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Exploring the interaction of ecosystem processes and ecosystem services for effective decision-making

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Abstract

In this paper we develop a joint ecological-economic model using a Bayesian Belief Network (BBN) to assess options for riparian zone management. The riparian zone is the interface between terrestrial land management and the aquatic environment which provides key ecosystem services including water quality and flow regulation. These services contribute to benefits including clean water, recreation and reduced flood risk. Our approach aims to capture the complexity of interactions between the underlying ecosystem process and services of interest. This ecological model is then integrated with economic values to determine optimal approaches to riparian zone management where flood risk and water quality are objectives.

Whilst we can identify optimal management actions the probabilistic nature of the BBN model raises some key issues. Uncertainty over outcomes has implications for how to approach valuation particularly where preferences might exhibit non-linearities or thresholds. The interaction between probabilistic outcomes and the statistical nature of valuation estimates point to the need for further exploration of sensitivity in such models. Although the BBN is a promising participatory decision support tool, there is 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

Introduction

Recent years have seen the growing adoption of ecosystem services as a framework for both analysis and decision-making in with respect to the environment. This framework has also allowed the development of a common language across natural and social science disciplines that in turn has led 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 develop common models to explore our joint understanding of ecosystem services and how this may in turn better inform management and policy. To this end there have been some targeted attempts to foster interdisciplinary working such as the UK's Valuing Nature Network[1].

[1] http://www.valuing-nature.net/

A common problem with developing interdisciplinary analysis has been the degree of complexity that different disciplinary and modelling approaches are able to incorporate and the consequent problem of then integrating these approaches. Physical science approaches to ecosystems implicitly recognise and address complexity by adopting context specific field studies or models (or specifically the parameterisation of models). Conversely, economic approaches such as valuation often need to be broad-brushed both to avoid overburdening respondents in stated preference approaches; incorporating often fairly aggregate economic data; and in applying results at suitable scales to meet the needs of policy. Therefore, there is a need to either scale-up biophysical approaches to meet scaled-down economics, or to explore jointly developed models where we explicitly trade-off precision in disciplinary approaches to achieve outcomes that are still of use to decision making.

A further consideration in developing these joint approaches is the need to unpack the commonly used classifications of ecosystem services and the underlying ecosystem processes (functions and intermediate services) to identify the key ecosystem processes and the degree of interactions between them. In this paper we present the development of an interdisciplinary approach and hope to provoke discussion and debate about how such an approach can be operationalised. In the next section we discuss the issues of complexity and interactions in ecosystem service analysis; we then outline our adopted approach before describing an application. We then discuss outputs from this model before introducing possible enhancements.

Ecosystem services – complexity and interactions

The natural world entails a great extent of complexity due to significant interdependencies between the components of an ecosystem as well as between different ecosystems. As Rodríguez et al. (2005) have stressed, "ecosystem services do not operate in isolation" (p. 443). On the contrary, they are highly dependent on each other (Heal et al. 2001, Pereira et al. 2005) and, hence, describing and quantifying adequately their interactions within and across ecosystems has been a principal challenge in valuing nature. In some cases, there is a range of ecological mechanisms that interact within an ecosystem in order to generate either a single service or multiple services, whereas in other cases a single mechanism contributes to more than one ecosystem service. Moreover, the provision of a single ecosystem service may be dependent on the contributions of many different ecosystems (Defra, 2007). Subsequently, a policy decision that affects any part of those interactions can cause changes in multiple services or multiple ecosystems. From an economic perspective, that means that the economic value of any ecosystem service may be determined by its relationship with other services (UK NEA, 2011); and, thus, environmental valuation should attribute values to ecosystem services taking into account the dynamics and interdependencies of ecosystem functioning.

In this context, the generic conceptual framework of the Millennium Ecosystem Assessment (Figure 1), although it remains useful, is insufficient because it does not incorporate the "production chain" that underlies behind the generation of any ecosystem service (Haines-Young & Potschin, 2009). There has been considerable debate that this typology (i.e. provisioning, regulating, cultural and supporting services) does not distinguish the "means" which contribute to the production of ecosystem benefits from the "end-products" that people actual experience (Wallace, 2007). 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" that contribute to their delivery. Ecological "functions" and "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. Moreover, according to this definition, "ecosystem services" are not equivalent to "benefits" or "goods" as it is suggested in MA (2005). These complex interrelationships among ecosystem structures, services and benefits are emphasized by Haines-Young and Potschin (2009), who 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 2). This development is also reflected in the framework adopted by the UK NEA.

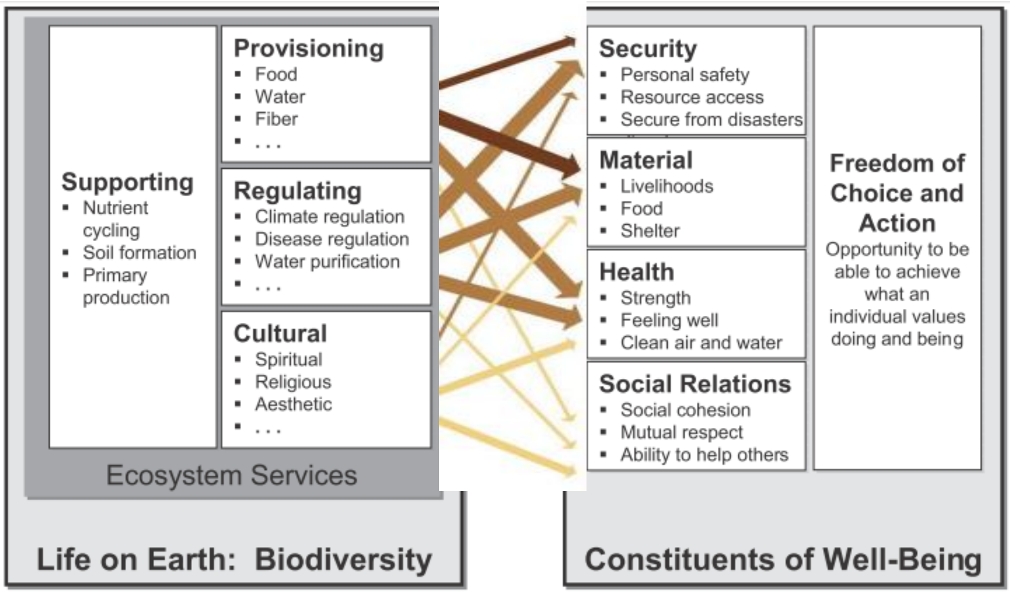

Figure 1 Ecosystem services and their links with well-being (Source: MA, 2005)

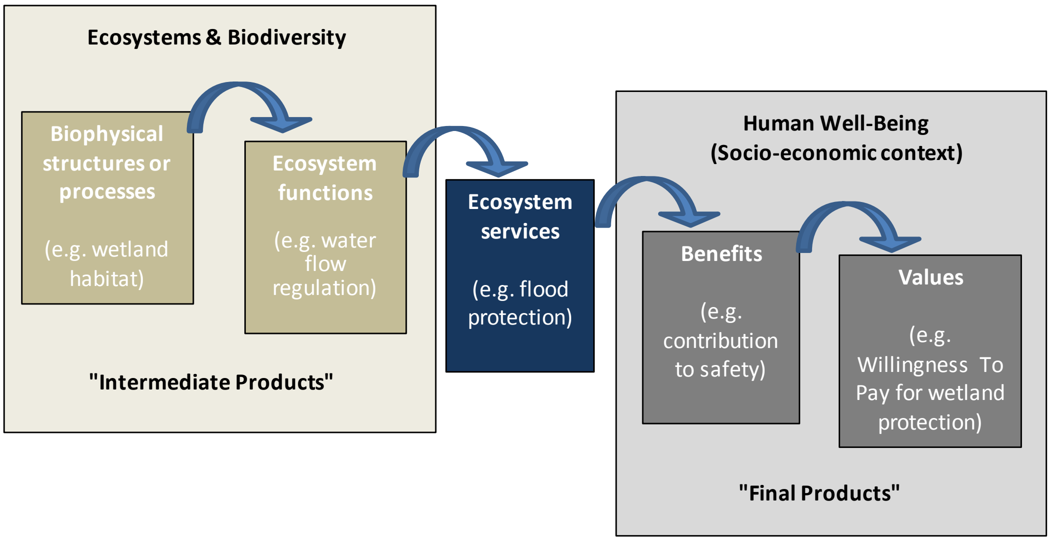

Figure 2 Ecosystem service cascade (Adapted from Haines-Young & Potschin 2009)

In the context of environmental valuation, this alternative classification of ecosystem services into "intermediate processes", "final services" and "benefits" addresses the problem of "double counting" the values of ecosystem services (Boyd & Banzhaf, 2007; Fisher & Turner 2008; Fisher et al., 2009). For instance, considering the case of a wetland, the intermediate functions of nutrient cycling and water regulation interact to deliver clean water. But, the actual benefit that humans derive from water provision is recreation (e.g. angling, swimming) or potable water (Fisher et al., 2009), values should only be placed to the end-products that are directly consumed, because these estimates would also contain the contribution of the ecological structures and processes. Consequently, intermediate processes are still valuable and policy makers should not concentrate only on final services, putting underpinning ecological assets at risk through overexploitation (Gren et al. 1994; Turner, 1999).

