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Modelling the benefits of habitat restoration in socio-ecological systems

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Abstract

Decisions affecting the management of natural resources in agricultural landscapes are influenced by both social and ecological factors. Models that integrate these factors are likely to better predict the outcomes of natural resource management decisions compared to those that do not take these factors into account. We demonstrate how Bayesian Networks can be used to integrate ecological and social data and expert opinion to model the cost-effectiveness of revegetation activities for restoring biodiversity in agricultural landscapes. We demonstrate our approach with a case-study in grassy woodlands of south-eastern Australia. In our case-study, cost-effectiveness is defined as the improvement in native reptile and beetle species richness achieved per dollar spent on a restoration action. Socio-ecological models predict that weed control, the planting of trees and shrubs, the addition of litter and timber, and the addition of rocks are likely to be the most cost-effective actions for improving reptile and beetle species richness. The cost-effectiveness of restoration actions is lower in remnant and revegetated areas than in cleared areas because of the higher marginal benefits arising from acting in degraded habitats. This result is contingent on having favourable landowner attitudes. Under the best-case landowner demographic scenarios the greatest biodiversity benefits are seen when cleared areas are restored. We find that current restoration investment practices may not be increasing faunal species richness in agricultural landscapes in the most cost-effective way, and that new restoration actions may be necessary. Integrated socio-ecological models support transparent and cost-effective conservation investment decisions. Application of these models highlights the importance of collecting both social and ecological data when attempting to understand and manage socio-ecological systems.

Keywords: Bayesian Networks; Socio-ecological systems; Expert opinion; Restoration; Revegetation; Decision making; Uncertainty; Species richness

1 Introduction

Biodiversity loss is occurring on an international scale and habitat loss and fragmentation resulting from clearing for agriculture is a major contributor (Gibbs et al., 2009; McIntyre and Hobbs, 1999). Land management agencies are increasing their investment in biodiversity conservation efforts on private land because it covers a higher proportion of many continents and habitat loss is increasingly occurring in these areas (Soulé et al., 2004; Soulé and Sanjayan, 1998). However, this raises potential difficulties for decisions about biodiversity conservation, as decisions are often complicated by multiple and competing social, ecological and economic objectives (Allison and Hobbs, 2004; Olsson et al., 2006). Some of the most important management decisions are about how to improve biodiversity cost-effectively and how to involve private landowners in conservation efforts (Holzkamper and Seppelt, 2007; Sebastián-González et al., 2011).

Landowner decisions about conservation initiatives are influenced by their values, beliefs, and personal and social norms (Stern et al., 1995; Whittaker et al., 2006). Having an understanding of these drivers of landowner management decisions and how these decisions impact biodiversity on privately owned land can better inform natural resource management actions (Carr and Hazell, 2006; Jellinek et al., 2013b). However, few ecological studies have researched landowners' attitudes towards remnant and restored land, their adoption of restoration activities, and how this influences faunal persistence in agricultural landscapes (Morton et al., 2010; Smith, 2008). By incorporating social and ecological data into the decision making process we can better understand the impacts of landowner attitudes and management on biodiversity (Olsson et al., 2006; Ticehurst et al., 2011). A socio-ecological approach identifies management needed to achieve conservation objectives, and defines the social constraints and opportunities for implementing that management (Wyborn et al., 2012).

Given environmental management budgets are small relative to the scale of biodiversity loss, it is critical that we have tools that enable managers to improve biodiversity cost-effectively (Menz et al., 2013; Polasky et al., 2011; Sebastián-González et al., 2011), and evaluate and justify the budgets required to achieve biodiversity objectives (Rumpff et al., 2011). This includes understanding the social opportunities and constraints that are likely to enable this management to occur, or inhibit it (Smith, 2008; Wyborn et al., 2012). To reduce the uncertainty about what environmental benefits and ecosystem services can result from restoration (Duncan and Wintle, 2008; Vesk and Mac Nally, 2006), there is also a need to explicitly calculate the cost-effectiveness of competing restoration options (Measham, 2007; Rumpff et al., 2011).

Uncertainty about investment effectiveness can be partly addressed by integrating existing expert knowledge with available field data into a process model that represents the relationships between restoration actions and biodiversity outcomes (Rumpff et al., 2011). Expert opinion is increasingly being used to make ecological decisions as field data is often lacking (McBride and Burgman, 2012), although there is often uncertainty about the robustness of this expert knowledge (Marcot et al., 2006). Such models enable investigation and evaluation of competing management investment options and underpin transparent management decision-making (Duncan and Wintle, 2008; Rumpff et al., 2011). Bayesian Networks are a good basis for building process models as they help to structure reasoning and to identify causal relationships between multiple variables (Marcot et al., 2006). Bayesian Networks are an ideal tool for facilitating better management decisions as they allow the quantitative integration of field data and expert opinion (Burgman et al., 2010; Ticehurst et al., 2011); allow the explicit incorporation of uncertainty; and can be updated with new monitoring data to reflect a better understanding of the natural system over time (Glendining and Pollino, 2012; Rumpff et al., 2011).

