all over the year with the highest density during spring/summer (i.e. is not much affected by seasonal changes) • Arable land: has the highest density during summer, does not grow during autumn • Natural vegetation: has the highest density during spring/summer, moderate density during autumn, the lowest density during winter
	Infiltration capacity	The ability of soil and plants to absorb water.	• Low • Medium • High	• Soil type • Vegetation coverage	• The greater the vegetation coverage, the higher the infiltration capacity will be. • Sand has high water permeability, whilst clay is more resistant to water infiltration.
	Overland flow	Water that flows across the	• Low	• Rainfall	• The higher the rainfall rate, the lower the infiltration capacity and the

		land after rainfall. It does not include the water volume intercepted by vegetation or infiltrated by soil and plants.	• Medium • High	• Infiltration capacity • Slope	steeper the slope, then the higher the overland flow will be and vice versa. • In order to minimise the number of nodes, evapotranspiration and volume of groundwater were regarded to contribute less to overland flow volume and were not included in the system.
	Soil erosion rate	The rate of soil erosion.	• Low • Medium • High	• Soil type • Vegetation coverage • Overland flow	• Clay is less erodible than sand. • Overland flow is assumed to have a greater impact (i.e. low overland flow will result in low erosion rate regardless of the soil type and vegetation coverage).
	Sedimentation load	The amount of sediments that reach water bodies (i.e. eroded soil particles that are not trapped by riparian vegetation).	• Low • Medium • High	• Soil erosion rate • Riparian management	• Grass covered surfaces facilitate greater rates of sediment deposition due to their high root density. • Sediment load is likely to be higher when no riparian management is applied.
	Water nutrient concentration	The amount of nutrient content in stream water. Increased levels of nutrients in water bodies can cause water quality problems such as excessive plant growth rates (e.g. algae blooms) and eutrophication (Hime et al., 2009).	• Low • High	• Land use • Sedimentation load	• Arable land is assumed to result always in high water nutrient concentration due to use of fertilizers. • The greater the sedimentation load, then the higher the water nutrient concentration will be (because sediments transport substances such as plant and animal wastes, nutrients, pesticides, metals etc.). • Nutrient plant uptake is assumed to be fixed regardless of the land-use type. • Soil type effects are captured indirectly through erosion and sedimentation load.
	Aquatic vegetation	The volume and density of vegetation growing into the water bodies. Aquatic vegetation is considered to have an effect on the velocity of river flow.	• Algae • Vascular plants	• Water nutrient concentration • Seasons	• Under conditions of high nutrient concentration and high temperature (spring/summer), algae blooms will occur in water bodies (Borsuk et al., 2004). • The level of nutrients has been assumed to have a greater impact than temperature (i.e. despite high temperatures, algae will not bloom unless the water nutrient level is high).
	Temperature	Water temperature	• Low • Medium • High	• Riparian management • Season	• Natural vegetation has a decreasing effect on temperature via shading.
	Biological oxygen demand (BOD)	The amount of dissolved oxygen required by microorganisms (e.g. aerobic bacteria) in the oxidation of organic matter. In the scope of this study, BOD is used as an indicator of water quality.	• Lower than 4 mgl^{-1} • 4-6 mgl^{-1} • 6-9 mgl^{-1} • Higher than 9 mgl^{-1}	• Aquatic vegetation • Water nutrient concentration • Temperature	• High temperatures and high level of water nutrient concentration result in algae blooms. This implies increased organic matter and thus higher level of BOD (i.e. the process of decomposition leads to oxygen depletion). • Characteristics such as the surrounding atmospheric pressure and the salinity of water regarded to contribute less to BOD and were not included in the model.
	Water quality	Suitability of water for fishing, swimming, boating, or unsuitability for any use	• Blue • Green • Yellow	• BOD	• Each water quality category was converted into a BOD level, as following: • Blue = 0 - 4 mgl^{-1},