Non-linearity in ecosystem services

Apart from identifying the important ecosystem services and their interactions with ecological functions and processes, considerations of spatial and temporal scales as well as irreversible effects are also important in understanding the dynamics of an ecosystem. Even though valuation methods generally assume that ecosystem services are provided at a steady rate (i.e. linearly), there are many concerns that the interrelationships among the ecosystem services are remarkably non-linear (Farber et al., 2002; Koch et al., 2009; van Jaarsveld et al., 2005). Farnsworth (1998) argues that ecosystem services and ecosystem functions tend to respond non-linearly across location and time due to forcing variables such as disturbance, species interactions and seasonality. Wave attenuation, for instance, has a dynamic value which can vary across different time periods, reaching a minimum level in the winter (when biomass as well as density are low) and a maximum level in the summer (when the majority of plant species are reproduced) (Chen et al., 2007). In addition, ecosystem functioning can be dramatically changed over space and time due to irreversible effects that occur when certain environmental limits and thresholds are surpassed (Koch et al., 2009). Given our lack of understanding such non-linear interactions between the ecosystem components (Aburto-Oropeza et al., 2008), incorporating a spatial and temporal context as well as considerations about critical thresholds is an additional challenge in environmental policy appraisals. Nevertheless, from a management perspective, it is significant the value of an ecosystem and its ability to provide benefits to humanity to be evaluated across its full extent in space and time. Following, concerns about spatial and temporal scales as well as irreversibility will be discussed in more depth.

Policy and spatial issues

Specifying the spatial boundaries (i.e. geographical location) of an ecological analysis is a critical step in a valuation process. For instance, Defra (2007) argue that, "water quality for a given community may depend more on the condition of the upstream portions of the catchment than on the areas within the community. As a result, the analysis will need to carefully consider the spatial scale required for ecological analysis, particularly when linking indirect to direct drivers of change or ecosystem services to human wellbeing" (p. 41-42). Furthermore, policy-makers should take into account the spatial scale in which human population is affected by a policy. In other words, it is important to identify whether a change of an ecosystem service affects people in a local, regional or a global level. Considering the uniqueness of an ecosystem service, for example, "a rare species in England may have significant non-use values attached to it across a wide population, whereas for a less unique species, values may be held only by a local population" (Defra, 2007: p.42). Finally, it is essential for decision-makers to understand that in some cases the site where people derive the benefits of a policy does not match to the place where the costs are borne. For example, better management of a river catchment could be a benefit to downstream landowners, but a cost for a upstream community (Carpenter et al., 2009).

Human interventions, trade-offs and synergies

As it is already noted, the delivery of ecosystem services is fundamentally based on their interactions with ecosystem structures and processes. We argued that the quantification of these interrelationships is a major challenge in valuing the environment. However, apart from these direct interactions, the literature review suggests that ecosystem services may also interact indirectly through their response to a shared driver (i.e. through the impacts that a human intervention has on multiple services) (Bennett et al., 2009). For instance, converting coastal mangroves to shrimp farming has an impact on multiple services, affecting at the same time coastal protection against floods, wood collection and habitat provision for fisheries (Barbier et al., 2008). In this case, a policy appraisal that accounts only for the value of shrimp production and ignores the interconnections among the services provided by mangroves would cause simultaneous failures in many ecosystem services, leading in environmental degradation. Therefore, there is a growing consensus across the literature that knowledge and awareness of how ecosystem services interact to certain management practices are essential for managing the natural world appropriately (Balvanera et al., 2001; Higgins et al., 1999; Grasso, 1998; Kearns et al., 1998; Rodríguez et al., 2005; Rose & Chapman, 2003). According to Bennett et al. (2009), the above-mentioned indirect interconnections between ecosystem services can be classified into two broad categories: "trade-offs" and "synergies".

As an example of trade-offs, increased use of fertilizer affects more than one service. On the one hand, it enhances crop production, but, on the other it decreases water quality through polluted agricultural runoff (Carpenter et al., 1998). In such cases that human interventions have unintended consequences, Bennett et al. (2009) argue that significant trade-offs are created between the ecosystem services. In other words, both of them are affected by a common driver, but their impacts are in opposite directions (i.e. one service is being enhanced while another is being diminished) since the supply of one service is in conflict with the delivery of the other.

In contrast, synergies are associated with cases that the impacts occurred by a shared driver are in the same direction (i.e. enhancing or decreasing both services). For instance, a management policy like wetland restoration affects positively both flood protection and water quality (Hey, 2002; Zedler, 2003). Similarly, restoring riparian vegetation can improve both crop productivity and flood control (Kramer et al., 1997). The lack of integration in policy making can exacerbate trade-offs or lead to sub-optima synergies.

Bennett et al. (2009) stress how important is to determine the mechanism that causes the relationships between the ecosystem services in order to develop a strategy suitable to take advantage of synergistic opportunities or avoid unwanted trade-offs. In cases, for example, that a trade-off between two services is only due to a common driver, then there is no direct interaction between the two services and policy-makers can address the trade-off by manipulating the effects of the common driver (see Figure 3.2: case 1). On the contrary, if apart from the shared driver there is also a true interconnection between two ecosystem services, then a strategy that addresses only the common driver is not sufficient to diminish the trade-off (see Figure 3: cases 2 & 3). Therefore, trade-offs and synergies require a different strategy according to the cause of their relationship.

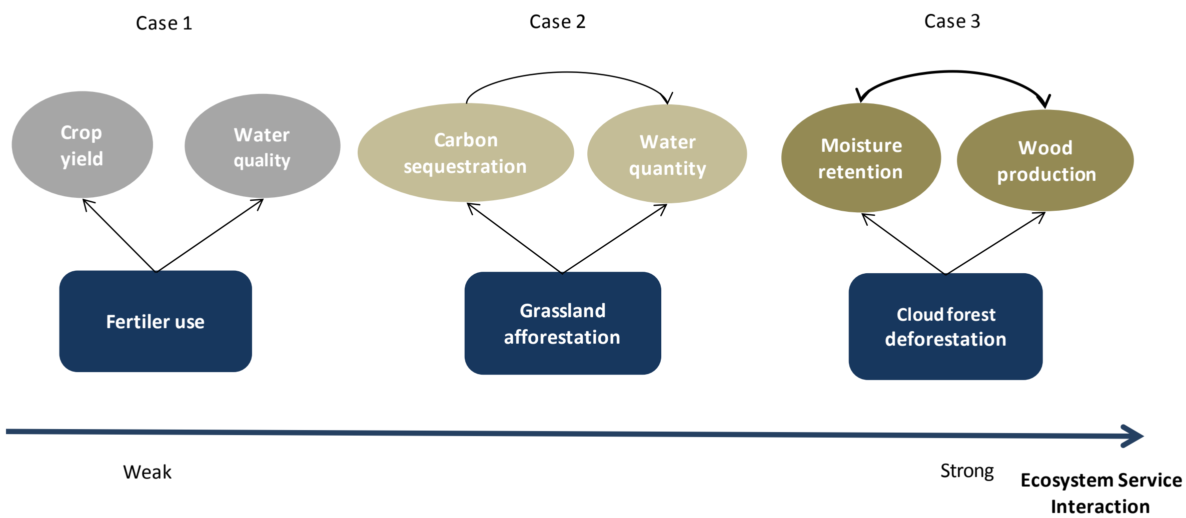

Figure 3 Relationships among ecosystem services (Source: Adapted from Bennett et al., 2009)

Inadequacy of the existing valuation methods

Economic valuation approaches may be inadequate to manipulate the complexities associated with natural systems. In general, the assessment of the natural environment tends to have a narrow focus on only one service each time (Turner et al., 2003), disregarding the ecosystem functioning and how other services are being affected. Even the Millennium Ecosystem Assessment (2005) appraised each ecosystem service individually, with only a few exceptions of dealing with the relationships between more than two services. Due to the inherent interconnections within an ecosystem, however, it may be insufficient to isolate and study the delivery of an ecosystem service without simultaneously considering other services (NRC, 2005). In fact, there are many concerns that efforts to optimize only a single service or a limited subset of services can lead to dramatic declines of other services and sudden shifts of natural systems (Diaz & Rosenberg 2008; Gordon et al., 2008; Holling & Meffe, 1996).