Combining sources of data in a Bayesian Network framework to analyze the impact of management scenarios on a performance measure (or variable) of interest is not new (McCann et al., 2006; Pollino et al., 2007; Ticehurst et al., 2011). However, we aim to use Bayesian Networks to combine expert opinion with ecological and social data from two agricultural regions in south-eastern Australia to enable investigation of: (i) the restoration actions that most cost-effectively increase reptile and beetle species richness (as a measure of a biodiversity objective); and (ii) how landowner demographics and management decisions influence reptile and beetle species richness. This approach provides a useful tool for management agencies attempting to understand the social opportunities and constraints associated with making cost-effective decisions about biodiversity conservation in agricultural landscapes.

2 Materials and methods

We developed a process model to represent existing knowledge about the ecological and social cause-and-effect relationships relevant to our aim of predicting the outcomes of restoration actions on a performance measure of interest. In our case-study, we focussed on how social and ecological processes mediate the impact of restoration actions on the richness of reptile and beetle species at the patch scale (Fig. 1). The environmental, social and management variables included in the model were recorded during a field study of reptile and beetle communities and their response to habitat restoration in two agricultural landscapes in south-eastern Australia: the Wimmera and the Benalla regions (Jellinek et al., 2013a, 2013b, in preparation).

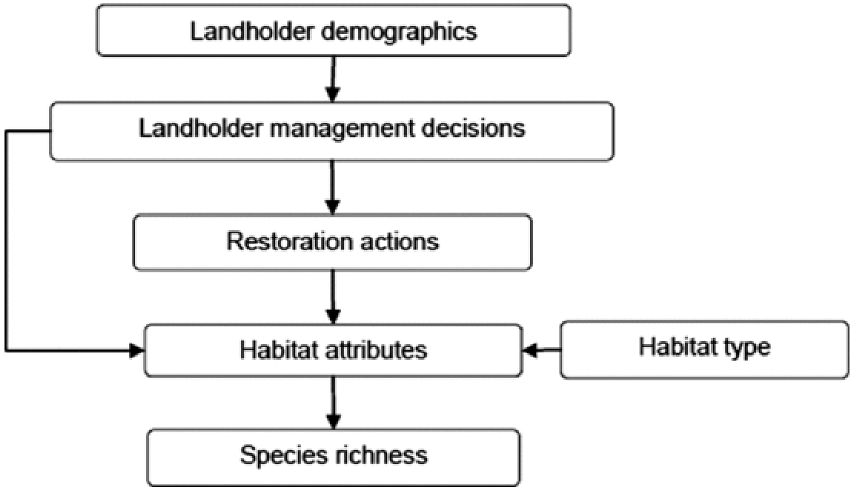


Fig. 1 A schematic representation of the Bayesian Network developed to illustrate the causal links between landowner demographics, landowner management decisions, restoration actions and habitat attributes. These variables ultimately influence reptile and beetle species richness. Arrows represent a flow of information from one group of variables to another.

2.1 Data collection

2.1.1 Study area

The Wimmera region (36.3333°S, 141.6500°E) receives an average annual rainfall of 350–500 mm with mean daily temperatures varying from 14 to 40 °C (Bureau of Meteorology, 2010). Prior to European settlement, this area supported grassy woodlands dominated by buloke (*Allocasuarina luehmannii*) and black-box (*Eucalyptus largiflorens*) on rises and flats, and grasslands on clay pans and shallow depressions (Morcom and Westbrooke, 1998). The Benalla region (36.5519°S, 145.9817°E) has an annual average rainfall of 400–670 mm and mean annual temperatures vary from 15 to 31 °C in different months (Bureau of Meteorology, 2010). Vegetation varies from box-ironbark forests containing red-ironbark (*E. tricarpa*) or yellow gum (*E. leucoxylon*) and grey-box (*E. macrocarpa*) to grey-box, white-box (*E. albens*), yellow-box (*E. melliodora*) and river red gum (*E. camaldulensis*) grassy woodlands in the more fertile soils (Radford et al., 2005). Since the 1850s these regions have been heavily cleared for intensive agriculture such as cropping and livestock production (Radford et al., 2005). These landscapes are now highly fragmented and contain less than 10% of their native vegetation cover, negatively impacting native flora and fauna (Duncan et al., 2007).

2.1.2 Landowner data

We used demographic information and data on landowners' intentions to manage remnant and revegetated areas – obtained from landowner social surveys – to describe the social drivers that influenced habitat variables and faunal species richness at a patch scale (Fig. 1) (Jellinek et al., 2013b). We defined the patch scale as the local level of habitat and species variables located within a landscape. In this study patches were usually larger than 4 ha but smaller than 600 ha and contained either remnant or restored habitat. This differs to landscape scale processes that incorporate multiple patch scale processes (Graham and Blake, 2001).

Our study of landowners in the Wimmera and Benalla region investigated their adoption of revegetation activities and their opinions on remnant vegetation and revegetated land. Private landowners were surveyed using postal questionnaires to determine (i) their previous and/or future revegetation activities and (ii) their attitudes towards remnant and revegetated areas, and how these attitudes influenced their intention to manage these areas for conservation. As far as possible we ensured that samples were unbiased by randomly selecting landowner names and addresses from publically available documents, and by minimising interviewer and researcher bias (Bryman, 2004). Questions were developed with the assistance of social scientists and the questionnaires were trialled with local landowners and natural resource managers prior to their distribution. Overall, two hundred postal questionnaires were sent to landowners in each of the two regions (Jellinek et al., 2013b).

