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REVIEW

Integrating socio-economics and ecology: a taxonomy of quantitative methods and a review of their use in agro-ecology

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Summary

1. Answering many of the critical questions in conservation, development and environmental management requires integrating the social and natural sciences. However, understanding the array of available quantitative methods and their associated terminology presents a major barrier to successful collaboration.

2. We provide an overview of quantitative socio-economic methods that distils their complexity into a simple taxonomy. We outline how each has been used in conjunction with ecological models to address questions relating to the management of socio-ecological systems.

3. We review the application of social and ecological quantitative concepts to agro-ecology and classify the approaches used to integrate the two disciplines. Our review included all published integrated models from 2003 to 2008 in 27 journals that publish agricultural modelling research. Although our focus is on agro-ecology, many of the results are broadly applicable to other fields involving an interaction between human activities and ecology.

4. We found 36 papers that integrated social and ecological concepts in a quantitative model. Four different approaches to integration were used, depending on the scale at which human welfare was quantified. Most models viewed humans as pure profit maximizers, both when calculating welfare and predicting behaviour.

5. *Synthesis and applications*. We reached two main conclusions based on our taxonomy and review. The first is that quantitative methods that extend predictions of behaviour and measurements of welfare beyond a simple market value basis are underutilized by integrated models. The second is that the accuracy of prediction for integrated models remains largely unquantified. Addressing both problems requires researchers to reach a common understanding of modelling goals and data requirements during the early stages of a project.

Key-words: agriculture, ecological model, interdisciplinary, socio-economic model

Introduction

The importance of interdisciplinary collaboration is recognized by both social scientists (Mascia *et al*. 2003) and ecologists (Lawton 2007), particularly to inform policy in the management of socio-ecological systems (Sutherland *et al*. 2008). Collaboration through the use of integrated methodologies can help policy-makers anticipate complex interactions between systems that affect ecosystem health (Watkinson *et al*. 2000), factor economic costs into conservation planning

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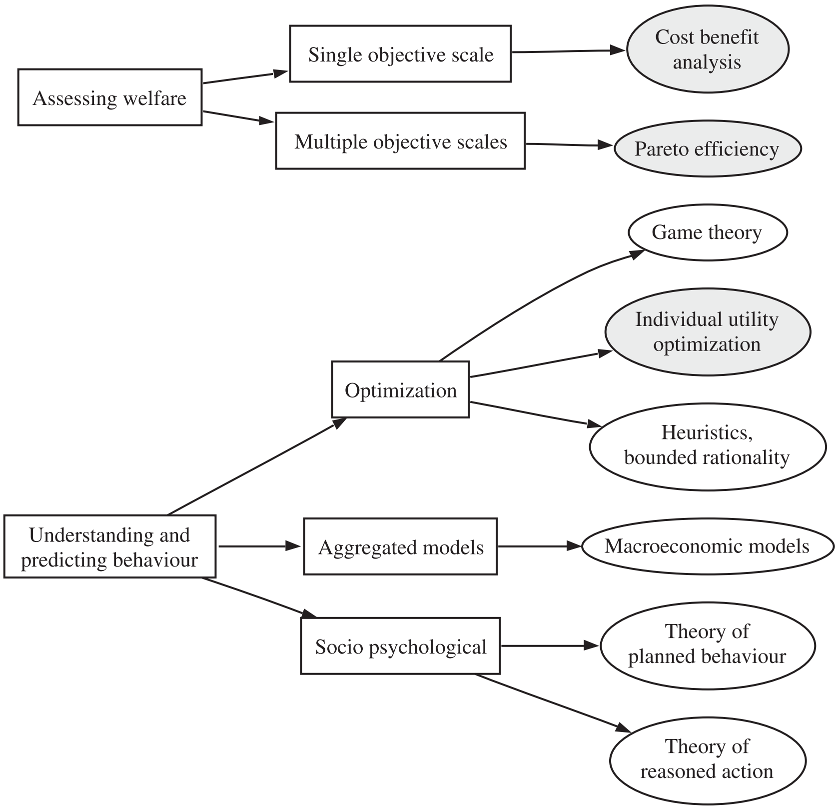

Fig. 1. A taxonomy of socio-economic methodologies and modelling approaches. Specific techniques are indicated by ellipses and boxes indicate broader groupings. Shaded techniques rely heavily on the concept of utility.

(Naidoo *et al*. 2006), and estimate the effects of conservation policy on human wellbeing (Millennium Ecosystem Assessment 2005; Boone *et al*. 2006).

