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Bayesian Belief Networks as a tool for evidence-based conservation management

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Summary

Effective conservation management is dependent on accessing and integrating different forms of evidence regarding the potential impacts of management interventions. Here, we explore the application of Bayesian Belief Networks (BBN), which are graphical models that incorporate probabilistic relationships among variables of interest, to evidence-based conservation management. We consider four case studies, namely: (i) impacts of deer grazing on saltmarsh vegetation; (ii) impacts of burning on upland bog vegetation; (iii) control of the invasive exotic plant *Rhododendron ponticum*; and (iv) management of lowland heathland by burning. Each of these themes is currently a significant conservation issue in the UK, and yet the potential outcomes of management interventions are poorly understood. Through these examples, we demonstrate that BBNs can be used to integrate and explore evidence from a variety of sources, including expert opinion and quantitative results from research investigations. Incorporation of such information in BBNs enables different sources of evidence to be compared, the potential impacts of management interventions to be explored and management trade-offs to be identified. BBNs also offer a highly visual tool for communicating the uncertainty associated with potential management outcomes to conservation practitioners, and they can also be readily updated as new evidence becomes available. Based on these features, we suggest that BBNs have outstanding potential for supporting evidence-based approaches to conservation management.

Introduction

Recent years have witnessed growing interest in evidence-based approaches to conservation,

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reflecting widespread concern that much conservation practice is based on tradition or the experience of practitioners, rather than on the results of scientific research (Pullin & Knight, 2001, 2003; Pullin, Knight, Stone, & Charman, 2004; Sutherland, Pullin, Dolman, & Knight, 2004). This process has been inspired by the 'effectiveness revolution' that has occurred in medicine during the past 20 years, aimed at incorporating the results of medical research into medical practice (Egger, Smith, & Altman, 2003; Stevens & Milne, 1997). Evidence-based frameworks have subsequently developed in other areas of public policy, including psychology (Petticrew, 2001), education (Nye, Schwartz, & Turner, 2005), social welfare (Stagner, Ehrle, & Reardon-Anderson, 2003), and criminology (Gadon, Cooke, & Johnstone, 2005).

The common element of these evidence-based frameworks is the process of 'systematic review'. All reviews are retrospective, observational research studies and are therefore subject to systematic and random error (Cook, Mulrow, Haynes, & Brian, 1997). The quality of a review therefore depends on the extent to which scientific methods have been used to minimise error and bias. Systematic reviews locate data from published and unpublished sources, critically appraise methods using pre-defined criteria and synthesise evidence to provide empirical answers to research questions. They differ from conventional reviews in that they follow a strict methodological and statistical protocol making them more comprehensive, minimising the chance of bias and improving transparency, repeatability, and reliability. Rather than reflecting the views of authors or being based on a (possibly biased) restricted sample of literature, they provide a comprehensive assessment and summary of available evidence (Cook et al., 1997).

Guidelines for the production of ecological systematic reviews have been established (Pullin & Stewart, 2006) and the results of the first systematic reviews of conservation evidence are now becoming available (Stewart, Coles, & Pullin, 2005; Tyler, Pullin, & Stewart, 2006). Further unpublished and ongoing reviews can be obtained from http://www.cebc.bham.ac.uk/. These reviews have focused on the identification of experimental and monitoring evidence most analogous to the controlled trials, observational studies, and diagnostic test trials synthesised in medical meta-analyses (Egger et al., 2003). Criteria for inclusion in such meta-analyses include the presence of a quantitative comparator before and after application of an experimental treatment, and/or between experimental treatment and controls.

These initial systematic reviews of conservation evidence have highlighted the fact that experimental investigations meeting these criteria are relatively rare, even for management approaches that are widely used. For example, of 317 articles with relevant titles concerning the impact of burning on blanket bog, only eight (2.5%) had comparators allowing quantitative synthesis (Stewart, Coles, & Pullin, 2004). Similarly, reviews regarding burning of dry heathland, the impact of wind farms on bird abundance, and bracken control utilised 1.7%, 12%, and 4.2% of material with relevant titles respectively (www.cebc.bham.ac.uk). As a consequence, the results of meta-analyses can lack statistical power (as a consequence of small sample sizes), especially when numerous effect modifiers are included in the analysis. Despite the rigour underlying the methodology, review outcomes are therefore often highly tentative, allowing few firm conclusions to be drawn. Such reviews can clearly be used to identify experimental knowledge-gaps, allowing priority areas for needs-led research to be identified. However, additional forms of evidence or information exist, and can be retrieved using systematic review methods (for example, published studies that fail to meet the criteria for inclusion in a meta-analysis, perhaps because a suitable comparator or control was not included). The inclusion of such information could increase the utility of reviews to practitioners, allowing tentative management recommendations to be made on all the available evidence rather than just a subset.

Methods are therefore required that would enable additional forms of evidence to be incorporated into analyses. Although experimental investigations of conservation management interventions are relatively few, much information is collected during environmental monitoring activities. Conservation practitioners often also possess deep working knowledge of the ecological communities with which they are familiar, based on the results of their practical experience and anecdotal observations of the outcome of management interventions. An analytical approach is needed that would enable such types of information to be analysed together with more rigorous scientific evidence, in an integrated manner. Ideally, the outcomes of such analyses should be made available in a form that can readily support decision-making, including the development and implementation of appropriate conservation policies.

We propose that Bayesian Belief Networks (BBN) offer a uniquely powerful tool to address these problems, by providing a structured combination of diverse lines of evidence. BBNs have developed at the interface between statistics, applied artificial intelligence, and expert system development (Pearl, 1986, 1988). BBNs are graphical models that encode probabilistic relationships among variables of interest (Bøttcher & Dethlefsen, 2003; Heckerman, 1996), and may be considered as tools for graphically representing the relationships among a set of variables (Castillo, Gutierrez, & Hadi, 1997). A BBN comprises a network of nodes connected by directed links, with a probability function attached to each node (Jensen, 2001). BBNs are therefore statistical models of a domain. The network of a BBN is referred to as a directed acyclic graph (DAG), which is used to model a domain containing uncertainty, and therefore provides a tool for reasoning under uncertainty (Jensen, 2001). This uncertainty can arise owing to an imperfect understanding of the domain, incomplete knowledge of the state of the domain, randomness in the mechanisms governing the behaviour of the domain, or any combination of these.

Bayesian networks evolved in the early 1990s, based on a deep body of theory developed for graphical models in general. Statistical graphical models have a history that can be traced back to Wright (1934) who developed them in the context of path analysis. In order to develop the directed graphs used in BBNs, several challenges had to be overcome, as described by Spiegelhalter, Dawid, Lauritzen, and Cowell (1993). The calculus of probability underlying Bayesian networks was at one time considered to be both epistemologically inadequate and computationally infeasible for complex domains. The principal difficulty was that complex applications require the specification of what can be huge joint probability distributions. In addition, evidence propagation within such a framework requires the computation of probabilities for events of interest that are conditional on what could be arbitrary configurations of other variables. The computational hurdles have now been overcome, due in large part to the advances derived from the seminal work of Pearl (1986, 1988). Later, Pearl (1995) showed how graphical models can be used for causal inference, thus strengthening the underlying justification behind contemporary applications of BBNs that use expert knowledge to determine both the structure and parameters of the networks (Spiegelhalter & Cowell, 1993). As a result of software developments and the increased availability of computing power, construction of large BBNs is now feasible (Neil, Fenton, & Nielson, 2000), and the method can readily be implemented on a personal computer.