The demographic information we recorded included the respondent's age, property size, primary source of income, enterprise type, and their membership of a Landcare group. Landcare is a community-based natural resource management group operating in 22 countries (Landcare International, 2013). A landowner's intention to undertake management actions such as weed control or the removal of ground cover were gathered using a Likert scale that offered five possible answers ('definitely' through to 'definitely not') (Bryman 2004). A Likert scale was used as it measures levels of agreement or disagreement and is the most relevant method for assessing attitudes (Seale, 2004). These answers were split at the mid-point (3) to give a binary (yes/no) response (Fielding et al., 2005). Management decisions to plant native trees, shrubs and/or grasses, and to revegetate along linear strips or in patches were gathered on a binary scale (Appendix A). A landowner's intention to manage revegetated and remnant areas was a function of their attitudes to these areas.

2.1.3 Ecological data

We undertook a two-year study to investigate the response of reptile (Class Reptilia) and beetle species (Order Coleoptera) to habitat type and environmental variables (Jellinek et al., 2013a, in preparation). In this study, we simplify the species-level data to examine the species richness of reptiles and beetles as a coarse surrogate for the ecological condition of survey locations (Fleishman et al., 2006). While it would be optimal to employ a number of habitat condition measures, we use species richness to illustrate how Bayesian Networks can be used to model multiple data sources that characterize and quantify multiple interacting social and ecological processes. Reptiles and beetles were selected as the study taxa because they are strongly influenced by vegetation type and structure (Mac Nally and Brown, 2001; Schaffers et al., 2008) but are seldom studied in the restoration literature (Munro et al., 2007).

Reptile and beetle species were surveyed using pitfall traps in several different habitat types to reflect differences in habitat floristics and structure in the two study regions (Wimmera and Benalla). In each region there were five survey locations, each location containing five sites. Each site represented five different site-types; remnant patch, revegetation adjacent to the patch, remnant linear strip, revegetated linear strip, and cleared linear strip. Linear strips were defined as areas of habitat along roadsides or fence lines that were approximately 20–40 m wide and at least 500 m long, originating from a remnant patch. Patches were at least 4 ha in size. Revegetated areas were 8–15 years old, containing native trees and shrubs that were planted; cleared linear strips contained few trees or shrubs and remnant areas contained remnant native vegetation. Reptiles and beetles were surveyed in the Wimmera region in 2008 and in the Benalla region in 2009. Traps were checked twice daily for fifteen days over three months (January–March) in each of the two years.

Habitat structure and floristic attributes that would be influenced by restoration actions (weed control, planting trees and shrubs, planting tussock grasses, adding litter and timber and adding rocks) were also measured at each site (Appendix A). Weed control, in this case the application of herbicides on non-native plants, was included as a reduction in weed species can facilitate the establishment of native plants (Nichols et al., 2010). Planting native trees, shrubs and tussock grasses can provide a variety of structural attributes required by faunal species (Munro et al., 2009). Adding structural attributes such as litter, fallen timber and rocks was included because these actions can result in increased habitat availability for reptiles and invertebrates (Croak et al., 2010; Grimbacher and Catterall, 2007).

2.2 Modelling framework

We used Bayesian Networks (Netica 4.09, 2009) as our modelling framework (Appendix B). Bayesian Networks are graphical cause and effect models, in which the relationships between variables are quantified using probabilities (Korb and Nicholson, 2004; Pearl, 1988). Each variable (or node) in a Bayesian Network has a conditional probability table (CPT) that determines how a ‘child’ node changes in relation to its ‘parent’ nodes (Korb and Nicholson, 2004). Parent nodes are defined as the nodes immediately above the node of interest in a cause-effect network. Initially, CPTs that describe how habitat attribute nodes (i.e. floristic and structural variables) influence species richness nodes were parameterized using expert elicited priors that were obtained from an ecological scientist with experience studying reptiles and beetles (D. Driscoll pers. com.). Management nodes were not parameterized using expert elicited priors. Expert elicited data was used to illustrate how reptile and beetle species would be expected to respond to changes in habitat floristics and structure when field data were absent.

The CPTs containing expert elicited priors were then updated with the field data (social and ecological data) in the form of case-files to give a ‘posterior’ distribution. To mediate the influence of the field data on the expert elicited priors we used weightings that were based on the relative sample-size of each data set. The expert elicited data was given an effective field data equivalent sample size that indicated how confident we were in the expert data (Pollino et al., 2007). In addition to providing a coherent means by which to weight expert elicited priors when combining expert opinion and field data, this process also allowed us to analyze how far expert opinion deviated from the field data and influenced the final model parameter estimates.

We used the posterior (field data and expert elicited data) model to analyze: (i) the sensitivity of reptile and beetle species richness to the other model variables, (ii) the cost effectiveness of individual restoration actions, and (iii) the impacts of landowner demographic scenarios on biodiversity outcomes.