		(Hime et al., 2009).	• Red		• Green = 4 - 6 mgl^{-1}, • Yellow = 6-9 mgl^{-1}, • Red = higher than 9 mgl^{-1}
	Runoff rate	The rate of surface water that reaches water bodies (when soil is saturated and infiltration capacity is lower than the rainfall rate).	• Low • Medium • High	• Riparian management • Overland flow	• Natural vegetation is assumed to be more effective than grassland in reducing runoff. • Overland flow is assumed to have a greater impact (i.e. low overland flow will result in low runoff rate regardless of the applied riparian management). • It is assumed that the runoff rate is always likely to be higher when riparian management is not applied.
	River flow	Volume of water flow in any given time period	• Low • Medium • High	• Runoff rate • Rainfall • Aquatic vegetation	• The lower the runoff rate, rainfall rate and aquatic vegetation coverage, the lower the river flow will be. • Compared to algae, vascular plants are assumed to decrease more the velocity of river flow. Particular aquatic vegetation characteristics (e.g. height, rooting depth etc.) were not taken into consideration.
	Flood risk	Likelihood of a flood event	• Low • Medium • High	• River flow	• Flood risk has been modelled as a deterministic variable. The higher the river flow, the higher the flood risk will be and vice versa.
Utility	Satisfaction	The utility that stakeholders will gain from the management intervention.	• Continuous variable (scale 0-100)	• Flood risk, • Water quality	• It is assumed that the system objectives contribute equally to the output of the model (i.e. people will be totally satisfied only when both of the model objectives have been fully optimised).

Table 2 Conditional Probability Table (CPT) for the 'overland flow' node.

State of parent nodes			Probability of overland flow outcome		
Infiltration capacity	**Rainfall**	**Slope**	**Low**	**Medium**	**High**
Low	Low	Low	0.6	0.3	0.1
Low	Low	Medium	0.6	0.3	0.1
Low	Low	High	0.6	0.3	0.1
Low	Medium	Low	0.3	0.6	0.1
Low	Medium	Medium	0.1	0.6	0.3
Low	Medium	High	0.1	0.3	0.6
Low	High	Low	0.1	0.3	0.6
Low	High	Medium	0.1	0.3	0.6
Low	High	High	0.1	0.3	0.6
Medium	Low	Low	0.6	0.3	0.1
Medium	Low	Medium	0.6	0.3	0.1
Medium	Low	High	0.6	0.3	0.1
Medium	Medium	Low	0.3	0.6	0.1
Medium	Medium	Medium	0.3	0.6	0.1
Medium	Medium	High	0.1	0.6	0.3
Medium	High	Low	0.1	0.6	0.3
Medium	High	Medium	0.1	0.3	0.6
Medium	High	High	0.1	0.3	0.6
High	Low	Low	0.6	0.3	0.1
High	Low	Medium	0.6	0.3	0.1
High	Low	High	0.6	0.3	0.1
High	Medium	Low	0.6	0.3	0.1
High	Medium	Medium	0.6	0.3	0.1
High	Medium	High	0.3	0.6	0.1
High	High	Low	0.3	0.6	0.1
High	High	Medium	0.3	0.6	0.1
High	High	High	0.1	0.6	0.3

Table 3 Conditional Probability Table (CPT) of the model utility node.

Flood risk	Water quality	Utility value
Low	Blue	100
Low	Green	100
Low	Yellow	75
Low	Red	50
Medium	Blue	65
Medium	Green	65
Medium	Yellow	50
Medium	Red	35
High	Blue	50
High	Green	50
High	Yellow	25
High	Red	0

Table 4 Characteristics of three scenarios examined in this study

Scenario	**Region**	**Land cover**	**Soil type**	**Slope**
A	East England	Arable land	Light free draining (sandy)	Low
B	West England	Grassland	Heavy poor draining (clay)	Medium
C	West England	Grassland	Heavy poor draining (clay)	High

Table 5 Utility values for the three scenarios

Scenario	Buffer strip management (% increase in utility relative to status quo)			
	Status quo	Grassland	Natural vegetation	Mixed
A	55.39	56.71 (2.4%)	59.37 (7.2%)	58.04 (4.8%)
B	55.61	58.23 (4.7%)	59.91 (7.7%)	59.07 (6.2%)
C	54.53	57.42 (5.3%)	59.25 (8.7%)	58.33 (7.4%)

Table 6 Changes in the probability of outcomes under the optimal solution.

Scenario	Outcome	Status	Status quo (%)	Grassland (%)	Natural vegetation (%)	Mixed (%)	Change in prob. Status quo to optimal
A	Flood risk	Low	27.4	30.7	34.0	32.4	6.6
		Medium	49.0	47.4	47.3	47.4	-1.7
		High	23.6	21.9	18.6	20.3	-5.0
	Water quality	Blue	17.1	17.1	18.8	17.9	1.8
		Green	27.8	27.8	28.9	28.3	1.1
		Yellow	32.3	32.3	31.5	31.9	-0.8
		Red	22.8	22.8	20.9	21.8	-2.1
B	Flood risk	Low	23.0	26.1	29.1	27.6	6.1
		Medium	46.7	46.7	47.7	47.2	1.0
		High	30.4	27.3	23.3	25.3	-7.1
	Water quality	Blue	22.5	24.4	25.2	24.8	2.7
		Green	31.8	33.1	32.2	32.7	0.5
		Yellow	28.4	26.9	27.0	27.0	-1.4
		Red	17.4	15.5	15.6	15.6	-1.8
C	Flood risk	Low	21.3	24.5	27.7	26.1	6.4
		Medium	46.2	46.7	48.1	47.4	1.9
		High	32.5	28.7	24.2	26.5	-8.3
	Water quality	Blue	22.3	24.4	25.0	24.7	2.7
		Green	31.6	33.1	32.2	32.7	0.6
		Yellow	28.5	27.0	27.1	27.0	-1.4
		Red	17.6	15.5	15.7	15.6	-1.9