Models are used for many purposes in ecology, from the development of general theoretical concepts to simulation models of populations and communities that can be used to conduct computer experiments or make predictions for conservation (Peck 2004; Sutherland 2006). The outcome of interactions between ecological and social systems can be complex, and quantitative models will be important in understanding such systems, as well as making predictions about situations that lie outside the range of previous experience (Sutherland 2006). However, there are many barriers, both perceived and actual, to building quantitative integrated models (Wätzold *et al*. 2006) and to interdisciplinary research in general (Fox *et al*. 2006). Skills in both disciplines are required (Wätzold *et al*. 2006). There are also philosophical differences: ecologists tend to use models as tools (Sutherland 2006) while economists more often use them as vehicles for communication (Drechsler *et al*. 2007a). Importantly, each discipline's array of methods and associated terminology can make the conversation of collaborators and their literature bewildering (Ewers & Rodrigues 2006). Ecologists seeking to engage with the social sciences are confronted with an array of methodologies spread through textbooks and the primary literature. Only by understanding where each technique sits in comparison to others can a researcher fully appreciate the available options. Our objective in this sudy is to provide ecologists with an overview of quantitative social science methods that distils this complexity into a taxonomy of techniques. The branches on our taxonomic tree are chosen based on the manner that methods are (or could be) applied in an ecological context and we give examples where possible. Secondly, we review how linked socio-economic–ecological models are used in practice to inform the management of socio-ecological systems. We focussed on the agricultural and silvicultural literature although our results apply to other areas where human and ecological systems overlap.

A taxonomy of socio-economic models

In this section, we provide a taxonomy of socio-economic methods and concepts. The most basic division in our taxonomy separates the two main uses of social science models in ecology and conservation: predicting human behaviour and assessing human welfare (Fig. 1). Many of the techniques in these two branches make use of the concept of utility and so we also classify methods for quantifying and describing utility (Fig. 2). We begin this section by describing what utility is and how it can be measured. Then we summarize the available modelling techniques for predicting behaviour and assessing human welfare.

MEASURING UTILITY

Utility is a measure of reward, happiness or relative well-being. Economists have traditionally sought to measure utility associated with the consumption of goods and services and

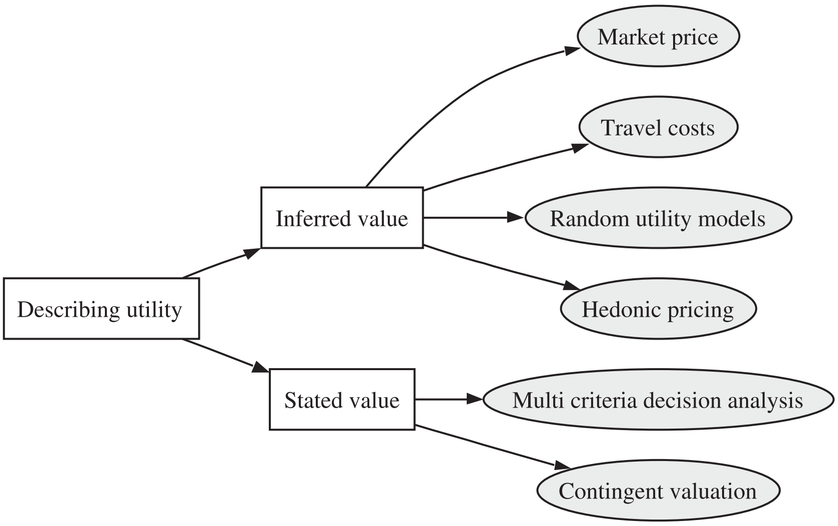

Fig. 2. A taxonomy of methods for quantifying and describing utility. Ellipses indicate distinct techniques and boxes indicate broader groupings. Many of these methods are applied to techniques in Fig. 1.

this is the most common form of utility used in quantitative models. Despite the dominance of this viewpoint, it is now well established that utility is often highly context-dependent, and only rarely intrinsic to the consumption of a good or service. For example, people place a higher value on things they have access to than those they do not (*endowment effect*, Lowenstein & Adler 1995) and the process by which something is obtained (e.g. whether given or bought) can affect its value to the individual. Formal models of utility have been developed to describe such effects (Brocas & Carrillo 2004), but they are not yet regularly used in environmental economics.