2.3 Data analysis

2.3.1 Sensitivity to findings

Sensitivity to findings was used to identify management, demographic and environmental variables that had the greatest influence on reptile and beetle species richness (Korb and Nicholson, 2004; Pearl, 1988; Pollino et al., 2007). As our habitat attribute nodes were categorical variables, we used a sensitivity to findings analysis in Netica called ‘entropy reduction’. Entropy measures the degree of uncertainty in a variable. Entropy reduction provides a ranking of parent nodes importance described as their ability to change the posterior probability of a given state of a child node (Korb and Nicholson, 2004; Netica 4.09, 2009). Entropy reduction was calculated for all of the nodes that influenced reptile and beetle species richness, including habitat attribute, landowner demographic and landowner management decision nodes.

2.3.2 Cost-effectiveness and demographic scenarios

Economic costs and the ecological benefits of each restoration action were combined to determine the most cost-effective options for increasing reptile and beetle species richness (Appendix C). The economic costs were estimated using government documents (Rushton, 2006; Schirmer and Field, 2000) and ecological studies (Croak et al., 2010). The opportunity cost of land retired for restoration was not included because sites available for restoration were generally of low agricultural value (Vesk and Mac Nally, 2006), and because restoration may have provided future economic benefits to the landowner (Dorrough et al., 2008).

Ecological benefits were measured as the change in the expected reptile and beetle species richness arising from a unit change in restoration effort. Expected reptile and beetle species richness (predicted by the Bayesian Network) was first recorded under a no-restoration scenario, and then recorded under six (for beetles) and seven (for reptiles) restoration scenarios. Other than the five management actions undertaken for both faunal species (undertake all management actions, plant trees and shrubs, plant tussock grasses, weed control and add litter and fallen timber), beetles included a scenario to plant trees and shrubs, and add litter, while reptiles included scenarios to add rocks, litter and fallen timber. To determine the cost-effectiveness of the restoration action or combination of actions we divided the degree of change in species richness by the economic cost of the action(s) (Joseph et al., 2009; Sebastián-González et al., 2011; Stewart-Koster et al., 2010). We also compared the expert elicited data with the updated expert and field data model to calculate how expert opinion compared to field data.

Each of the restoration investment scenarios was then embedded in five demographic scenarios (Appendix D) to analyze the influence of demographic and ecological variables on reptile and beetle species richness. The scenarios we present were chosen to represent the greatest relative change in species richness resulting from demographic processes in this study. The demographic variables were region, enterprise type, Landcare membership and primary source of income. Region was important because it may influence the type of farming enterprise undertaken (Pannell et al., 2006) and enterprise type has impacts on the remnant areas within and outside the property boundary (Smith et al., 2008). Membership in a Landcare group may indicate that a landowner is more likely to carry out conservation actions (Curtis and De Lacy, 1996), while an off-farm income can increase a landowner’s ability and propensity to undertake conservation activities (Pannell et al., 2006).

3 Results

Of the 400 questionnaires we mailed out to landowners in the Wimmera and Benalla regions, 180 (45%) were completed and returned. In our ecological study we recorded 22 reptile species from 6 families and 97 beetle species from

10 families.

3.1 Sensitivity to findings

Overall, we found reptile and beetle species richness was highly sensitive to habitat attribute nodes, where species richness in reptiles varied by up to 5.8% and in beetles by up to 9.2%. The percentage value represents the entropy reduction value of a given habitat attribute (e.g. 0.076 for weed cover) as a proportion of the total entropy for the variable of interest (e.g. 1.31 for reptile species richness) (Table 1). This indicated that habitat variables such as weed cover and mid-stratum density would have a substantial influence on reptile and beetle species richness (Table 1). Reptile and beetle species richness was much less sensitive to landowner demographic and management variables when compared to the influence of habitat variables. This result may be due to the landowner demographic and management variables being several nodes away from the species richness measure (Korb and Nicholson, 2004), or because other more important variables were not modelled. Landowner demographics and management variables to remove litter and timber and replant trees and shrubs had a stronger positive influence on species richness than other management decisions (Table 1).

Table 1 Sensitivity analysis using combined expert and field data models showing reptile and beetle species richness sensitivity to habitat attributes, landowner demographics and landowner management decisions.

Variables	Reptile species richness	Beetle species richness
Habitat attributes	*Entropy (Q) = 1.31*	*Entropy (Q) = 15.65*
Weed cover	0.076 (5.8%)	1.434 (9.2%)
Mid-stratum density	0.053 (4%)	1.409 (9%)
Litter cover	0.048 (3.6%)	0.692 (4.5%)
Tussock cover	0.016 (1.2%)	0.158 (1%)
Herb cover	n/a	0.115 (0.7%)
Rock cover	0.018 (1.3%)	n/a
Landowner demographics and management decisions	*Entropy (Q) = 1.31*	*Entropy (Q) = 15.46*
Region	0.0002 (0.02%)	0.0019 (0.01%)
Landcare membership	0.0000 (0.01%)	0.0016 (0.01%)
Income	0.0001 (0.01%)	0.0006 (0.00%)
Enterprise type	0.0003 (0.03%)	0.0014 (0.01%)
Remove litter and timber	0.0053 (0.41%)	0.0375 (0.24%)
Revegetate trees and shrubs	0.0033 (0.17%)	0.0329 (0.21%)
Revegetate tussocks	0.0007 (0.05%)	0.0002 (0.00%)
Weed management	0.0006 (0.05%)	0.0053 (0.04%)
Revegetation type	0.0003 (0.02%)	0.0021 (0.01%)