Contemporary software programs for implementing BBNs are extremely flexible. BBNs can be built directly from knowledge of the domain of interest. Alternatively, it is now possible for both the structure and the parameters of a BBN to be learnt directly from a data set, and for this reason, the method is widely used for automated data mining, particularly for market research. However, many of the issues identified by conservation managers (see Sutherland et al., 2006) do not generate the large quantities of replicated data needed for such data mining. In such circumstances, it is best to view BBNs as decision-support tools helpful for combining expert knowledge with available empirical data (Marcot, Holthausen, Raphael, Rowland, & Wisdom, 2001).

Bayesian analytical techniques have received growing interest from ecological researchers since publication of a special edition of *Ecological Applications* in 1996 (Crome, Thomas, & Moore, 1996; Ellison, 1996; Gertner & Zhu, 1996). Typically, Bayesian statistics are used to find parameter values when the stochastic component of a model is represented by one or more continuous probability density functions. The directed acyclical graphs used to represent these models can follow the same formalisms as BBNs. However, examples of the use of BBNs in ecology or resource management are few. Examples include predicting density of mountain aspen suckers (Haas, 1991), assessing population trends in aquatic and terrestrial vertebrate species (Marcot et al., 2001; Rieman et al., 2001), integrated water resource planning (Bromley, Jackson, Clymer, Giacomello, & Jensen, 2005), social aspects of resource management (Cain, Batchelor, & Waughray, 1999) and assessing the impact of commercialising non-timber forest products on livelihoods (Newton et al., 2006). We are not aware of any previous attempt to explore the potential value of BBNs specifically to evidence-based conservation management.

In this paper, we explore the application of BBNs to evidence-based conservation management through consideration of four case studies. These illustrate a range of different conservation issues and management options, and vary in the type and quality of evidence available. In two of the examples, BBNs are constructed that incorporate the results of systematic surveys of the conservation literature, and in one case, this form of evidence is contrasted with that based on the experience of conservation practitioners. In each case, BBNs are used to assess the potential impact of management interventions on some outcome or variable of conservation interest. The four examples are: (i) impacts of deer grazing on saltmarsh vegetation; (ii) impacts of burning on upland bog vegetation; (iii) control of *Rhododendron ponticum*;

and (iv) management of lowland heathland by burning. Each of these themes is currently a significant conservation issue in the UK, and yet the potential outcome of management interventions is uncertain.

Details of case studies

Case 1. Impacts of deer grazing on saltmarsh vegetation

Poole harbour, located on the south coast of England, is internationally recognised as a site for large numbers of wintering wildfowl and waders, which feed and roost on intertidal mudflats and saltmarshes. The saltmarshes are also important during the early summer months as breeding sites for waders, gulls, and terns. In consequence, the harbour has been designed as a Special Protection Area (SPA) under the European Birds Directive and as a Ramsar site. The saltmarsh vegetation is dominated by *Spartina anglica*, a species that has recently evolved through hybridisation between the native *Spartina maritima* and the American introduction *Spartina alternifolia*.

One of the main conservation issues relating to saltmarshes in Poole harbour is the impact of grazing by an exotic herbivore, Sika deer (*Cervus nippon*), which has rapidly increased in number since its introduction to the area at the turn of the 20th century. Some 3000 animals are now present in the area (Diaz, Pinn, & Hannaford, 2005). To assess the impacts of deer grazing on saltmarsh vegetation and the invertebrate communities on which wading birds feed, Diaz et al. (2005) established ten fenced exclosures in random positions on heavily grazed areas at Arne, a nature reserve on the western edge of Poole harbour. Each exclosure measured 2.5×2.5 m and was constructed using 2 m wooden stakes inserted to a depth of 1 m at each corner, enclosed in steel mesh. Twenty randomly located unexclosed plots were also established in heavily deer grazed areas, and a further 30 were established in lightly or ungrazed areas. Vegetation was surveyed by recording the percentage cover of each plant species present. Above-ground vegetation volume was assessed by visually recording the percentage occupancy of slices of the plot cuboid at 10 cm height intervals. The abundance of the macro-invertebrate fauna of the saltmarsh was also assessed using a 20 cm diameter augur drill, which was used to obtain three sediment core samples per plot to a depth of 10 cm. The cores were sieved through a 0.5 mm sieve and retained invertebrate fauna were identified to species level. Full details of the data obtained are described by Diaz et al. (2005).

Case 2. Impacts of burning on upland bog vegetation

Blanket bogs are an upland community of high conservation value, which are burned by land managers primarily in areas managed for sport shooting. However, the effects of burning on floristic composition and structure are uncertain, and for this reason, a systematic review of the available evidence was performed to determine whether or not burning is compatible with nature conservation objectives (Stewart et al., 2004, 2005). Studies were included if they fulfilled the following relevance criteria: (i) focus on upland British blanket bog or wet heath; (ii) include burning as a factor or variable; (iii) measure or monitor the effects of burning on favourable condition criteria; (iv) present primary data and include a quantitative comparison before and after intervention, and/or between treatment and control.

A comprehensive literature search was performed, capturing 24,484 database 'hits' of which only 13 were of sufficient quality and relevance to meet the inclusion criteria (above). Five of these articles reported on the same datasets and were excluded. The remaining eight articles reported on the results of 11 datasets. The outcome measure ('favourable condition') precluded meta-analysis and the small sample size hindered the interpretation of alternative multivariate analysis. In contrast to traditional narrative reviews (DEFRA, 2005; Glaves et al., 2005; Tucker, 2003), the systematic review highlighted a major knowledge-gap, suggesting that the outcome of burning is uncertain. Burning may either degrade blanket bog or be neutral in effect with controlled trials more indicative of degradation than site comparisons. However, interpretation of the results was constrained by the small sample size and variable timescales of the studies. Although the review identified a knowledge-gap, it was not possible to make simple predictions based on a combined index of favourable conservation status or to express the results in a way that captured the high uncertainty of the domain and effectively communicated this to decision-makers (DEFRA, 2005).

Case 3. Control of *Rhododendron ponticum*

R. ponticum is an invasive exotic species in many countries, including the UK, Ireland, Belgium, and France, where it alters entire semi-natural communities through its vigorous spread, and poses a threat to native flora and fauna (Cross, 1975). Control is essential if the conservation value of communities such as oak woodland and lowland heathland are to be successfully maintained. Commonly used interventions include application of various herbicides (such as Imazapyr, Glyphosate, Triclopyr, and Metsulfuron-methyl), either by direct application or stem injection, or by application following stem cutting. Alternatively, purely mechanical methods, such as cutting, hand-pulling, or winching (of larger individuals) may be used.

The effectiveness of these interventions for *R. ponticum* control was evaluated using systematic review methods (Tyler et al., 2006). Inclusion required articles to: (i) contain data on a *R. ponticum*; (ii) assess an intervention applied with the purpose of reducing the population size; (iii) describe an outcome including data on the change in cover, stand density, frequency or biomass of *R. ponticum*; and (iv) include a suitable comparator/control (untreated or uncontrolled *R. ponticum*). Of 196 references identified with relevant titles, only 12 contained data suitable for meta-analysis. Meta-analysis of 'best available evidence' demonstrated that application of Glyphosate following cutting, or direct application of metsulfuron-methyl or Imazapyr can effectively reduce *R. ponticum* abundance. However, the analysis suffered from a lack of statistical power resulting from the small sample size.