Furthermore, there are many considerations that the current economic valuation techniques cannot capture the true value of ecosystems. Existing valuation methods overall assess environmental assets by examining the goods produced by the ecosystem services and their contribution to the human welfare (e.g. use and non-use values). However, the value of those goods cannot be entirely assigned to ecosystem services, because in some cases their generation is based not only on natural resources, but also on human capital (Bateman et al., 2011). Moreover, as Bingham et al. (1995) highlight "even if ecosystem services were perfectly understood and evaluated in monetary units according to accepted economic percepts, the problems of ecosystem valuation would not be completely solved. The problems of uncertainty (both statistical and scientific), irreversibility, and large disparities in temporal and spatial scale remain." (p. 90).

NRC (2005), having reviewed a series of case studies that attempt to integrate ecological and economic knowledge in valuing either a single or multiple ecosystem services, conclude that our inability to estimate the true value of ecosystem 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.

Therefore, in order to tackle the methodological challenges of valuing ecosystem services, there is a growing consensus that integrated studies should be undertaken, which will account for the interactions and non-linear relationships among ecosystem components (Carpenter et al., 2009; Kremen & Ostfeld, 2005; Tallis & Kareiva, 2005; Turner et al., 2003). Many authors suggest that it is necessary to develop a more holistic (Turner & 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), to ensure that the full value of natural capital is reflected into decision-making (Defra, 2007). In other words, there is need for an approach that could quantify the economic value of the ecosystem service cascade, integrating the underlying linkages between services and processes and, thus, providing a more accurate estimate of the ecosystem value.

In that sense, Hein et al. (2006) suggest that the process of ecosystem service valuation should consist of five steps: 1) identification of the boundaries of the natural system to be valued; 2) evaluation of the ecosystem services provided by the system in biophysical terms and 3) in monetary terms; 4) aggregation or comparison of the different ecosystem service values; and 5) analysis at different scales and beneficiaries involvement. Similarly, Muller et al. (2010) stress the need for an integrative approach which integrates multiple ecosystem services (i.e. does not focus only on a single service or a limited set of services) as well as their interactions with ecological functions and processes; the interrelationships between the parts of the ecosystem service cascade; the different spatial and temporal scales; stakeholders into the decision making process.

Developing an integrated ecosystem-economic model

In this section we develop a decision support tool in an attempt to integrate land and water resource management for effective sustainable flood management and water quality. Restoration of riparian vegetation coverage will be the management intervention under consideration. The scope of the study was to investigate ecosystem service provision under alternative land-use and riparian management scenarios, and predict their likely impacts on aquatic systems with a particular focus on flood protection and water quality. The aim of the analysis is to contribute to understanding the potential synergistic and competitive interactions among ecosystem services in order to support effective decision making.

A Bayesian Belief Network approach was adopted to model the interactions among the ecosystem services of riparian zones. The aim was not to develop a model of a particular riparian system, but rather to create a generic model that represents knowledge of riparian ecosystem functioning. The effectiveness of riparian vegetation restoration will be investigated at a regional scale with alternative scenarios relevant to the East and West of England. These offers contrasting climatic, topographic and land use conditions.

Bayesian Belief Networks: What are they?

Bayesian Belief Networks (BBNs) are a modelling approach based on the probability theory that Thomas Bayes introduced in the 18th century (Pearl, 1988; Jensen, 2001). Applying the Bayes' Theorem, BBNs show mathematically how prior knowledge can be revised when new evidence is acquired (Kragt, 2009). They are particular useful in organizing knowledge and beliefs by depicting relationships of variables (based on cause-and-effect assumptions) and quantifying the likelihood of a variable to be affected by another (Henriksen et al., 2004).

This method is considered a powerful tool in modelling complex systems and supporting decision making, and thus it has been widely applied in a variety of fields, as for example in medicine (Andreassen et al., 1991; Hamilton et al., 1994); engineering (Heckerman et al., 1995); and artificial intelligence (Charniak, 1991). In ecological modelling, BBNs have been employed in numerous environmental studies including fisheries assessment (Kuikka et al., 1999; Lee & Rieman, 1997; Pollino et al., 2007); forest restoration (Haas et al., 1994); climate change problems (Gu et al., 1996; Kuikka & Varis, 1997); habitat restoration (Rieman et al., 2001); and watershed management (Ames et al., 2005; Borsuk et al., 2004; Bromley et al., 2005; Henriksen et al., 2004). The significance of BBNs in natural resource management underlies in predicting the links between management practices and ecosystem reactions (Clark et al., 2001; Borsuk et al., 2004), while 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 (Pearl, 1988; Bromley et al., 2005).

In practice, a BBN is a graphical representation of the expected interactions between a range of variables or else uncertain quantities. It is structured as a directed acyclic graph (DAG), 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, while the linkages among them indicate direct causal dependencies (Pollino et al., 2007) and 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, while those that are conditionally dependent on at least one other are called “child” nodes (Figure 4). Nodes that do not have any children constitute the output factors of the system.

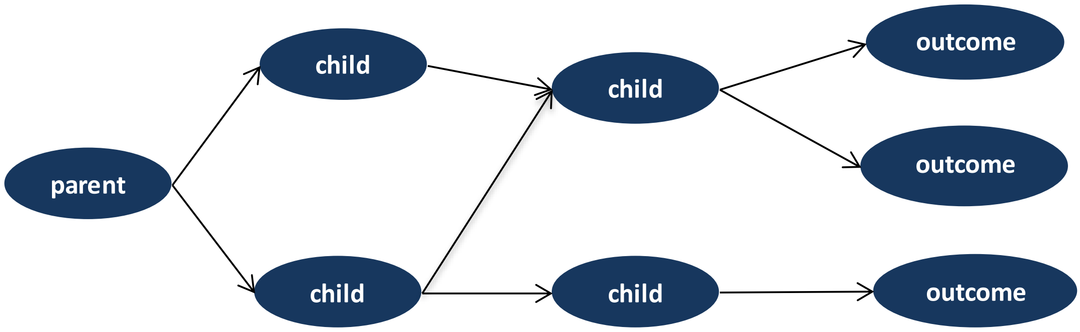

Figure 4 Graphical representation of a Bayesian network (adapted from: Bromley et al., 2005; Norsys, 2003)

The strengths of the causal relationships among the system variables are quantified by conditional probabilities, which are defined by a set of conditional probability tables (CPTs). In other words, CPTs specify the probability of each variable to have a particular “state” (i.e. value) considering every possible combination of states of the nodes linked to it (i.e. parent nodes) (Kjærulff & Madsen, 2005; Kragt, 2009; Pollino et al., 2007; Bromley et al., 2005). The state of the variables that do not have any prerequisites is determined by a marginal (or unconditional) distribution of probabilities (Pollino et al., 2007; Borsuk et al., 2004). Having specified the values of parent nodes, the a priori probabilities of their children can be calculated according to the Bayes’ rule using a CPT. Depending on the system, the variables can be determined either as discrete or continuous (Cain, 2001). The former are well-defined variables with a finite set of possible states, while the latter are variables such as temperature, whose value can range between two discrete values. The state of each variable can be either a numerical value, or a verbal description, or even a true or false statement (Bromley et al., 2005). Moreover, the probability values can be either observed data, information elicited from experts or a combination of these sources (Pollino et al., 2007).

It is important to note that BBNs are differentiated from other models in that the values associated with each variable can be quantified independently (Borsuk et al., 2004). This means that each CPT can be updated locally (Castelletti & Soncini-Sessa, 2007) and as a result it can be easily revised every time that new data or knowledge become available in order the improved data set to be reflected (Bromley et al., 2005). Therefore, BBNs are inherently adaptable and their implementation allows the use of the best available information at each time.

The major steps in developing a BBN include:

Network construction: conceptualizing the problem by determining which system factors are linked to which and producing a causal network.

Model parameter estimation: assigning states and values of probabilities to each variable (i.e. populating each CPT with probability values).

Model validation: reviewing the model to examine whether it behaves as expected.

Scenario analysis: examining how sensitive is the system in changes of particular variables.

Why the Bayesian Belief Networks approach was chosen

The BBN approach is considered a powerful integrated modelling method, which can capture system complexities by incorporating environmental, social and economic concerns (Kragt, 2009) and be particularly helpful in complex problems in which system variables are highly interlinked (Henriksen et al., 2004). In addition, it provides a framework that can take into consideration the contributions of multiple stakeholders (Bromley et al., 2005), allowing the development of a joint understanding of ecological systems and decision contexts. Thus the approach is suited to undertaking interdisciplinary analysis.