3.2 Cost-effectiveness

We found that compared to no restoration actions, undertaking all restoration actions led to a trend of increased reptile and beetle species richness in all habitat types, although a substantial increase in species richness occurred only in cleared linear strips (Fig. 2a and b). This result holds true irrespective of which model (expert-only or expert + field data) was used to make predictions. Similarly, both models predict that the most cost-effective actions for increasing reptile species richness were weed control in both revegetated and remnant patches, and planting trees and shrubs in remnant and cleared linear strips (Table 2). In remnant patches the expert model predicted that adding rocks would be as cost-effective as weed control. In revegetated linear strips the expert model predicted that weed control would be most cost-effective for increasing reptile species richness, whereas the combined expert and field data model predicted that planting trees and shrubs would be the most cost-effective action (Table 2).

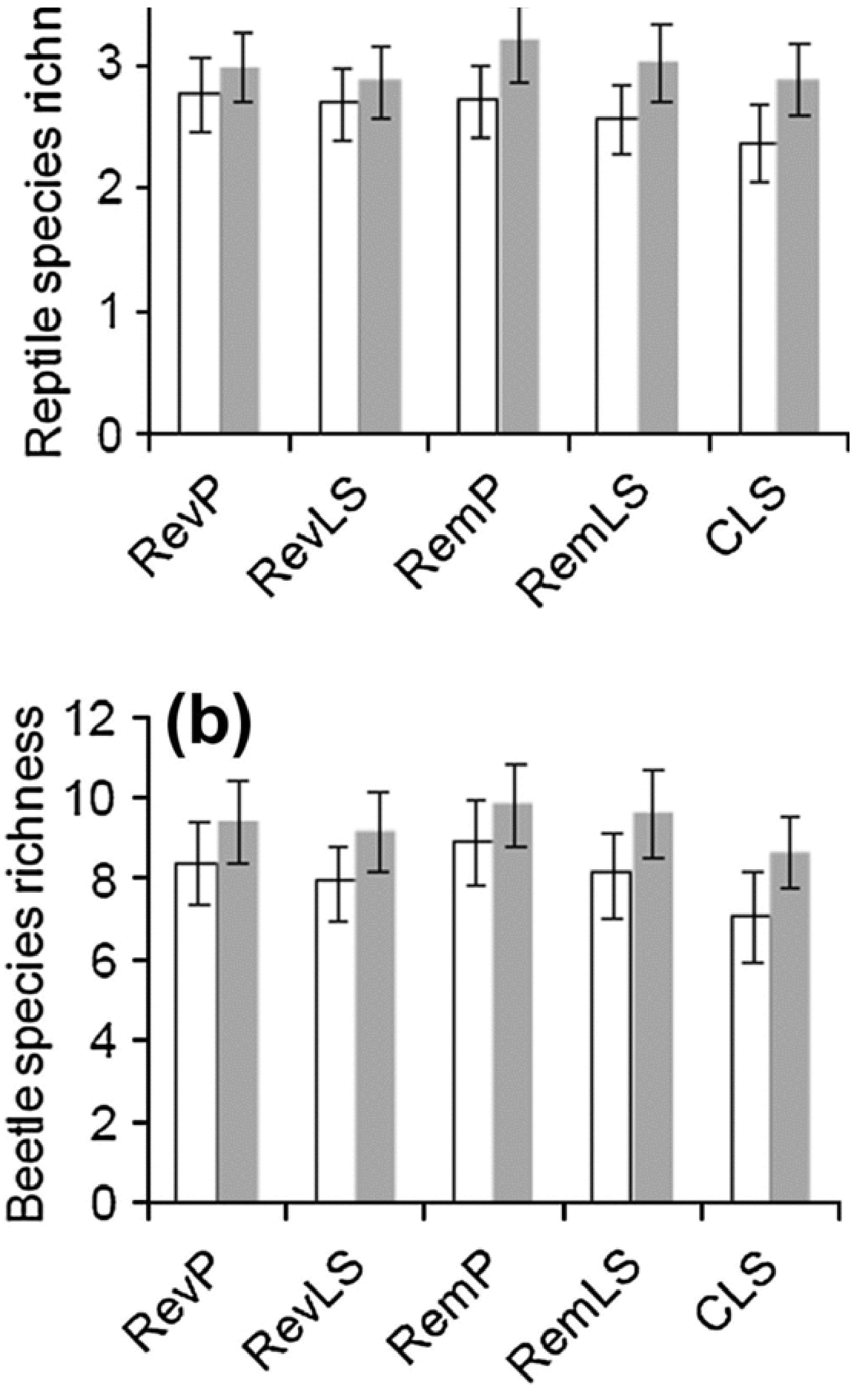


Fig. 2 The effectiveness of all restoration actions compared to no restoration actions on (a) reptile and (b) beetle species richness. Predictions were calculated from the combined expert and field data model. CLS = cleared linear strip, RemLS = Remnant linear strip, RemP = Remnant patch,

RevLS = Revegetated linear strip. Clear columns = no restoration, grey columns = all restoration actions. Bars represent 95% credible intervals, which are analogous to frequentist confidence intervals.