Case 4. Management of lowland heathland by burning

Lowland heathland is a priority habitat for nature conservation throughout north-west Europe. At the European level, heathlands are included in the Habitats Directive 92/43/EEC as "a natural habitat type of community interest whose conservation requires the designation of special conservation areas". Approximately 70,000 ha of lowland heath remains in the UK, which represents about 16% of its former extent. A UK Habitat Action Plan has been developed for lowland heathland, and a number of species associated with lowland heathlands are the focus of national Biodiversity Action Plans (BAPs). One of the most important causes of the loss of heathland habitat has been a change in the pattern of land use (Webb, 1986). Specifically, there has been a widespread decline in traditional use of heathlands, which included light grazing, controlled burning and cutting of vegetation for use as fuel and animal fodder (Webb, 1998). As a result, many heathlands have reverted to scrub or woodland through a process of natural succession. This process now represents one of the main threats to communities of plants and animals associated with heathland habitats (Rose, Webb, Clarke, & Traynor, 2000).

Current management responses to this problem include the use of fire, cutting of vegetation and reintroduction of grazing. The impacts of different management interventions on lowland heaths were reviewed by Bullock and Pakeman (1997), who found that there have been no definitive studies of the impacts of burning on heathland species (general overviews are provided by Webb (1986) and Gimingham (1992)). Controlled burning removes most above-ground biomass, but generally leaves the litter layer intact, which creates areas of bare ground that are recolonised by plant species primarily by resprouting. A key objective of the use of fire as a management tool on heathland is to create a mosaic of heather (*Calluna vulgaris*) patches of different age, as heathland species vary with respect to their association with heather stands of different ages. Most of the information that is available regarding the impacts of different management interventions on lowland heaths is based on studies that were insufficiently replicated and/or with insufficient monitoring (Lake, Bullock, & Hartley, 2001). Furthermore, few attempts have been made to compare fire with alternative management approaches, such as cutting or grazing (Lake et al., 2001). This highlights the need for a critical review of the evidence, to identify the conditions under which burning is likely to be most effective as a management approach, and to determine the relative impacts of burning compared to alternative management interventions.

A systematic review was performed (G. Stewart et al. unpublished data) using a suite of appropriate search terms. This identified 3431 potentially relevant references in bibliographic databases, of which 92 (<3%) were found to be relevant to management of lowland heath. Only 10 of these had appropriate comparators; three examined the impacts of burning, two examined vegetation cutting, three examined grazing, and a further two examined the impacts of grazing and burning in combination.

Construction of BBNs

BBNs were constructed using Hugin Developer 6.3, a commercial software package developed and distributed by Hugin Expert A/S, Aalborg, Denmark (http://www.hugin.com/). In each case, variables were represented as nodes in the networks, and connected by arrows (directed links), which are indications of conditional dependence. A link between two nodes, from node A (parent node) to node B (child node), indicates that A and B are functionally related, or that A and B are statistically correlated. Each child node (i.e. a node linked to one or more parents) contains a conditional probability table (CPT). The CPT gives the conditional probability for the node being in a specific state given the configuration of the states of its parent nodes. When networks are compiled, Bayes' theorem is applied according to the values in the CPT, so that changes in the probability distribution for the states at node A are reflected in changes in the probability distribution for the states at node B.

A BBN can be explored by changing the states of the nodes (or variables) incorporated within the model. When the state of a variable is known, it is said to be *instantiated* (Jensen, 2001). Once a node has been instantiated, then this will influence the probabilities associated with the states of other nodes to which it is linked, according to the values in the CPTs.

Case 1. Impacts of deer grazing on saltmarsh vegetation

In this case study, the variables included Sika deer density, *Spartina* volume, floristic diversity of vegetation, and abundance of macro-invertebrates (Fig. 1). In each of these nodes, CPTs were based on the experimental data provided by Diaz et al. (2005). In this investigation, data were not collected on redshank (*Tringa tetanus*) nest sites, numbers, or reproductive success, but these have been included in the network so that the potential impacts of deer grazing on bird populations can be visualised. CPTs for these nodes were based on expert knowledge.

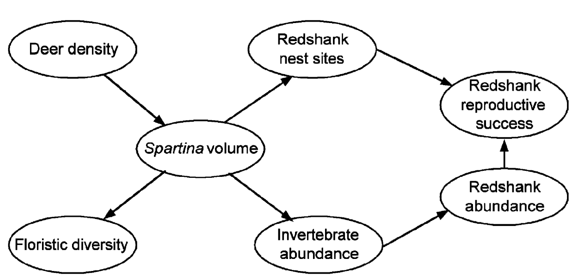

Figure 1. Bayesian Belief Network (BBN) constructed for examining the potential impact of grazing by Sika deer on saltmarsh communities (Case 1, see text). The ellipses (nodes) represent variables, and the arrows represent conditional dependencies between the nodes.

The network was instantiated for two different deer densities, zero (equivalent to fenced experimental plots) and 1.76 ha^{-1}, the estimated maximum density recorded in the survey. This enables the potential impact of a change in deer density on saltmarsh communities to be evaluated. The potential response of each variable is illustrated as a probability distribution associated with the node states. For example, at zero deer density, the most likely state of the variable '*Spartina* volume' is 1.5–2 m^3, with a probability of 0.53. However, under the maximum deer density, the most likely value of '*Spartina* volume' is 0–0.5 m^3, with a probability of 0.79. This uncertainty in terms of outcome is the result of variation in the data obtained from different experimental plots. Such variation is typical in field-based ecological investigations, and this example illustrates how such uncertainty can be incorporated in a BBN and communicated to the user. The example also illustrates how the BBN can be used as a model to evaluate the potential impact of a change in deer grazing on some variable of interest. For example, the BBN predicts that invertebrate abundance is likely to increase as a result of an increase in deer density, based on a change in the most likely category from 0–20 to 20–40 individuals per sample core. The effect is mediated via an effect on volume of *Spartina*, as illustrated in the network (Fig. 2a and b).

One of the main issues facing conservation managers in Poole harbour is the need to define an appropriate deer density as a management objective. Use of the BBN approach highlights a potential trade-off in terms of impacts of deer grazing on populations of redshank, a species of conservation concern. While redshank abundance is likely to be influenced by availability of macro-invertebrates, on which they depend for food, redshank reproductive success is also likely to depend on the availability of nesting sites. As the species prefers *Spartina* tussocks of large volume in which to nest, an increase in deer grazing could simultaneously result in an increase in food availability and a decrease in the availability of nesting sites. BBNs offer a tool by which such trade-offs can be explored, enabling the potential impacts of different management interventions to be evaluated.