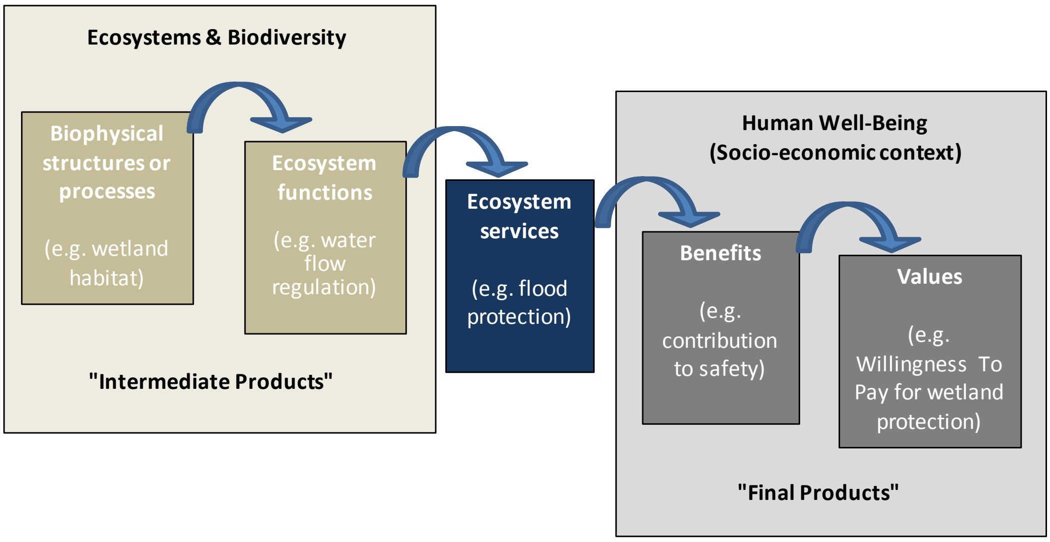

Figure 1 Ecosystem service cascade (Adapted from Haines-Young and Potschin 2009)