Moreover, in contrast to the majority of the environmental models, BBNs represent the relationships among variables using probabilistic expressions instead of deterministic ones (Borsuk et al., 2004). They are considered ideally suited to dealing with the uncertainty that is associated to any strategy or decision (Bromley et al., 2005), expressed in a BBN by the means of probabilities. To put it differently, BBNs can take into account uncertain knowledge or poor control on the impacts of causes (Henriksen et al., 2004) by modelling the highly unpredictable relationships among the system variables using weak conditional probabilities and vice versa (Varis & Kuikka, 1999).

An application – riparian management

Riparian areas are described as the land immediately adjacent to water bodies such as streams, rivers and lakes (Miller, 2006; Bannerman, 1998). From an ecological perspective, riparian land is defined as a three-dimensional zone (i.e. longitudinal, lateral, and vertical) in which terrestrial and aquatic habitats directly interact with each other (Gregory et al., 1991) exchanging energy and matter (NRC, 2002). Having described these areas as the "the interface between terrestrial and aquatic ecosystems", Gregory et al. (1991) suggest that their boundaries "extend outward to the limit of flooding and upward into the canopy of the streamside vegetation" (p. 540). In the same sense, Ward et al. (2002) define riparian areas - or riparian corridors - as landscape units and in particular as "linear features of the landscape structured along ribbons of alluvium from the headwaters to the sea" (p. 518).

Considered as one of the most heterogeneous and dynamic natural environments (Zogaris et al., 2009), riparian areas support key ecosystem functions such as water storage, water cycling (flow), infiltration capacity, sediment retention, nutrient cycling and habitat provision (Dwire & Lowerence, 2006; Hughes, 1997; Soman et al., 2007). On the whole, it is widely accepted that riparian vegetation affects to a great extent the stability of fluvial systems (Beschta, 1991; Gregory et al., 1991; Tabachi et al., 2000; NRC, 2002). Its role varies from regulating terrestrial and stream temperature to improving water quality and reducing flood risk. Hence, Holmes et al. (2004) argue that riparian environments deliver numerous critical ecosystem services, including both services with use as well as non-use values.

Ecosystem services provided by riparian areas

In this section we detail the ecosystem services delivered within riparian zones and map the complex interactions that occur between them. In general, ecosystem service provision of riparian areas differs from site to site and depends on numerous factors, such as their size, geomorphological characteristics, type of soil, slope of land etc. (NRC, 2002). Despite the variation that can be found in different sites, the most important ecosystem services performed in riparian ecosystems are summarised in Table 1.

Table 1 Examples of ecosystem services from riparian areas

Ecosystem Service	Underlying process
Flood mitigation	Vegetation increases hydraulic roughness and reduces surface runoff.
Water storage and infiltration	Soil and plant roots increase water storage capacity through infiltration and absorption.
Interception of precipitation and transpiration	Riparian vegetation captures precipitation and returns water to the atmosphere through transpiration.
Erosion control	Decreased volume and rate of runoff (through infiltration). Increased soil surface roughness enhances resistance to erosive forces of water flow.
Sediment retention	Trapping and storage of sediment particles carried in suspension by overland runoff. Reduces nutrient loads to water bodies, protecting aquatic systems from water quality problems (e.g. eutrophication).
Nutrient storage and cycling	Nutrient filtration and sink through vegetative uptake of excess nutrients from flowing water, groundwater and soil. Nutrients are stored in soil organic matter, recycled by biodiversity and converted back into living matter through decomposition.
Water quality protection	Assimilation of sediments and absorption of excess nutrients. Purification of groundwater proximal to roots (i.e. shallow groundwater).
Soil retention	The plant root matrix and soil biota play a role in soil retention in conjunction with erosion and siltation reduction.
Climate regulation	Temperature maintenance due to shading or by cooling through evapotranspiration.
Carbon regulation	Carbon sequestration during photosynthesis and storage in soil organic matter.
Habitat provision, biodiversity and ecosystem protection	Provides habitats and resources, including nutrients and organic matter, for both terrestrial and aquatic species. Also provide migration corridors and refugia.
Pollination	Riparian zones provide habitat and biotic pollinators for the reproduction of crops and natural vegetation.
Recreation	Riparian areas provide opportunities for a range of outdoor recreational activities
Aesthetic services	Riparian environments deliver numerous amenity benefits, and provide features that compose an attractive landscape

Conceptual framework of the interactions among riparian ecosystem services

In Figure 5 we demonstrate the inherent complexity of ecosystems by mapping the most important ecosystem services delivered within riparian areas. The focus will be on a small subset of services, with an emphasis on those connected to flood mitigation, water quality protection, and recreation. In order to explore how these services interact with the mechanisms that underpin their generation, the typology proposed by Boyd & Banzhaf (2007) is adopted, and ecosystem services are classified into “intermediate processes”, “final services” and “benefits”.

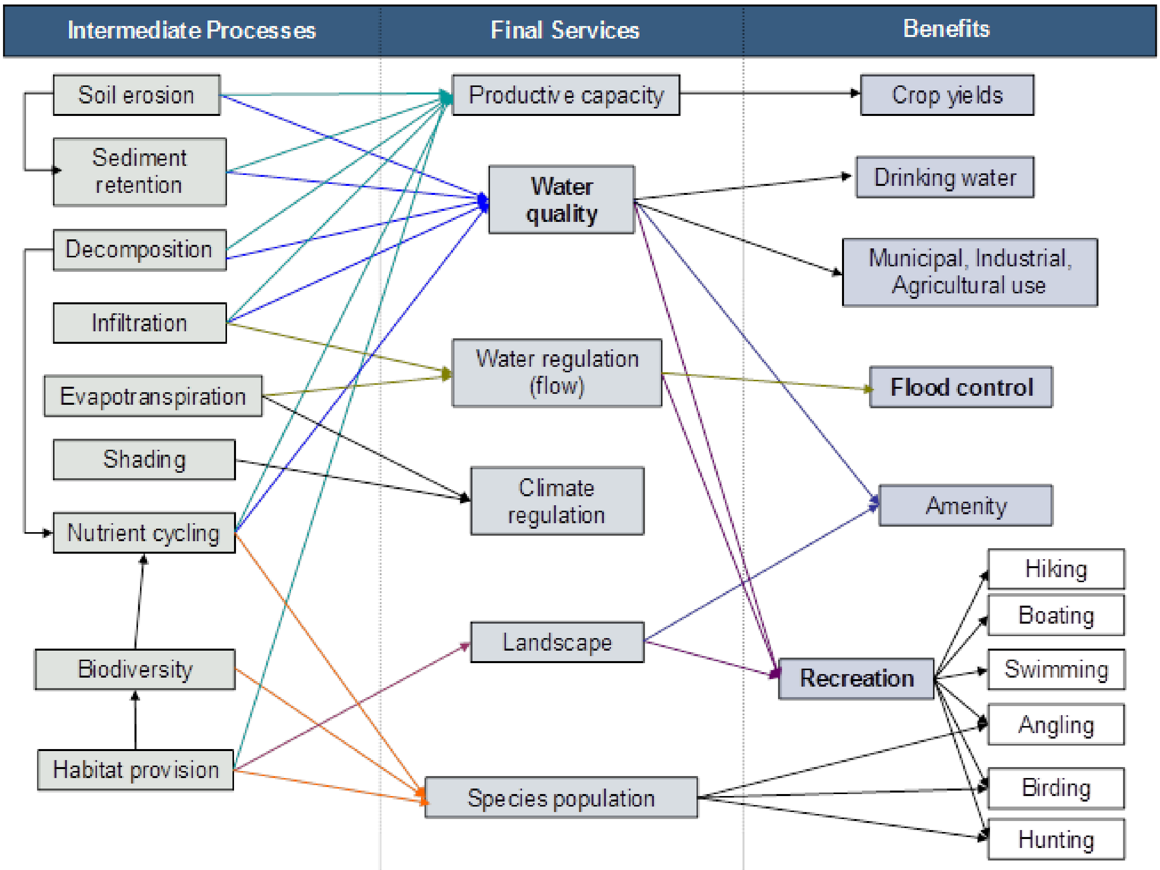

Figure 5 Map of interactions between ecosystem services of riparian areas

Flood control: vegetated riparian zones contribute to flood mitigation either by delaying the passage of downstream floodwater or reducing surface runoff through infiltration or interception of precipitation (Forest Research, 2010). Therefore, these processes act as intermediate functions that regulate hydrological flows and deliver the final benefit of flood mitigation. In that sense, flood control is portrayed as a benefit derived from the final service of water regulation (Boyd & Banzhaf, 2007; Turner et al, 2008), despite some considerations that regard flood alleviation as a final service in itself (Fisher et al., 2009).

Recreation: although often is called an ecosystem service, according to Boyd & Banzhaf (2007) it should be more appropriately regarded as a benefit delivered by final services, like the aesthetic beauty of the surrounding landscape.