Case 2. Impacts of burning on upland bog vegetation

A BBN was constructed for the impact of burning on different plant species, by representing the

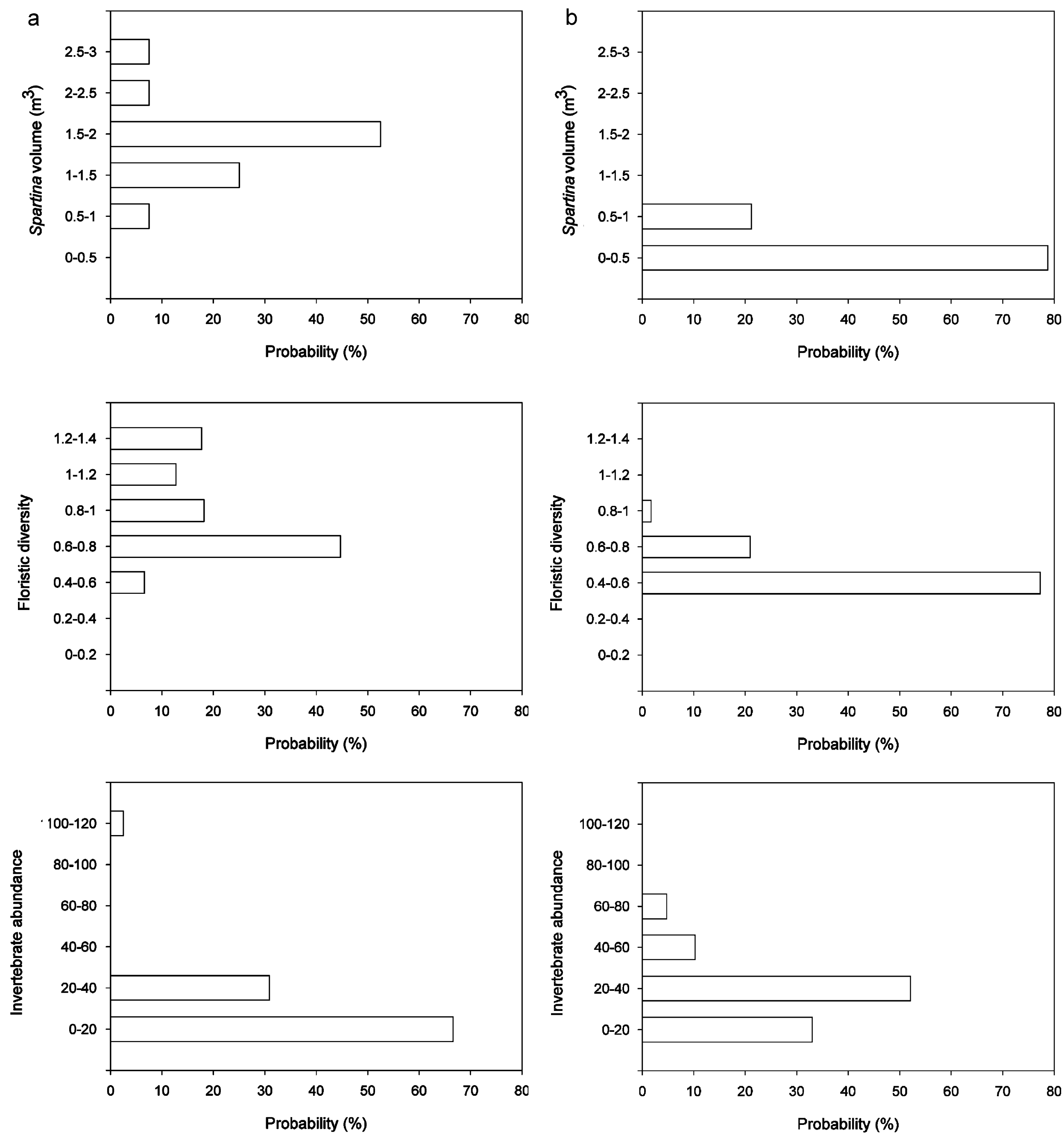

Figure 2. Results obtained from BBN constructed for examining the potential impacts of grazing by Sika deer on saltmarsh communities (Case 1). (a) Zero deer density (obtained by instantiating the 'Deer density' node with the zero density state). Bars and associated numerical values represent probabilities associated with each node state, for the variables *Spartina* volume, floristic diversity, and invertebrate abundance. (b) Deer density 1.76 ha^{-1} (obtained by instantiating the 'Deer density' node with the 1.76 density state). Bars and associated numerical values represent probabilities associated with each node state, for the variables *Spartina* volume, floristic diversity, and invertebrate abundance.

decision regarding whether or not to burn as a node ('Burn'), with two states ('Yes'/'No') (Fig. 3). Potential impacts on the abundance of different plant species were represented by creating a node for each species, with three states in each case: 'Increase', 'No change' and 'Decrease'. CPTs for these nodes were completed by reference to the results of the eight studies (with 11 datasets) that met the selection criteria in the systematic review. Each study was scored with respect to the change in abundance (either frequency or cover) of each individual plant species attributable to burning,

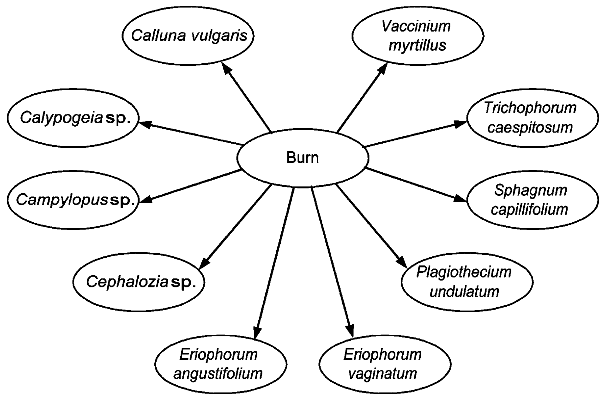

Figure 3. BBN constructed for assessing the impact of burning on upland bog vegetation (Case 2).

identified by comparison of measures obtained from burnt areas with those obtained from unburnt areas.

The potential impact of burning on bog plant species was explored by instantiating the 'Burn' node with the state 'Yes'. In response, changes in the abundance of individual species are represented as probability distributions associated with the node states 'Decrease', 'No change', and 'Increase' (Fig. 4). As with the previous case study, presentation of a probability distribution highlights the contrasting results obtained by different studies, and again indicates how BBNs can be used to illustrate the uncertainty surrounding the potential outcomes of a management intervention. The method also highlights the contrasting responses of different species to burning. For example, burning was associated with an increase in abundance of *Calluna vulgaris* in 64% of studies, but a decrease in abundance in 27% of studies. In contrast *Eriophorum vaginatum* increased in only 18% of studies, but decreased in 55%. A striking feature of these results is that no species displayed the same response to burning in all studies, indicating that the favourable condition outcome measure used in the original systematic review masks important trade-offs between individual species.

Case 3. Control of *Rhododendron ponticum*

Evidence was collected on 13 different methods of controlling *R. ponticum* (Table 1). In this case, evidence was drawn from three different sources: (i) research studies meeting all of the selection criteria for the systematic review, including use of a comparator or control; (ii) research studies that did not meet all of the selection criteria (for example, no comparator was included); and (iii) the beliefs of conservation practitioners, elicited through a questionnaire survey. In order to enable different sources of evidence to be integrated, the effectiveness of each method in controlling *Rhododendron* was scored using a standard scale: −2, very effective; −1, effective; 0, no impact; 1, ineffective (i.e. slightly promotes growth); 2, very ineffective (i.e. substantially promotes growth).

By drawing on evidence from a range of sources, a number of biases become apparent. For example cutting was reported by 17 questionnaire respondents (71%) but was only considered by a single study that met the selection criteria (4%) (Table 1). Other mechanical methods such as hand pulling and winching were similarly represented more strongly in questionnaire responses than published studies. In contrast, Imazapyr is used by a small proportion of questionnaire respondents (4%) but has repeatedly been investigated by research studies (Table 1).