Table 2 Reptile and beetle species richness increase as a result of different restoration actions (compared to a no-restoration scenario) and the cost-effectiveness of those actions per $1000 spent. Top restoration actions presented out of seven restoration scenarios. Sp. gain = increase in species richness, Cost = cost-effectiveness of species gain per dollar spent. RevP = revegetated patch, RevLS = revegetated linear strip, RemP = remnant patch, RemLS = remnant linear strip, CLS = cleared linear strip. Numbers in bold represent the greatest degree of species gain or cost-effectiveness for each restoration action.

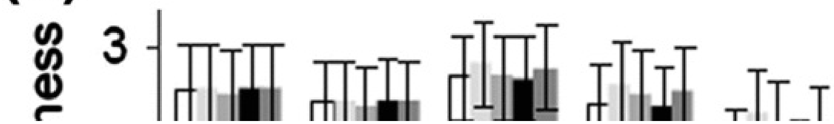
Habitat type	Restoration actions	Reptile species richness				Beetle species richness			
		Expert model		Combined expert and field model		Expert model		Combined expert and field model	
		Sp. gain	Cost	Sp. gain	Cost	Sp. gain	Cost	Sp. gain	Cost
RevP	All restoration	**0.47**	0.06	0.22	0.03	**0.57**	0.07	**1.06**	0.13
RevP	Weed control	0.08	**0.42**	0.06	**0.32**	0.24	**1.26**	0.03	0.16
RevP	Add rocks and litter	0.41	0.09	**0.39**	0.09	–	–	–	–
RevP	Add litter	–	–	–	–	0.47	0.16	0.51	**0.17**
RevLS	All restoration	**0.42**	0.05	**0.52**	0.06	**0.61**	0.07	**1.30**	0.15
RevLS	Weed control	0.08	**0.42**	–0.12	–0.63	0.25	**1.32**	–0.06	–0.32
RevLS	Plant trees and shrubs	–0.06	–0.07	0.29	**0.36**	–	–	–	–
RevLS	Add litter	–	–	–	–	0.47	0.16	0.68	**0.23**
RemP	All restoration	**0.61**	0.07	**0.47**	0.06	**1.34**	0.16	**0.92**	0.11
RemP	Weed control	0.03	**0.16**	0.02	**0.11**	0.10	**0.53**	0.10	**0.53**
RemP	Add rocks	0.24	**0.16**	0.16	0.05	–	–	–	–
RemLS	All restoration	**0.61**	0.07	**0.45**	0.05	**2.15**	0.25	**1.51**	0.18
RemLS	Plant trees and shrubs	0.16	**0.20**	0.13	**0.16**	1.30	**1.59**	0.80	**0.98**
CLS	All restoration	**0.86**	0.10	**0.52**	0.06	**3.93**	0.46	**1.61**	0.19
CLS	Plant trees and shrubs	0.30	**0.37**	0.29	**0.36**	2.87	**3.52**	0.28	0.34
CLS	Add Litter	–	–	–	–	0.20	0.07	1.30	**0.44**

Our models predicted that to increase beetle species richness in revegetated patches, revegetated linear strips and cleared linear strips, the addition of litter and timber would be the most cost-effective restoration action. In remnant patches it would be weed control, and in remnant linear strips planting trees and shrubs would be most cost-effective (Table 2).

3.3 Landowner demographic scenarios

We found that reptile species richness did not substantially change as a result of the different landowner demographic scenarios we compared (Fig. 3a). However, beetle species richness did increase in cleared linear strips (increase of 0.44 to 1.77 species per site, Fig. 3b) under scenario two (landowners from the Benalla region who were members of a Landcare group, and had an off-farm income and a livestock enterprise) compared to scenario four (landowners from the Wimmera region who were not Landcare members, and had an off-farm income and an 'other' enterprise type). In revegetated and remnant treatments there was no clear evidence of a difference in reptile and beetle species richness in response to any of the landowner demographic scenarios (Fig. 3a and b).

(a)