Water quality protection: considered in some cases a final service, whereas in other cases acts more as an intermediate process. In the case of drinking water extraction, water is the good that is directly consumed and quality is the final service that affects its delivery. In the case of recreational angling, however, water quality should be considered an ecological process rather than a service itself (Boyd & Banzhaf, 2007). In that case, the final services directly connected to angling are the targeted fish species as well as the surrounding landscape (which contributes to the aesthetic enjoyment of angling); while water quality is more an intermediate function in the provision of those services; the value of water quality is already included in the delivery of an aesthetic landscape and fish species.

Intermediate processes such as carbon regulation, pollination and soil retention, despite their important role in riparian areas, are not represented in Figure 5, since they do not have a direct impact on the services of interest in this study. It should be also noted that ecosystem service provision differs from site to site depending on the particular characteristics of each location (e.g. soil texture, type of vegetation, slope etc). Nonetheless, the main purpose of this study was to contribute to understanding how riparian systems function and what lessons can be learned by the potential synergies and trade-offs that arise, rather than representing the ecosystem service interactions of a real system. Figure 5 shows the range of interdependencies associated with ecosystem service provision. For instance, changes in the infiltration capacity of any riparian area will have effects on multiple services, moderating simultaneously the productive capacity and water quality of the area (i.e. nutrient uptake by plant roots will be decreased) and affecting water regulation (i.e. water storage is decreased and surface runoff increased), in turn impacting on the benefits derived from these services (e.g. flood control, recreational activities, water supply etc.).

Network construction

The initial stage in the development of a BBN is to construct a conceptual model of the problem under investigation by specifying the cause-and-effect relationships among the system components. First, the objectives of the model had to be defined (Kragt, 2009; Jakeman et al., 2006). Water quality and flood protection were selected as the variables desired to be improved by the determined management intervention, and thereby these factors were identified as the end-points of the network. Once the output nodes were defined, the BBN started being developed by identifying the most important variables related to its end-points and establishing the linkages between them. Given that the lower number of nodes a model has, the more easily understood will be by the involved parties (Cain, 2001; Marcot et al., 2006), the challenge to be addressed in that stage was which variables to select in order to provide a realistic representation of a riparian system and at the same time to keep the model as simple as possible.

Having identified all the system variables and their interrelationships, the BBN was created using the Netica package using decision, nature and utility nodes (Norsys, 2003). Decision nodes are associated to factors controlled by decision makers, while utility nodes represent those variables that need to be optimised (i.e. system outputs). In that sense, the management practice that was the subject of this study (i.e. riparian vegetation restoration) was depicted as a decision node, while the end-points of the system were both connected to a utility node. The latter reflects the satisfaction that people will gain from a given decision. All the other variables express conditional probability distribution and were drawn as nature nodes. BBNs that include decision and utility nodes are called influence or decision diagrams (Norsys, 2003; Oliver & Smith, 1990) because they enable reasoning particularly about decision making under conditions of uncertainty (Kjærulff & Madsen, 2005) and hence can estimate the utility that stakeholders will gain from particular decisions.

The network developed in the end of this process is shown in Figure 6. The identification of the system variables was based on literature review of existing BBNs for catchment modelling. The variables used in the model, how they have been defined and assessed in the scope of this study, as well as all the assumptions made in the analysis are summarised in Appendix 1. The restoration of riparian vegetation was examined from a perspective of alternative management practices (i.e. grassland, natural vegetation, mixed, no riparian restoration). Effects of alternative land uses, soil texture, slope, as well as seasonality were also taken into consideration.

On the whole, vegetation coverage in riparian zones is considered to affect the system objectives indirectly, through a chain of effects on multiple ecosystem services. As already noted, a riparian ecosystem is rather a complex system, in which a great range of ecosystem services is performed. However, the analysis was concentrated on flood risk reduction and water quality improvement, therefore only the factors associated with them (directly or indirectly) were taken into account.

Flood risk has been modelled as a variable determined by the level of 'river flow'. It is affected indirectly by the surface 'runoff rate', 'rainfall' rate and 'aquatic vegetation' coverage. The core concept is that riparian vegetation restoration has an indirect impact on flood risk by modifying the 'runoff rate' and thus slowing down the river flow.

Water quality can be defined by a range of biological, chemical, hydrological and morphological characteristics, such as levels of dissolved oxygen, PH, temperature, soluble nutrients content, fish populations etc. (UK NEA, 2011). In this study, Biological Oxygen Demand (BOD) was selected as a water quality indicator. 'Temperature', 'water nutrient concentration' and 'aquatic vegetation coverage' are considered to have an indirect impact on water quality through their effect on BOD. Regardless of their contribution on BOD control, characteristics such as the surrounding atmospheric pressure and the salinity of water (Rauch et al., 1998) regarded as not being affected by the management intervention under consideration. Consequently, they were not included in the system in order to keep the model as simple as possible. Riparian vegetation restoration is considered to affect water quality indirectly, by moderating the 'sediment load' in water and the stream water 'temperature' (via shading).

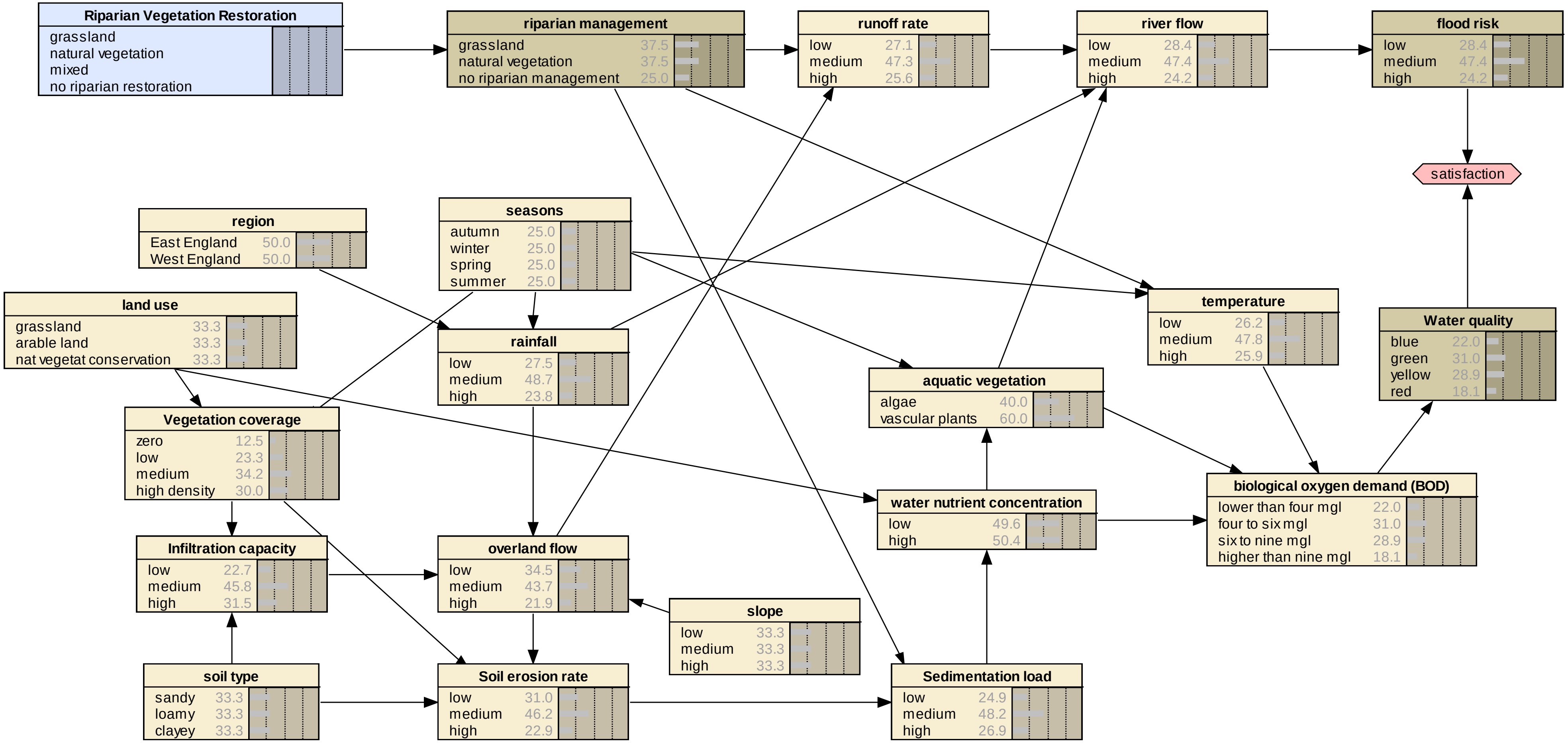

Figure 6 BBN model of riparian management system

Model parameter estimation

Once the conceptual network was designed, the next step was to populate each CPT with probabilistic values. First, each variable was discretised into certain states (see Appendix 1). All the system components were identified as discrete variables, apart from 'satisfaction', which regarded a continuous one ranging from 0 to 100. After discretisation, the model needed to be parameterized by filling the CPT of each node with probability values. Decision and parent nodes are deterministically known and their states are provided by decision makers (Castelletti & Soncini-Sessa, 2007; Cain, 2001); hence, these nodes did not need to be populated.