A BBN was constructed with a very simple structure (Fig. 5). A single node, with a state defined for each of the 13 control measures considered, was connected to a single node representing effectiveness of control, with five states according to the scoring approach adopted ('Very effective', 'Effective', 'No impact', 'Ineffective', and 'Very ineffective'). The CPTs were completed separately (in individual networks reproducing the same structure) for each of the three different sources of evidence, by calculating the proportion of studies (or questionnaire responses) for which each score was reported.

As noted in Table 1, no single source of evidence included all of the 13 methods, highlighting the value of drawing on more than one source of information. For this reason, a combined BBN was produced, incorporating evidence from all three sources. One of the features of a BBN is that different sources of evidence can be incorporated and weighted in different ways. For example, published studies which met the selection criteria might be considered more reliable, and therefore accorded greater weight, than purely anecdotal observations. The process of refining the conditional probabilities in the CPTs is referred to as 'sequential learning', which involves incrementally updating the knowledge incorporated in the network. The sequential learning algorithm implemented in Hugin is described by Spiegelhalter and Lauritzen (1990); see also Cowell and Dawid (1992) and Olesen, Lauritzen, and Jensen (1992). The procedure sequentially updates the initial values given in the CPTs by incorporating values derived from different cases. The algorithm performs a series of iterations, and maximises the logarithm of

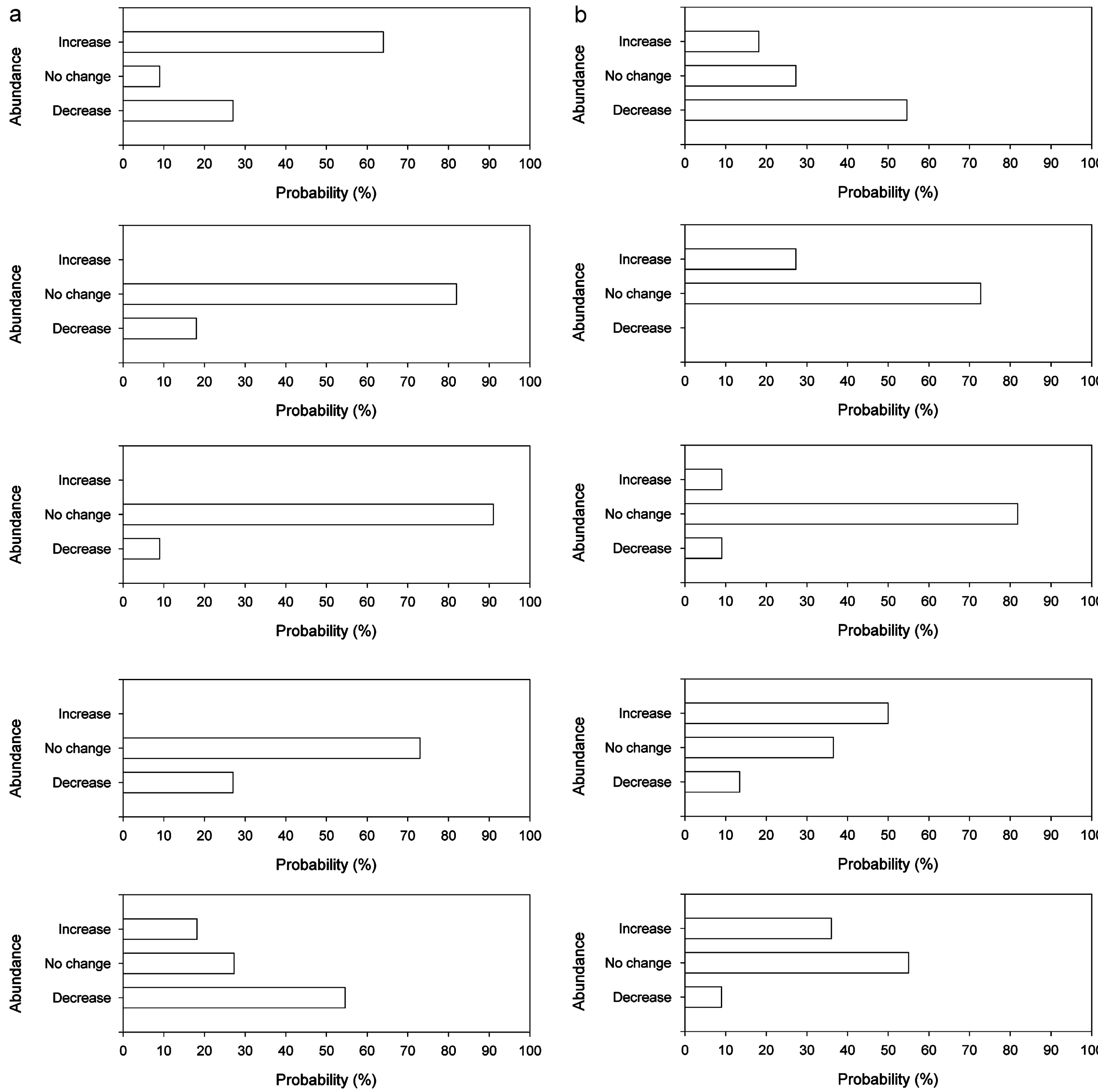

Figure 4. Results obtained with the BBN for assessing the impact of burning on upland bog vegetation (Case 2). Bars represent probabilities associated with each node state. The states represent the change in abundance of different plant species, namely (Fig. 4a, from top to bottom, *Calluna vulgaris*, *Calypogeia* sp., *Campylopus* sp., *Cephalozia* sp., *Eriophorum angustifolium*; Fig. 4b from top to bottom, *Eriophorum vaginatum*, *Plagiothecium undulatum*, *Sphagnum capillifolium*, *Trichophorum caespitosum*, *Vaccinium myrtillus*).

the probability of the case data given the current joint probability distribution (Hugin, 2003).

The potential effectiveness of different control methods can be explored by instantiating the 'Method' node with one of the states, representing one of the control methods. Four examples are illustrated, incorporating the combined evidence from the three sources (Fig. 6a–d). In each case, the result was a probability distribution associated with the different states of the 'Effectiveness' node. In the case of Glyphosate injection, the most likely outcome (60% of cases) was 'Effective' (Fig. 6a). Cutting displayed a wider range of potential outcomes, the most likely being 'No impact' (36.4%) (Fig. 6b). The sequential learning process was then repeated, weighting the three sources of evidence in the ratio 3:2:1, for studies that did meet selection criteria, studies that did not meet selection criteria and questionnaire responses respectively. This resulted in a different

Table 1. Comparison of the amount of evidence available for different methods of controlling *Rhododendron ponticum*, from three different sources

Control method	Questionnaire responses	Studies that did not meet selection criteria	Studies that did meet selection criteria
Imazapyr surface application	1	0	5
Imazapyr stem injection	1	0	2
Glyphosate surface application	15	2	5
Glyphosate stem injection	8	0	2
Triclopyr surface application	2	0	5
Metsulfuron-methyl surface application	0	0	1
Cutting	17	5	1
Imazapyr application post-cutting	0	0	4
Glyphosate application post-cutting	16	0	4
Triclopyr application post-cutting	2	2	2
Metsulfuron-methyl application post-cutting	0	0	0
Hand-pulling	16	0	1
Winching	12	3	0

Numbers indicate either the number of respondents to questionnaires or the number of published studies in which the method was featured.