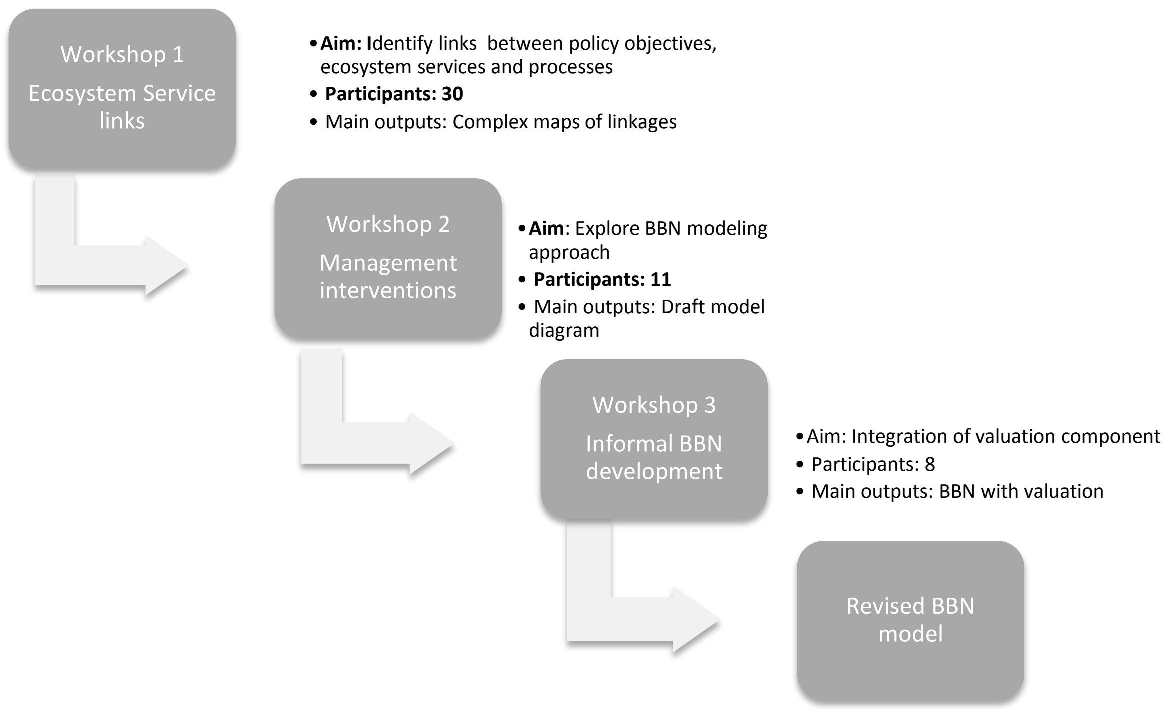

Figure 2 Sequence of interdisciplinary workshops used for BBN development

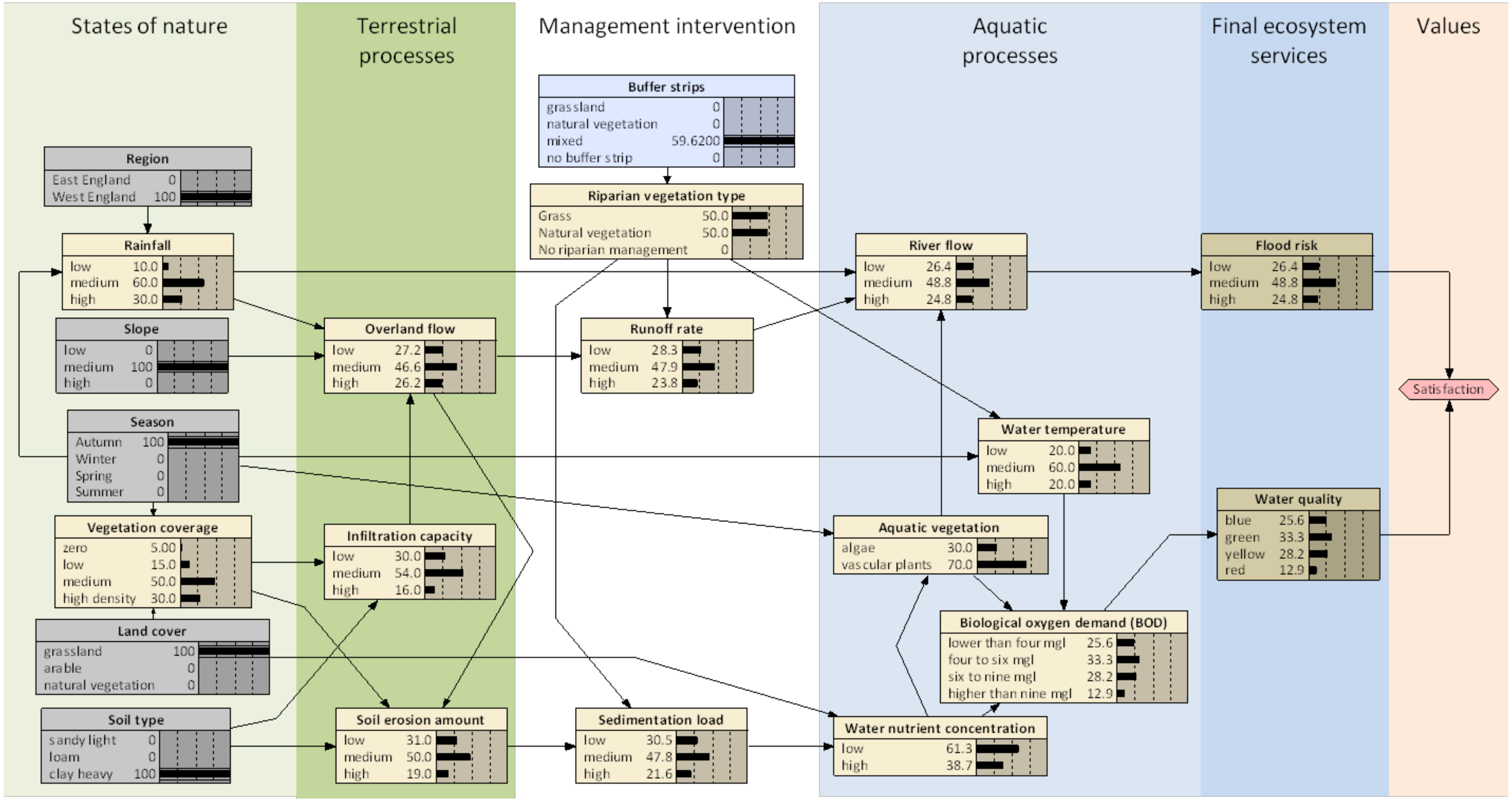