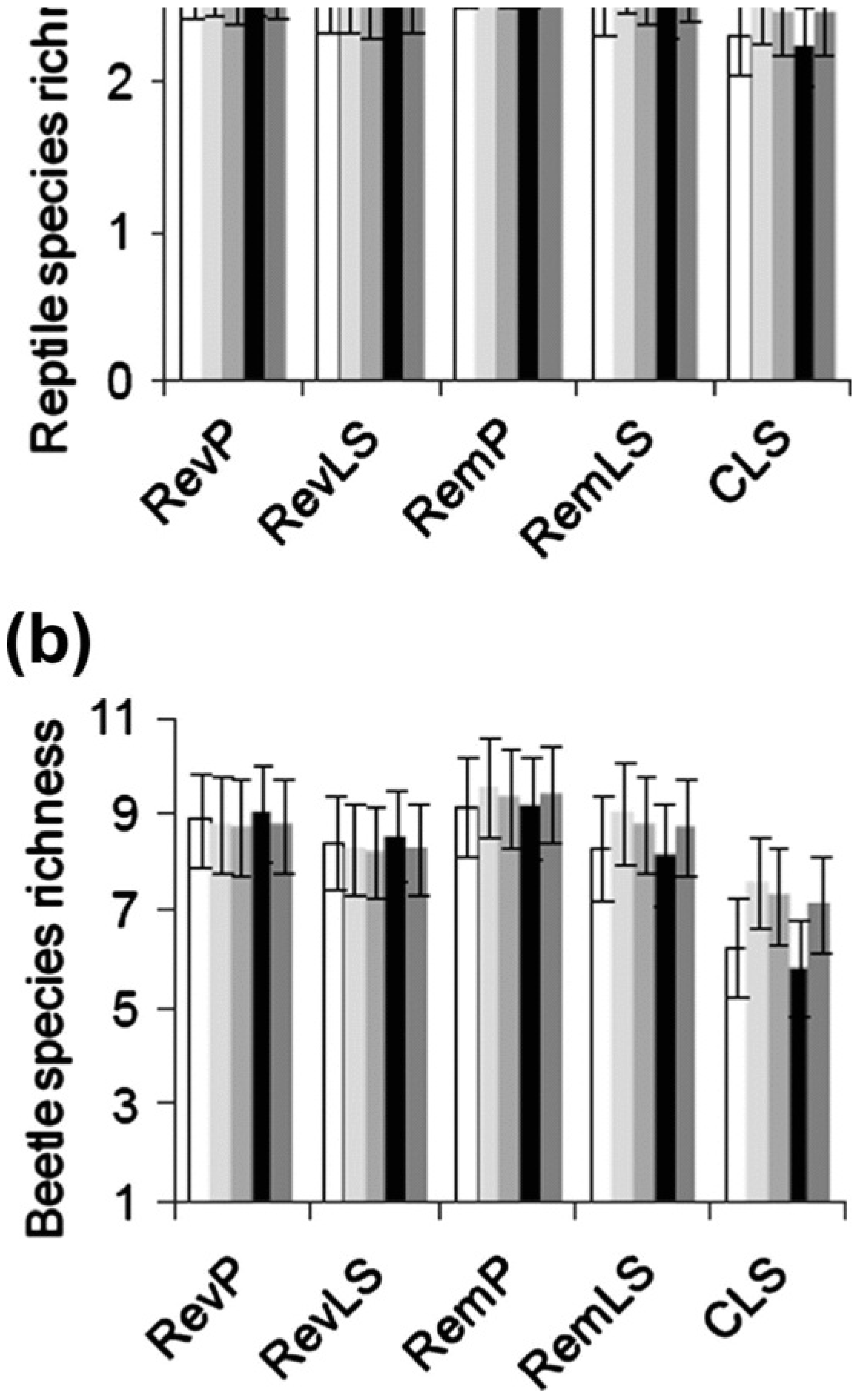


Fig. 3 Influence of changes in landowner demographics under different scenarios on (a) reptile and (b) beetle species richness. White columns = scenario 1; light grey columns = scenario 2; mid-grey columns = scenario 3; black columns = scenario 4; and dark-grey columns = scenario 5. Bars

represent 95% credible intervals.

4 Discussion

4.1 Integrating social and ecological data in a cost-effectiveness analysis

As biodiversity continues to decline in human-dominated landscapes, the need to integrate social and ecological knowledge and data into biodiversity conservation becomes increasingly important (Liu, 2001; Pannell et al., 2006). While many researchers have discussed the importance of integrating these factors in conservation planning and investment prioritization, few studies have been successful in doing so in a transparent and quantitative way (Chan et al., 2007; Olsson et al., 2006). The work presented here synthesizes landowner attitudinal data, expert opinion, and ecological field data into a single model that can be used to predict how human demography and ecology interact to influence biodiversity. These predictions can then be incorporated in feasibility and cost-effectiveness analyses to help support conservation investment decisions that explicitly address uncertainty.

In our case-study, the socio-ecological modelling revealed that when all restoration actions are undertaken, and under the best-case landowner management scenarios, the greatest species richness increases occur when cleared linear strips are restored. In contrast, species richness gains are predicted to be minimal in revegetated and remnant habitats, irrespective of the settings of the other ecological and human demography variables. In cleared areas, restoration action can lead to increases in beetle species richness of up to 23% and in reptile species richness of up to 12%, compared with relatively minor gains (3–5%) in other habitats for roughly the same level of investment. This example of diminishing marginal returns is interesting as it may indicate that as the area of native vegetation increases, species richness increases, but at a decreasing rate. Cost-effective gains in species richness may be obtainable by undertaking weed control, planting trees and shrubs and through the addition of ground layers such as fallen timber, leaf litter and rocks. This sort of information is potentially extremely valuable when conservation investors and managers are deciding where to target investment to achieve the greatest gains in biodiversity.

Species richness is unlikely to substantially increase in most habitats under current management scenarios because the agricultural landscapes studied here are highly degraded and remaining remnant habitats are isolated from larger continuous habitat (Jellinek et al., 2013a). However, other management actions not included in our model may result in greater species richness increases. These include removing livestock grazing (Lindsay and Cunningham, 2009), pest animal control (Arthur et al., 2010), creating connections and broad-scale revegetation (Munro et al., 2007). The cost-effectiveness of these actions should be quantified as a priority to measure the relative biodiversity benefits of these actions.