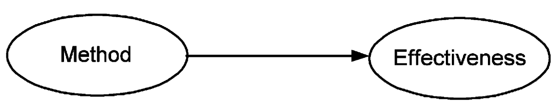

Figure 5. BBN constructed for assessing the effectiveness of different methods of controlling the invasive exotic plant *Rhododendron ponticum* (Case 3).

outcome: the most likely outcome of Glyphosate injection was 'Very effective' (57.1%) (Fig. 6c) and the most likely outcome for cutting was 'Effective' (37.9%) (Fig. 6d). Clearly, other outcomes would have been produced had different weightings been employed. This illustrates how sensitivity analyses can be used to provide explore the uncertainty associated with different forms of evidence.

Case 4. Management of lowland heathland by burning

This example illustrates an additional feature of BBNs that strengthens their value as a decision-support tool. A network can be constructed as an *influence diagram* by including *decision* and *utility* nodes. A decision node represents a decision to be made by the user. A utility node represents the utility or value of each of the states of the parent node to which it is connected, calculated by an appropriate function. When a decision is made, the probabilities of the configurations of the network are altered. The utility nodes can be used to calculate the expected utility of each decision alternative, which can be summed across the whole network, enabling the alternative with the highest expected overall utility to be identified. An influence diagram is often characterised by a sequence of decision nodes, reflecting the sequence in which decisions are made. Decision nodes are therefore each connected to the next one in the decision sequence.

Influence diagrams can accurately represent the process of decision-making by conservation managers. Decisions are usually part of a complex web of potential options in which one decision influences the next. This interconnectivity may be due to spatial or temporal linkages between habitat elements, or reflect the phasing of activities in a management plan. In the context of lowland heathland, the decision to burn a whole reserve is never taken. Rather, small patches of shrub vegetation are usually exposed to a controlled burn, which is typically performed to create or maintain small-scale heterogeneity in vegetation structure and composition.

A BBN was constructed as an influence diagram to illustrate the potential impact of burning on four habitat patches (patches 1–4) of a lowland heathland. A decision node was included for each patch, representing the decision regarding whether or not to burn. To simplify the network, for the purposes of illustration, simulated data were used to complete the CPTs. Two species are incorporated in the network: Species A, which occurs preferentially in juvenile heather; and Species B, which

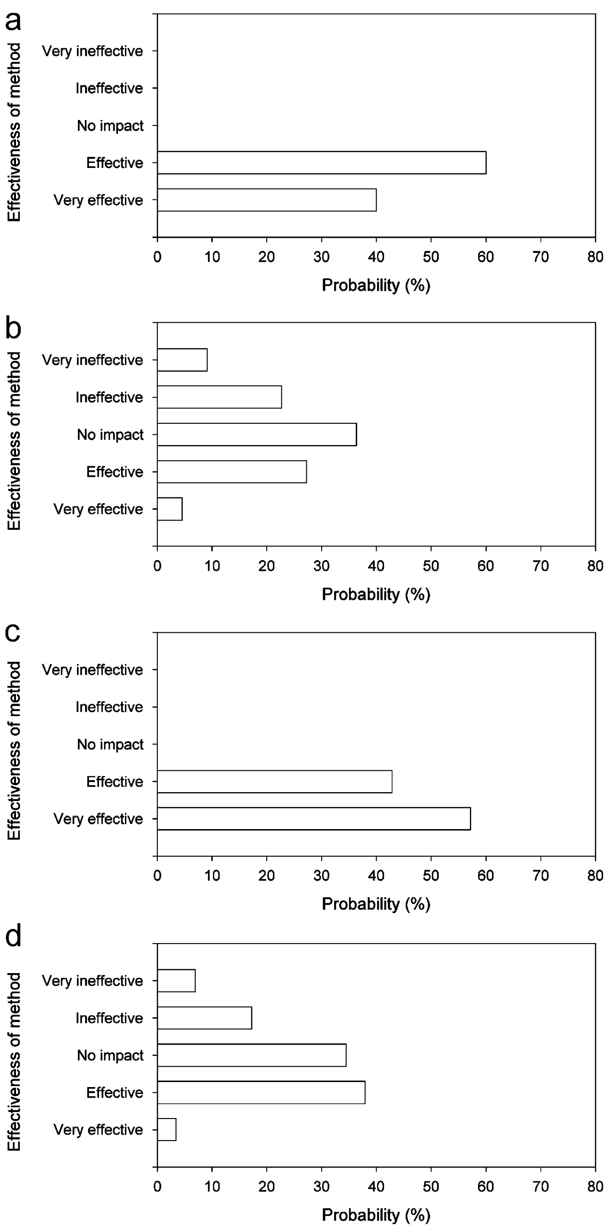

Figure 6. Results obtained with the BBN constructed for assessing the effectiveness of different methods of controlling the invasive exotic plant *Rhododendron ponticum* (Case 3). In figure, the CPTs were based on three sources of evidence combined. Figs. (c) and (d), were based on weighted sources of evidence (see text). Bars and associated numerical values represent probabilities associated with each node state. The states represent the effectiveness of different control methods. (a) Inferred effectiveness of Glyphosate injection. (b) Inferred effectiveness of cutting. (c) Inferred effectiveness of Glyphosate injection, based on weighted evidence. (d) Inferred effectiveness of cutting, based on weighted evidence.

prefers mature heather. Examples of heathland species of conservation concern with such contrasting habitat requirements are the Silver-studded blue butterfly (*Plebejus argus*) and Hen Harrier (*Circus cyaneus*), respectively, the former requiring short vegetation for breeding and the latter requiring mature heather for nesting. Change in the heather structure within any one patch can reduce the ability of that patch to support populations of each species for a specific period. The spatial pattern of habitat patches within a landscape may be important in determining the long-term persistence of these species in the area, and therefore controlled burning may be preferred as a management option. However, public opinion and financial cost can constrain burning as a management activity, which can be represented in the utility nodes.

While the BBN cannot represent the spatial element of the decision explicitly, it can be used to produce models for each individual patch that can then be linked together to take into account spatial processes. As an illustration, it is assumed here that the heathland area contains only four patches (although same approach could be extended to larger patch numbers). Decisions taken regarding the burning of one patch will influence the decision to burn any of the other three patches. The influence diagram for a single patch (Fig. 7) illustrates the decision node ('Burn'), representing the decision whether or not to burn, and the potential impacts on both heather structure (pre- and post-fire structure being represented by separate nodes), and the probability each of the two species (represented by separate nodes) being present.

The four patches are included in the influence diagram using the ability of BBNs (as implemented in Hugin software) to be represented as an *object-oriented network* (OOBBN). This is a network that contains *instance nodes*, which represent an instance of another network. An OOBBN can therefore be viewed as a hierarchical model, making it easier to include repeated elements (such as the individual patch networks in this case) and improving the visual clarity of complex networks (Hugin, 2003). In the OOBBN, two additional nodes are included that represent the risk of extinction of each species from the entire heathland area (Fig. 8). Utility nodes have then been added to represent the costs of burning each patch, which may differ between patches, and the weights attached to extinction risks for each of the two species, which might reflect relative conservation value (e.g. international versus local conservation priority).