The probability values in a BBN can be either quantitative evaluations from observed data or information elicited by experts (Pollino et al., 2007). However, given that the analysis was not applied in a particular site (i.e. lack of data measurements) the parameterisation process was based on subjective evaluations of the general patterns of riparian ecosystem functioning drawn from literature and expert knowledge. In other words, the probabilities used in this model were rather generic values, which contained subjectivity and were intended to reflect some contrast between the different states of the variables (i.e. low, medium, high). Although observed data would lead to more robust results, it would also make the model context specific, and thus it would limit the potential to examine alternative scenarios.

The assessment of water quality was based on the water quality ladder proposed by Hime et al. (2009). This is a use-based measure according to which water quality takes different values depending on its suitability for fishing, swimming, boating, or its unsuitability for any use. Each of these ecological categories is associated to 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 Appendix 1.

Moreover, flood risk has been modelled as a deterministic variable whose values are determined by the level of 'river flow'. In general, the lower the 'river flow', the lower the 'flood risk' will be and vice versa.

Water quality and flood protection are critical ecosystem services, which deliver multiple benefits and are associated to both use and non-use values. In this study, it is assumed that both of them contribute equally to the model outcome. In other words, people will be totally satisfied only when both the model objectives have been fully optimised. With a range from 0 to 100, it is assumed that beneficiaries will be totally satisfied when the model outcome is 'low' level of flood risk and 'blue' or 'green' level of water quality. In contrast, they will be totally dissatisfied when 'high' level of flood risk and 'red' level of water quality occur. For all the other possible combinations of states, utility varies according to best judgment (Table 2). Due to time and resource issues these values were developed by the authors. Ideally utility values would be directly obtained through stated preference valuation or some form of weighting exercise (either aggregating individual preferences or through some form of participatory exercise). Instead of examining environmental benefits jointly, most valuation studies tend to consider single types of benefit (Turner et al., 2003). Given the complex interrelationships among ecosystem services and the mechanisms that underlie ecosystem service generation, simply adding up a range of separately obtained values is probably dubious. In that context, a joint approach is may be more appropriate for the valuation of multiple ecosystem services.

Once all CPTs were populated with probability values (see Appendix 2), the model was compiled and the decision network was 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 solutions 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 2 Conditional Probability Table (CPT) of the model utility node

Flood risk	Water quality	Utility
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

Results

To investigate the effectiveness of the management intervention at a regional scale three scenarios based on different regional profiles were examined (Table 3) under alternative land use and riparian management practices. The case of no riparian restoration will be referred as the ‘status quo’, in which is assumed that vegetation in the riparian zone has been degraded. In each scenario, the system parameters being controlled were the ‘region’, ‘land-use’, ‘soil type’, and ‘slope’. Given the state of those factors, the aim of the analysis was: (i) to suggest the optimal riparian management practice in each scenario; and (ii) to compare how the system objectives are affected between the ‘status quo’ and the ‘optimal solution’ of each scenario. Considerations of seasonal changes (associated with the rainfall rate, vegetation coverage and temperature) were also taken into account.

Table 3 Three scenarios examined in this study

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

Scenario A

Under the conditions of Scenario A, a low level of overland flow is implied (i.e. East England: less rainfall; light soils with greater infiltration capacity; low slope). Regardless of any seasonal changes, the findings showed that it was more likely a moderate level of flood risk to be occurred as well as a 'yellow' level of water quality. Only during winter was water quality more likely to be at the 'green' level (i.e. improved). This can be justified by the deteriorating effect of 'arable land' on water quality because of the use of fertilizers. During the winter vegetation coverage of arable land is rather low, thus there is no significant impact of fertilizers on water quality (i.e. improved water quality). On the other hand, during spring and summer when arable land is in high density, water quality reported to be highly impaired.

As shown in Table 4, in scenario A 'natural vegetation' proved to be the most appealing riparian management practice (utility score: 59.37). However, examining all the possible combinations of land uses and riparian management practices, the optimal solution proved to be 'grassland' as land use combined with 'natural vegetation' as riparian management (utility score: 62.83). This illustrates an inherent trade-off between land use and ecosystem services although it should be noted that the positive benefits (largely private) of land use are not incorporated in the model. The results indicate that the optimal solution would affect both the system objectives positively (i.e. the probabilities of 'low' flood risk level and 'blue' level of water quality have been improved). Table 5 illustrates how the management objectives were moderated when the optimal solution was applied.

Table 4 Utility values for Scenario A

	Riparian management			
Land use	**Status quo**	**Grassland**	**Natural vegetation**	**Mixed**
Arable	55.39	56.71	59.37	58.04
Grassland	58.76	61.38	62.83	62.11
Natural vegetation	58.66	61.28	62.78	62.03

Table 5 Changes in the probability of outcomes under the optimal solution for Scenario A

		Status quo (%)	Optimal (%)	Change in pr
Flood risk	**Low**	28.6	35.2	6.6
	Medium	48.6	46.8	-1.8
	High	22.8	18.0	-4.8
Water quality	**Blue**	22.2	25.1	2.9
	Green	31.6	32.2	0.6
	Yellow	28.5	27.0	-1.5
	Red	17.6	15.7	-1.9

Scenario B

In contrast to Scenario A, the conditions of Scenario B (Table 6) imply a higher level of overland flow (i.e. West of England: higher rainfall; heavy soil with low infiltration capacity; medium slope). Under this scenario, a moderate level of flood risk and a 'green' level of water quality were more likely to occur regardless of season, since the effect of the selected land use (i.e. grassland) on temperature or vegetation coverage does not change dramatically in different seasons (see assumptions in Appendix 1).

In this scenario, 'natural vegetation' proved to be the optimal riparian management practice, but combined with the selected land use it was also the optimal solution of the model (satisfaction value: 59.91 – Table 6). Table 7 shows the changes that occurred in the management objectives when this solution was applied.

Table 6 Utility values for Scenario B

	Riparian management			
Land use	**Status quo**	**Grassland**	**Natural vegetation**	**Mixed**
Arable	52.06	53.54	56.36	54.95
Grassland	55.61	58.23	59.91	59.07
Natural vegetation	55.41	58.10	59.80	58.95

Table 7 Changes in the probability of outcomes under the optimal solution for Scenario B

		Status quo (%)	Optimal (%)	Change in pr
Flood risk	**Low**	23.0	29.1	6.1
	Medium	46.7	47.7	1.0
	High	30.4	23.3	-7.1
Water quality	**Blue**	22.5	25.2	2.7
	Green	31.8	32.2	0.5
	Yellow	28.4	27.0	-1.4
	Red	17.4	15.6	-1.8

Scenario C

The conditions of this scenario are similar to Scenario B, but with steeper slope. Again 'Natural vegetation' was the optimal solution, but with less overall utility (score: 59.25 – Table 8) than in scenario B (score: 59.91 – Table 6). Regardless of the steeper slope, in this scenario the optimal solution had a greater improvement in flood reduction (Table 9) than in the previous scenario. This is because in the 'status quo', flood risk is likely to be higher as steeper slopes increase overland flow rates. As a result, riparian vegetation restoration has a greater impact on flood protection and is more effective in areas with steeper slopes.

Table 8 Utility values for Scenario C

	Riparian management			
Land use	**Status quo**	**Grassland**	**Natural vegetation**	**Mixed**
Arable	51.15	52.83	55.79	54.31
Grassland	54.53	57.42	59.25	58.33
Natural vegetation	54.34	57.34	59.15	58.24

Table 9 Changes in the probability of outcomes under the optimal solution for Scenario C

		Status quo (%)	Optimal (%)	Change in pr
Flood risk	**Low**	21.3	27.7	6.4
	Medium	46.2	48.1	1.9
	High	32.5	24.2	-8.3
Water quality	**Blue**	22.3	25.0	2.7
	Green	31.6	32.2	0.6
	Yellow	28.5	27.1	-1.4
	Red	17.6	15.7	-1.9

Discussion

Our analysis explored a combined ecological-economic model using a framework that is suited to an interdisciplinary approach. The model was based on a review of the biophysical relationships between the ecosystem process that lead to final ecosystem services and ultimately benefits that can be valued. Essentially we have unpacked and operationalized the ecosystem services cascade developed by Haines-Young and Potschin (2009). A further step in this operationalization was the introduction of specific management actions. Although the utility values used were determined within the study we believe the relative weights are not unreasonable. However, a number of interesting issues arise with the approach that warrants further investigation.