Figure 3 BBN model (Scenario B) of riparian buffer strip management system

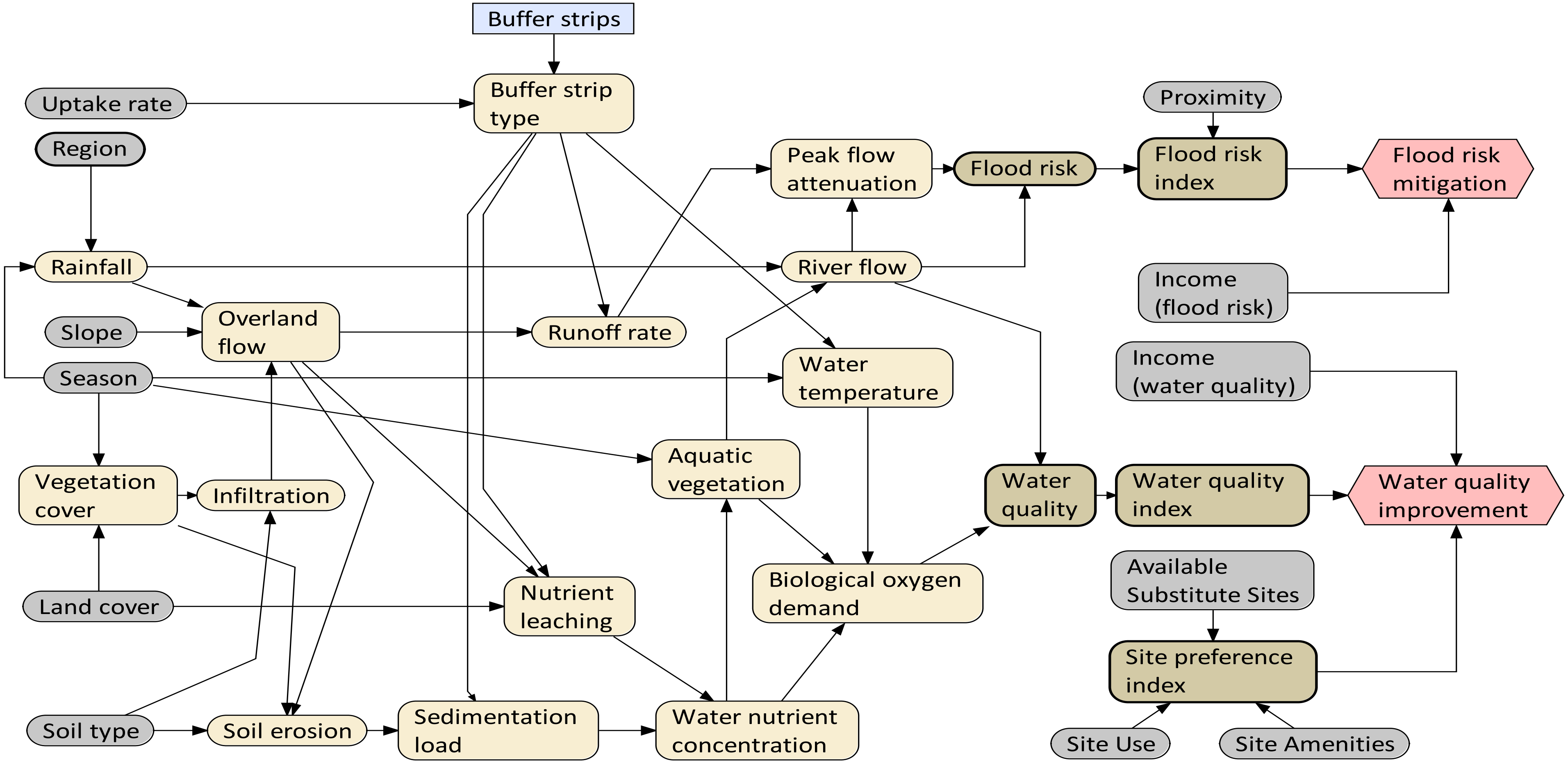

Figure 4 Expanded BBN incorporating socio-economic drivers of preferences