The state of the OOBBN before the first decision has been taken is illustrated on Fig. 9a. The preferred option is not to burn patch one, based

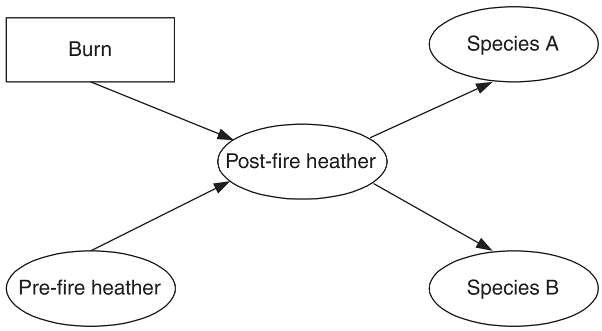

Figure 7. BBN constructed for assessing the impacts of burning lowland heathland (Case 4). This diagram represents a BBN constructed for a single habitat patch. The rectangular node ('Burn') is a 'decision node', referring to the decision whether or not to burn the heathland. Pre- and post-fire heather nodes refer to the structure of heather (*Calluna vulgaris*) stands, respectively before and after the burning event. The nodes species A and species B represent the abundance of two species with different habitat requirements, in terms of heather structure.

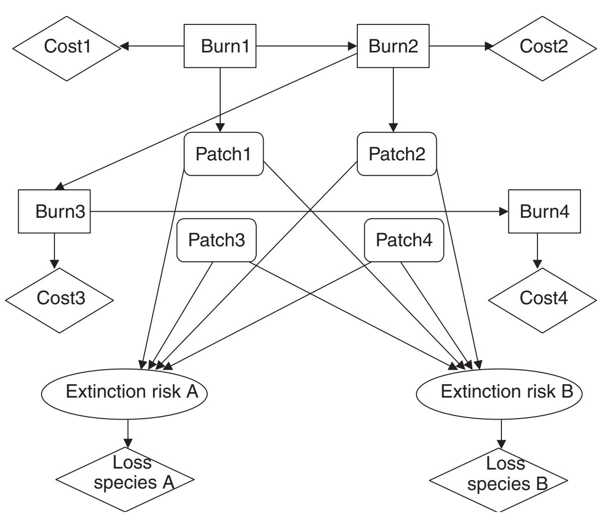

Figure 8. Structure of an object-oriented BBN (OOBBN) constructed for assessing the impacts of burning lowland heathland (Case 4). 'Patch 1–4' are instance nodes, each representing a BBN for each individual habitat patch. The rectangular nodes ('Burn 1–4') are decision nodes, each referring to the decision whether or not to burn one of the four individual patches of heathland. The diamond-shaped nodes are utility nodes, representing the utility or value of each of the states of the parent node to which each is connected. The nodes 'Extinction risk A' and 'Extinction risk B' refer to the likelihood of extinction of Species A and Species B respectively, on the heathland under consideration. This is modelled as a function of the probability that each species is present on one or more habitat patches in the heathland, considering all patches together.

on least negative value of overall utility. In contrast, patches 2 and 3 should be burnt, whereas patch 4 should not be burnt, again based on assessment of utility values. This illustrates how the inclusion of decision and utility nodes can be used to identify the preferred management option (in this case, minimising financial cost and extinction risk). The second example (Fig. 9b) illustrates the OOBBN for the case where patch 1 has in fact been burnt, achieved by instantiating the decision node for patch 1. Results indicate that patches 2 and 4 should not be burnt, whereas patch 3 should be burnt. This reflects the increased extinction risk of Species A (the likelihood of high risk of extinction having increased from 22% to 32.8% as a result of burning patch 1). Exploration of the model in this way can enable the potential impacts of different sequences of management decisions to be explored.

Discussion

BBNs possess a number of features that make them particularly valuable as a tool for supporting evidence-based conservation management. First, the graphical interface of a BBN provides a highly intuitive means of representing the features of a system of interest. The first step in producing a BBN is to illustrate the system (or domain) as a diagram, in which variables (nodes) are represented as ellipses. These nodes are then connected by arrows, which indicate conditional dependencies between the variables. The process of producing such a diagram is equivalent to producing a conceptual model, and can most readily be achieved by consulting relevant experts through an iterative process, or by consulting the scientific literature. The value of such diagrams as a tool for eliciting expert knowledge is highlighted by Burgman (2005). Conceptual modelling is also recognised as an essential feature of adaptive management approaches (Margoluis & Salafsky, 1998; Salafsky, Margoluis, & Redford, 2001, 2002; Salafsky, Margoluis, Redford, & Robinson, 2002), and therefore BBNs could potentially be of value in this context. A conceptual model could potentially be developed as part of a systematic review, subjected to peer review and finalised prior to obtaining data, to provide transparency regarding the process.

One of the main strengths of a BBN is that different types of information can be integrated according to a common framework. A range of different types of evidence may be available in relation to a particular conservation management