First, the probabilistic nature of outcomes captured by the BBN approach highlights an important consideration for valuation, namely that the water quality and quantity (flood risk) outcomes of the ecosystem processes represented in the model are not fixed or deterministic but are instead probabilities for different states (e.g. water quality classifications). This has the advantage of reflecting the inherent uncertainty of such outcomes, however this is problematic from a valuation perspective in two respects: i) if using existing values (e.g. benefits transfer) these need to be apportioned across changes in probabilistic outcomes, values would need to be deconstructed across a shifting probability distribution between policy-on and policy-off scenarios; ii) the probabilistic nature of the outcomes raises questions with respect to the formation of values (where those values themselves might be subject to uncertainty) that requires specific exploration.

An implication of the probabilistic outcomes is the potential need to explore whether thresholds or other non-linearities exist that influence preferences and values. For instance in Scenario C the optimal management action (grassland with natural vegetation in riparian areas) sees an increase in probability of low flood risk from 21.3% to 27.7% with a concurrent decline in high risk from 32.5% to 24.2% (see Table 9). The question is whether there is some threshold level reduction in high flood risk that must be crossed to allow the benefits of the increased probability of low flood risk to be realised. For example, the value of any increase in the probability of low flood risk may be contingent on the probability of high flood risk 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. Across multiple ecosystem services, there may be complex and interrelated non-linearities in preferences.

Second, the model as formulated shows little apparent variation in utility values. Again this reflects the probabilistic nature of the outcome, i.e. the utility values are weighted by these probabilities. In specifying the conditional probability tables we tested more extreme probabilities for node outcomes in response to the states of parent nodes but found that this had little effect on model outcomes. This raises the question of to what extent we can say there are significant differences between outcomes. In turn this will be influenced by the statistical properties of any valuation data used (e.g. confidence intervals associated with willingness to pay estimates). Further exploration is needed to determine the sensitivity of the outcomes to variation in both biophysical and economic elements. The effect of mixing probabilistic and deterministic relationships across nodes is also of interest.

We have more recently extended the BBN model (see Figure 7) to incorporate socio-economic factors that might influence values for both water quality (income, water use, availability of substitutes, site amenities) and flood risk (income, proximity). In this extension we have separated out the utility associated with water quality and flood risk, although there are compelling reasons for joint consideration of utility it may also be the case that the benefiting populations are different. Indeed there may be important trade-offs between different ecosystem service benefits. These extensions are not intended to be comprehensive, but will allow us to explore the sensitivity of the BBN to both bio-physical and socio-economic assumptions. Further extensions could include terrestrial ecosystem services (landscape, biodiversity, recreation etc.) and the socio-economic aspects of land manager decision making. 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 extension of the original BBN to more accurately represent both the biophysical and socio-economic elements of system does raise an important further issue. The attraction of the BBN approach is its relative simplicity and flexible data requirements. As the model approaches a degree of 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, then the relative simplicity of the BBN approach may be sufficient to make optimal decisions.

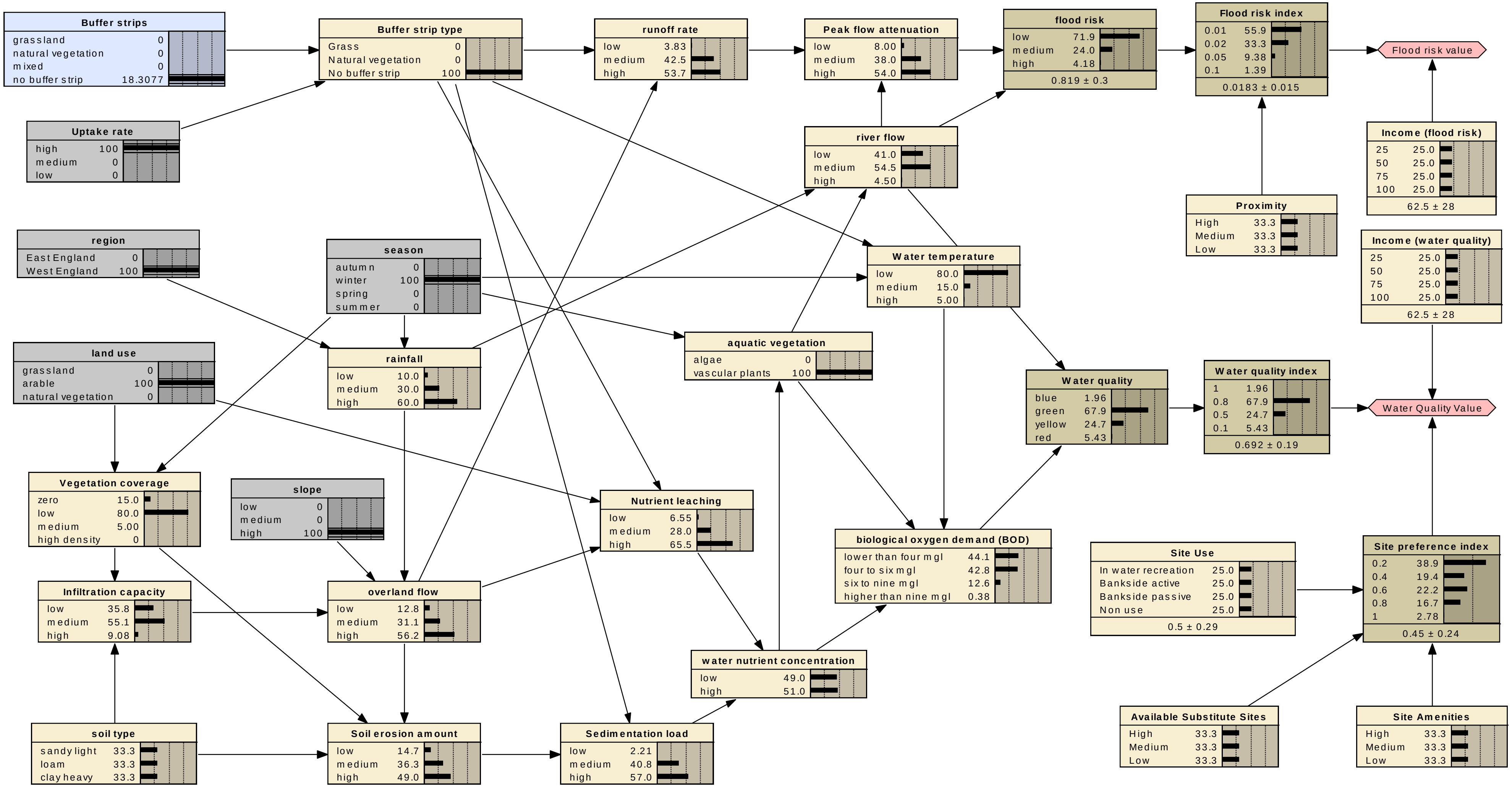

Figure 7 Expanded BBN incorporating socio-economic drivers of preferences

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Appendix 1: Summary of the system variables, their definitions, their states and the assumptions made.

Type of node	Variable	States
DECISION	**Riparian Vegetation Restoration**	grassland, natural vegetation, mixed, no riparian restoration
PARENTS	**Region**	East England, West England
	Land use	grassland, arable land, conservation of natural vegetation
	Seasons	autumn, winter, spring, summer
	Soil type	sandy, loamy, clayey
	Slope	low, medium, high

Type of node	Variable	Definition	States	Dependencies	Assumptions
CHILD NODES	**Riparian management**	The vegetation type and level of coverage determined by the management intervention.	grassland, natural vegetation, no riparian management	Riparian vegetation restoration	
	Rainfall		low, medium, high	region, seasons	West England is assumed to have higher rainfall rates than East England.

Type of node	Variable	Definition	States	Dependencies	Assumptions
CHILD NODES	**Vegetation coverage**	The proportion of ground surface covered by vegetation.	zero, low, medium, high	land use, 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 • Nat. 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 a high water permeability, while clay is more resistent to water infiltration (Marrs, 1993).
	Overland flow	Water that flows across the land after rainfall, either before it enters a watercourse, or after it leaves a watercourse as floodwater. It does not include the water volume intercepted by vegetation or infiltrated by soil and plants.	low, medium, high	rainfall, infiltration capacity, slope	• The higher the rainfall rate, the lower the infiltration capacity and the 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 (Craft & Casey, 2000). • 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).

Type of node	Variable	Definition	States	Dependencies	Assumptions
CHILD NODES	**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 surfases facilitate greater rates of sediment deposition due to their high root density (Lyons et al., 2000; Trimble, 1997; Allmendinger et al., 1999). • 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.) (Craft & Casey, 2000). • Nutrient plant uptake is assummed 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 be occurred 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		low, medium, high	riparian management, seasons	Natural vegetation has a decreasing effect on temperature via shading (Arizpe et al., 2008).