While our particular study places an emphasis on the role of restoration in highly degraded sites, we must add a number of caveats. Firstly, our data spanned a relatively short period following restoration action (8–15 years), which can impact on the range of activities that appear to be cost-effective. Restoration effectiveness is likely to differ over seasonal, temporal and spatial scales (Golet et al., 2009), and our snap-shot might not represent faunal changes in other time periods, highlighting the need for data gathered over extended time frames. Secondly, we have used a relatively simple measure of investment performance (species richness) that does not explicitly account for differential responses of more or less common species to the range of conservation actions considered (Fleishman et al., 2006). Nonetheless, we argue that the analytical approach we describe here can inform management decisions and cost-effectively maintain biodiversity.

Cost-effectiveness analysis provides an economically realistic approach to prioritizing investment and management of degraded agricultural areas given limited budgets (Holzkamper and Seppelt, 2007; Possingham 2001; Segan et al., 2011). Cost-effectiveness also provides a way of comparing competing management strategies to achieve the best ecological result (Holzkamper and Seppelt, 2007). The alternative to considering cost-effectiveness is a less systematic approach to restoration investment prioritization that may be ineffective in maintaining biodiversity and inappropriately allocate conservation funding (Lunney et al., 1997).

4.2 Expert opinion and field data

We found that there was a high degree of similarity in the predictions made by the expert elicited data and the combined expert and field data. This result was unlikely to be an artefact of the expert priors as this data was given a low level of confidence. If more data becomes available from field studies then the posterior is likely to be influenced to a greater extent by the field data rather than the expert opinion. The high degree of similarity in the predictions made is not always the case as expert opinion can differ substantially from field data, highlighting the need for rigour in the collection of both field data and expert opinion (Martin et al., 2005). Gaining robust expert opinion is important because this information can inform management decisions when ecological data are lacking (Kuhnert et al., 2010; Marcot et al., 2006; McBride and Burgman, 2012).

There are limitations and risks in using expert opinion due to cognitive biases and overconfidence (Burgman et al., 2011; Czembor et al., 2011; McBride and Burgman, 2012). Similarly, gaining opinions from a limited number of experts, as our study did, can reduce the accuracy of the data gathered. Approaches proposed to increase the accuracy of expert-elicited data include training experts in how to provide better judgment and training researchers in how to measure expert opinion objectively and limit bias (Burgman et al., 2011; Speirs-Bridge et al., 2010). A systematic literature review on the subject of interest, a meta-analysis of a number of studies (Cumming 2008), using Bayesian Networks along with conventional statistics (Ticehurst et al., 2011) and using multiple experts may also increase information accuracy.

4.3 Future research

The Bayesian Network we developed highlights the benefits of this method in supporting management decisions. For example, cost-effectiveness analysis can contribute to the process of optimizing revegetation investments, benefiting natural resource managers by allowing them to show, in a transparent way, how to conserve the most biodiversity for their limited management budgets (Duncan and Wintle, 2008). Given the high degree of uncertainty in the models developed here, refinement of CPTs could be achieved through ongoing data collection and integration with our Bayesian Network. Information on other potential management options could be used to expand the structure of the Bayesian Network we used, allowing managers to adapt revegetation actions to reflect the best scientific knowledge in the spirit of adaptive management (Duncan and Wintle, 2008; Walters, 1986). A major contribution of this work is to provide a coherent framework for doing just that. While there are drawbacks to this method, such as subjectivity in structuring and parameterizing the Bayesian Network, these can be addressed systematically using rigorous expert elicitation processes.

Our work takes a small step towards achieving a multi-objective approach to decision analysis in agricultural landscapes by illustrating how landowners, driven by multiple objectives, influence biodiversity through their management choices. However, we do not explicitly model and trade-off the competing objectives of economic prosperity and biodiversity conservation, or the often synergistic objectives of increasing biodiversity and aesthetic value of land. This is an important area of future research that can be partly informed by what has been shown here.

5 Conclusion

Our study illustrates how expert opinion, ecological and social data can be combined to inform management decisions to maximize biodiversity gains given limited budgets. This integrated information can be used to investigate realistic scenarios describing how landowner attitudes influence patch scale processes and biodiversity outcomes. While in our study we focus on particular ecological and landowner demographic parameters, any of these could be substituted with the variables most appropriate to a given decision context. The resulting Bayesian Networks, with the input of robust data, can help evaluate management options and effectively allocate funding to maximize biodiversity gains. However, there are limitations to the implementation of this method and the high degree of variability inherent in ecological and social data is a universal problem that can be only partially overcome with more sampling. Future improvements to the models would aim to address these shortcomings. The alternative to the systematic investment prioritization approach demonstrated here is the continued application of less systematic decision making that hampers the ongoing refinement of restoration knowledge and progress toward effective conservation decision making.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.biocon.2013.10.023.

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Appendix A. Supplementary material

Supplementary data 1

Multimedia Component 1

Highlights

- Bayesian Networks can predict how social and ecological factors influence biodiversity.
- Faunal gains are greatest when cleared areas are restored compared with remnant and revegetated habitat.
- Cost-effective management actions differ between habitats and faunal groups.
- Landholder demographics can influence habitat variables and faunal species richness.
- Current restoration actions may not increase faunal species in agricultural areas.