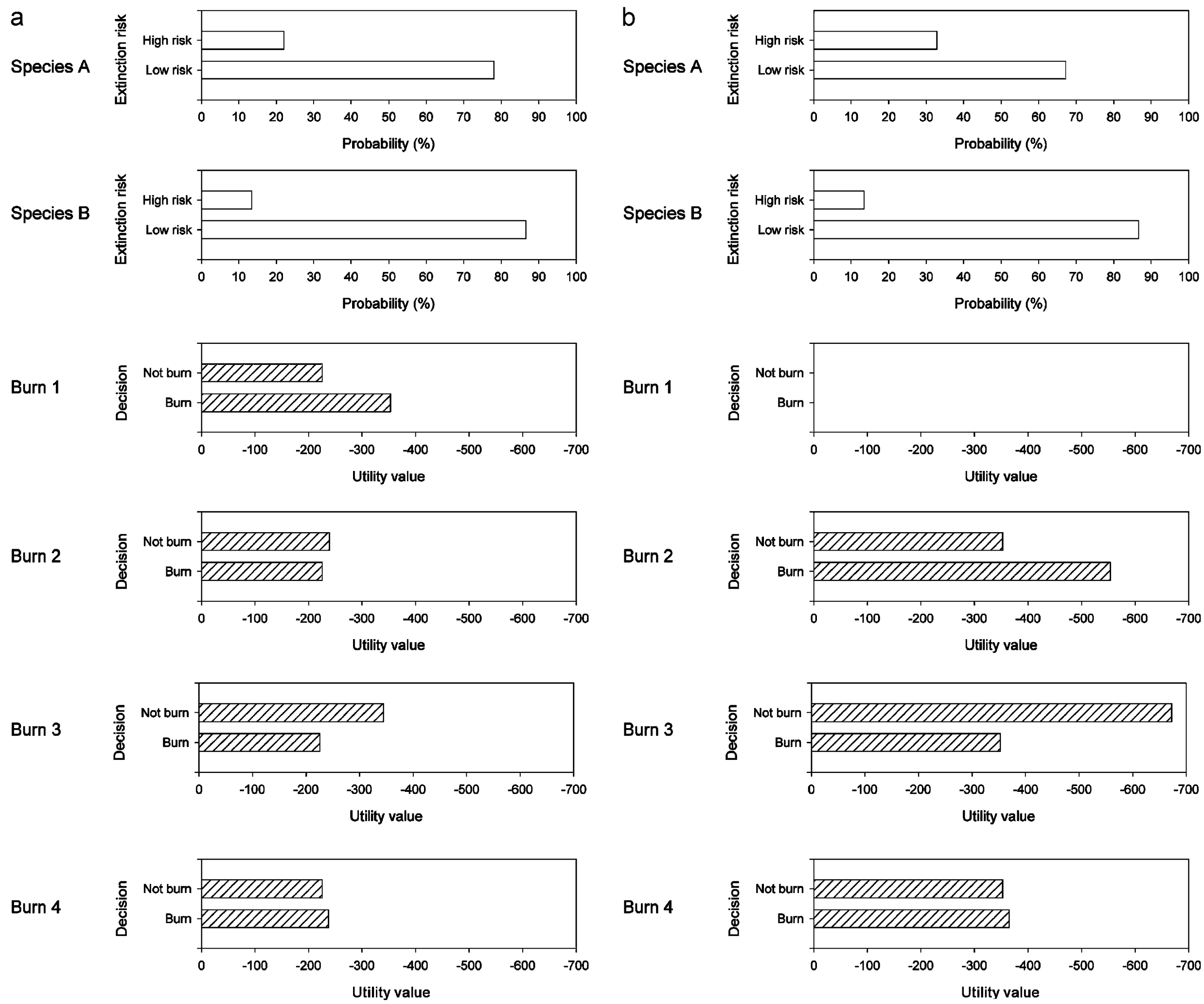

Figure 9. Results of an object-oriented BBN (OOBBN) constructed for assessing the impacts of burning lowland heathland (Case 4). In the case of 'Extinction risk A' and Extinction risk B', bars and associated numerical values represent probabilities associated with each node state, for the risk of extinction of Species A and B respectively. For the other nodes, bars represent combined utility value across the entire net. In this example, using simulated data, arbitrary values of the utility functions have been used, such that loss of Species B is weighted more heavily than loss of Species A. The more negative the utility value, the less desirable the outcome. (a) The OOBBN prior to any decision being taken. (b) The OOBBN once a decision has been taken to burn Patch 1, achieved by instantiating the appropriate decision node (Burn 1).

issue, including both quantitative results from scientific research investigations and qualitative information such as expert knowledge. As illustrated by Case 3, BBNs offer a means of integrating and comparing such different types of evidence, which can also be weighted according to their relative reliability or value. Furthermore, BBNs can be used to highlight inconsistencies and biases between different sources of evidence, and also to identify conflicting evidence; for example the Hugin software used here provides quantitative measures of the degree of conflict (Hugin, 2003). By enabling a range of different sources of evidence to be integrated, BBNs offer a method of overcoming one of the main problems facing systematic review, namely the very small number of statistically robust experimental investigations that have been performed. However, it must be remembered that results should be viewed with considerable caution when synthesising large volumes of relatively "low-quality" evidence.

BBNs can be used to support management decision-making in a variety of ways, for example by enabling the potential impacts of different management interventions to be explored (illustrated by all of the case studies presented here) and trade-offs to be identified (e.g. Cases 1, 2 and 4). A key feature is that the results are presented as probability distributions or relative likelihoods of different outcomes. This provides a highly visual means of representing the uncertainty surrounding the potential outcomes of management interventions. Uncertainty is an intrinsic feature of all ecological systems, and a consideration of this uncertainty should always inform the decision-making process, for example through a formal risk assessment or hazard analysis (Burgman, 2005). Ultimately, conservation management is primarily concerned with identifying the principal threats to species and habitats, and identifying how such threats can most effectively be countered. While the immediate causes of biodiversity loss may often be difficult to identify with precision, BBNs offer a powerful tool for integrating the available evidence relating to a particular conservation issue and for identifying which management response is likely to be most effective under a given set of circumstances. This is particularly the case when constructed as an influence diagram, as illustrated by Case 4.

The case studies illustrate a range of different types of conservation issue, and the different types of evidence and uncertainty that may be encountered. For example, Case 1 describes a replicated experiment performed on a single site. While quantitative relationships between *Spartina* volume, plant species diversity, and macro-invertebrate abundance were established experimentally, generating quantitative evidence, variation was encountered between different field plots in the results obtained. Estimation of Sika deer densities and the potential impacts on redshank populations were subject to even greater uncertainty, reflecting the difficulties of measuring these variables; consequently for the latter variables, the model was dependent on expert knowledge.

In other situations, results from more than one study may be available. Cases 2 and 3 highlight the problems of integrating results from multiple investigations, as contrasting results are frequently obtained on different sites. Studies also differ with respect to the precise methods adopted, such as sampling approaches and experimental designs, the environmental characteristics of the sites where they are performed, and the variables measured. We illustrate here that BBNs not only enable the results of different studies to be integrated (for example by using a common scoring approach), but that the uncertainties surrounding potential outcomes can be illustrated visually as part of model output. BBNs can also help communicate results to decision-makers by simplifying complex problems. For example, the changes in abundance of different plant species recorded in Case 2 could be combined into a single overall node representing favourable condition, enabling information to be provided in a way consistent with indicators used by standard approaches to monitoring (Hurford & Schneider, 2006).

BBNs also suffer from a number of limitations. The method is dependent on the use of specialist software, such as Hugin (as used here), Netica or MSBNX. Although sharing the same basic structure, these programs differ in the precise details of how Bayesian inference is performed, and may therefore produce slightly different results, although this has never been critically examined. In all BBN software, nodes are most readily presented as discrete categorical variables rather than as continuous variables, partly because the underlying theory for the latter is still being developed (Hugin, 2003). While this does not necessarily present a substantial problem, it does affect how variables can be described; careful consideration needs to be given to the definition of categories used for each variable. Unlike compartment-flow modelling approaches widely used for modelling ecological systems (e.g. see Costanza & Voinov, 2001), BBNs cannot incorporate feedbacks. Another key consideration when building a BBN relates to the dependences and independences among the variables, which have a major influence on the outcome. These can be analysed using the rules of d-separation presented by Pearl (1988), which enable the logic of the model structure to be verified.

The main challenge when developing a BBN relates to completion of the CPTs (Neil et al., 2000). BBNs are able to learn CPT values directly from a data set, and consequently they are widely used for automated data mining. However, this is rarely possible in investigations relating to conservation management, where available data sets are often limited. CPTs can be completed using expert knowledge or results of research investigations, as illustrated here, but in situations where information is lacking, conditional probability values may be based on very restricted information. This should be borne in mind when interpreting results (Marcot et al., 2001). It is important to remember that absence of evidence is not the same as evidence of absence; a zero probability may therefore simply reflect lack of appropriate evidence rather than absence of a probabilistic relationship. CPTs can also become large and unwieldy in complex networks, and for this reason, the number of parent nodes used is usually less than four. The use of proxy variables that are not measured directly is a useful technique in this context.

Another consideration when interpreting results is that BBNs are explicitly tools for modelling belief. Under the Bayesian paradigm, evidence that is consistent with a given hypothesis (e.g. the effectiveness of a particular management intervention) has a high likelihood. When Bayes' theorem is used, this results in a strengthening in the belief in the hypothesis. In this context, the 'effectiveness of a management intervention' does not strictly refer to a predicted future state, but rather represents a strong belief based on accumulated evidence that the intervention is effective. This needs to be borne in mind when using a BBN as a decision-support tool or predictive model. In common with any other modelling approach, the reliability of the method depends on the accuracy and precision of the information employed, and the results should be viewed with caution (Newton, 2007).

Despite such caveats, we believe that BBNs could make a significant contribution to current efforts supporting evidence-based conservation. A number of complementary internet-based information resources have recently been developed to disseminate evidence regarding the effectiveness of different management techniques to conservation practitioners (e.g. http://www.conservationevidence.com, and http://www.cebc.bham.ac.uk/). BBNs can potentially be web-enabled, offering the possibility of using them as an interface to such internet-based evidence bases. The sequential learning capacity of BBNs could be of particular value in this context, enabling them to be readily updated as new evidence becomes available, and the networks to evolve over time.

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