Type of node	Variable	Definition	States	Dependencies	Assumptions
CHILD NODES	**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 oxyden depletion). • Characteristics such as the surrounding atmosperic 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 (Hime et al., 2009).	blue, green, yellow, red	BOD	Each water quality category was converted into a BOD level, as following (Hime et al., 2009): • blue = 0 - 4 mgl-1, • 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.

Type of node	Variable	Definition	States	Dependencies	Assumptions
CHILD NODES	**River flow**		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		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).

Appendix 2 – Conditional Probability Tables (CPTs) of system variables.

Conditions		Probability of VEGETATION COVERAGE			
land use	seasons	zero	low	medium	high density
grassland	autumn	5	15	50	30
grassland	winter	5	15	50	30
grassland	spring	5	15	30	50
grassland	summer	5	15	30	50
arable land	autumn	50	30	15	5
arable land	winter	30	50	15	5
arable land	spring	5	15	50	30
arable land	summer	5	15	30	50
natural veg. conservation	autumn	15	30	50	5
natural veg. conservation	winter	15	50	30	5
natural veg. conservation	spring	5	15	30	50
natural veg. conservation	summer	5	15	30	50

Conditions		Probability of INFILTRATION CAPACITY		
soil type	Vegetation coverage	low	medium	high
sandy	zero	30	60	10
sandy	low	10	60	30
sandy	medium	10	30	60
sandy	high density	10	30	60
loamy	zero	60	30	10
loamy	low	30	60	10
loamy	medium	10	60	30
loamy	high density	10	30	60
clayey	zero	60	30	10
clayey	low	60	30	10
clayey	medium	30	60	10
clayey	high density	10	60	30

Conditions		Probability of RAINFALL		
seasons	region	low	medium	high
autumn	East England	30	60	10
autumn	West England	10	60	30
winter	East England	10	60	30
winter	West England	10	30	60
spring	East England	30	60	10
spring	West England	10	60	30
summer	East England	60	30	10
summer	West England	60	30	10

Conditions		Probability of WATER NUTRIENT CONCENTRATION	
sedimenta-tion load	land use	low	high
low	grassland	70	30
low	arable land	30	70
low	natural vegetation	70	30
medium	grassland	70	30
medium	arable land	30	70
medium	natural vegetation	70	30
high	grassland	30	70
high	arable land	30	70
high	natural vegetation	30	70

Conditions		Probability of SEDIMENTATION LOAD		
soil erosion rate	riparian management	low	medium	high
low	grassland	60	30	10
low	natural vegetation	60	30	10
low	no riparian management	30	60	10
medium	grassland	30	60	10
medium	natural vegetation	10	60	30
medium	no riparian management	0	70	30
high	grassland	10	60	30
high	natural vegetation	10	30	60
high	no riparian management	0	0	100

Conditions		Probability of AQUATIC VEGETATION	
seasons	water nutrient concntration	algae	vascular plants
autumn	low	30	70
autumn	high	30	70
winter	low	30	70
winter	high	30	70
spring	low	30	70
spring	high	70	30
summer	low	30	70
summer	high	70	30

Conditions		Probability of TEMPERATURE		
seasons	riparian management	low	medium	high
autumn	grassland	10	60	30
autumn	nat. vegetation	30	60	10
autumn	no riparian management	10	60	30
winter	grassland	60	30	10
winter	nat. vegetation	60	30	10
winter	no riparian management	60	30	10
spring	grassland	10	60	30
spring	nat. vegetation	30	60	10
spring	no riparian management	10	60	30
summer	grassland	10	30	60
summer	nat. vegetation	10	60	30
summer	no riparian management	10	30	60

Conditions			Probability of BIOLOGICAL OXYGEN DEMAND (BOD)			
water nutrient concetration	temperature	acquatic vegetation	< 4 mgl-1	4 to 6 mgl-1	6 to 9 mgl-1	> 9 mgl-1
low	low	algae	5	50	30	15
low	low	vascular plants	50	30	15	5
low	medium	algae	5	50	30	15
low	medium	vascular plants	50	30	15	5
low	high	algae	5	30	50	15
low	high	vascular plants	30	50	15	5
high	low	algae	5	15	50	30
high	low	vascular plants	30	50	15	5
high	medium	algae	5	15	30	50
high	medium	vascular plants	15	30	50	5
high	high	algae	5	15	30	50
high	high	vascular plants	5	15	50	30

Conditions		Probability of RUNOFF RATE		
overland flow	catchment management	low	medium	high
low	grassland	60	30	10
low	natural vegetation	60	30	10
low	no riparian management	40	50	10
medium	grassland	10	60	30
medium	natural vegetation	30	60	10
medium	no riparian management	0	70	30
high	grassland	10	30	60
high	natural vegetation	10	60	30
high	no riparian management	0	10	90

Conditions	Probability of RIPARIAN MANAGEMENT		
riparian vegetation restoration	grassland	natural vegetation	no riparian management
grassland	100	0	0
natural vegetation	0	100	0
mixed	50	50	0
no riparian managemnt	0	0	100

Conditions	Probability of WATER QUALITY
BOD	
< 4 mgl-1	blue
4 to 6 mgl-1	green
6 to 9 mgl-1	yellow
> 9 mgl-1	red

Conditions	Probability of FLOOD RISK
river flow	
low	low
medium	medium
high	high

Conditions			Probability of SOIL EROSION AMOUNT		
overland flow	soil type	Veget. Coverage	low	medium	high
low	sandy	zero	60	30	10
low	sandy	low	60	30	10
low	sandy	medium	60	30	10
low	sandy	high density	60	30	10
low	loamy	zero	60	30	10
low	loamy	low	60	30	10
low	loamy	medium	60	30	10
low	loamy	high density	60	30	10
low	clayey	zero	60	30	10
low	clayey	low	60	30	10
low	clayey	medium	60	30	10
low	clayey	high density	60	30	10
medium	sandy	zero	10	60	30
medium	sandy	low	10	60	30
medium	sandy	medium	10	60	30
medium	sandy	high density	10	60	30
medium	loamy	zero	10	60	30
medium	loamy	low	10	60	30
medium	loamy	medium	30	60	10
medium	loamy	high density	30	60	10
medium	clayey	zero	10	60	30
medium	clayey	low	10	60	30
medium	clayey	medium	30	60	10
medium	clayey	high density	30	60	10
high	sandy	zero	10	30	60
high	sandy	low	10	30	60
high	sandy	medium	10	30	60
high	sandy	high density	10	60	30
high	loamy	zero	10	30	60
high	loamy	low	10	30	60
high	loamy	medium	10	60	30
high	loamy	high density	10	60	30
high	clayey	zero	10	30	60
high	clayey	low	10	30	60
high	clayey	medium	10	60	30
high	clayey	high density	10	60	30

Conditions			Probability of OVERLAND FLOW		
infiltration capacity	rainfall	slope	low	medium	high
low	low	low	60	30	10
low	low	medium	60	30	10
low	low	high	60	30	10
low	medium	low	30	60	10
low	medium	medium	10	60	30
low	medium	high	10	30	60
low	high	low	10	30	60
low	high	medium	10	30	60
low	high	high	10	30	60
medium	low	low	60	30	10
medium	low	medium	60	30	10
medium	low	high	60	30	10
medium	medium	low	30	60	10
medium	medium	medium	30	60	10
medium	medium	high	10	60	30
medium	high	low	10	60	30
medium	high	medium	10	30	60
medium	high	high	10	30	60
high	low	low	60	30	10
high	low	medium	60	30	10
high	low	high	60	30	10
high	medium	low	60	30	10
high	medium	medium	60	30	10
high	medium	high	30	60	10
high	high	low	30	60	10
high	high	medium	30	60	10
high	high	high	10	60	30

Conditions			Probability of RIVER FLOW		
runoff rate	**rainfall**	**aquatic vegetation**	**low**	**medium**	**high**
low	low	algae	60	30	10
low	low	vascular plants	60	30	10
low	medium	algae	30	60	10
low	medium	vascular plants	60	30	10
low	high	algae	10	60	30
low	high	vascular plants	30	60	10
medium	low	algae	30	60	10
medium	low	vascular plants	60	30	10
medium	medium	algae	10	60	30
medium	medium	vascular plants	30	60	10
medium	high	algae	10	30	60
medium	high	vascular plants	10	60	30
high	low	algae	10	60	30
high	low	vascular plants	30	60	10
high	medium	algae	10	30	60
high	medium	vascular plants	10	60	30
high	high	algae	10	30	60
high	high	vascular plants	10	30	60