Most decisions involve making choices about whether to have or to do more or less of something. It is therefore important to estimate marginal rather than total utility: marginal utility is the utility gain (or loss) associated with one more (or fewer) unit of consumption. Marginal utility typically decreases as total consumption increases. This phenomenon underlies the economist's downward sloping demand curve that shows how willingness to pay for extra units of consumption of an item declines as more is consumed in total (assuming other components, such as the decision maker's income, remain constant). This idea can be applied to ecology. Since consumption of a good is constrained by its availability (supply), as species become rarer their marginal utility increases. This trend might provide a strong incentive for exploitation (Courchamp *et al.* 2006) or could otherwise justify a high willingness to pay for species protection. Marginal utility can be estimated by asking what price people would be willing to either (i) pay to secure an extra unit of an environmental good, or (ii) accept in compensation for the loss of one unit to which they are already entitled.

In considering utility, a key economic concept of relevance to ecology is *discounting*. This attempts to express benefits and costs that occur in different time periods in common 'present values'. Consumption of goods and services in the present is more valuable than consumption some time in the future, reflecting a preference for the present. Similarly, the future value of resources that provide flows of goods and services in the future must be discounted to give present values. In practice, the choice of discount rate reflects an ethical or moral viewpoint as witnessed in the recent debate over climate change mitigation (Stern 2007), and can vary from a low value based on giving equal weight to future generations (e.g. 0·1%, Beckerman & Hepburn 2007) to a high value that reflects market norms (e.g. 3%).

Marginal utility can be divided into components corresponding to different goods or services, with each good or service providing different types of value to a person (a single good or service may also provide multiple types of value). Values are usually divided into two classes. Firstly, there are those associated with actual use (use values), such as enjoyment from visiting a nature reserve or money obtained from harvesting fish. Secondly, there are those that are not dependent upon measurable use (existence values, option of use in the future). Use values can be measured by observing behaviour related to the use, but non-use values can usually only be measured using statements made in interviews (except where the non-use value is expressed through a proxy behaviour such as payments for conservation).

Inferred value

One way to infer value is to use the market price. In this case, the relevant behaviour being used to infer value is the buying and selling of goods and services in a market. Many environmental goods and services are not directly traded but are required for the production of other goods that are. For example, a unit decline in the abundance of pollinating insects (environmental service) would have a calculable effect on crop yields (Morandin & Winston 2006), which can in turn be valued according to market prices.

In cases where a direct relationship between use value and market valued quantities is not available, methods are available for inferring values based on estimated demand. When valuing a location, a straightforward measure is to determine the '*travel costs*' (Parsons 2003), how much individuals have spent in total to visit. The complete cost of travel including time (via wages), entry and transport should be included. Each individual is assumed to repeat visits to the site until their marginal utility per visit is equal to the travel cost. The relationship between costs per visit across different individuals and number of visits gives the demand curve, which can in turn be used to calculate the overall value of a site.

In contrast, random utility models are based around a discrete choice between options. Each of the options in a random utility model is associated with a range of attributes that contribute to utility and the likelihood that a site is chosen depends on the total of these contributions. For example, Adamowicz, Louviere & Williams (1994) constructed a random utility model for recreational use of lakes based on travel cost, fish catch rate, swimming access, water quality and boating access. Each of these attributes (except water quality) was a significant predictor. The fitted coefficients for each attribute provided an estimate of the marginal utility of that component.

Random utility models relate the attributes of a good to the likelihood that it is chosen. In contrast *hedonic pricing* relates attributes to market prices. For example, houses are more expensive if they are large, in a quiet street or close to the sea, and regression analyses can be used to quantify the value of each attribute (Taylor 2003).

Stated values

There are circumstances where it is impossible to infer values through observed behaviour. Data may be inadequate, the attributes of a good or service that determine its utility may co-vary, or the values themselves may be non-use values. In such cases, alternative approaches based on interviews or questionnaires are necessary. The most frequently used technique is *contingent valuation* in which respondents are asked how much they would be willing to pay for a good or service *contingent* on some hypothetical scenario (such as restoring or removing a habitat, species or service). The technique has been criticized because there are numerous potential biases such as participants giving a value for broad goods or services rather than the specific one being valued (Venkatachalam 2004) and because respondents may neither properly understand what is being asked nor give consistent responses (Stevens *et al*. 1991). Careful survey design (see Bateman *et al*. 2002; White *et al*. 2005) can be used to mitigate these problems.

Multiple-criteria Decision Analysis comprises a suite of approaches that were developed to assist decision-making with multiple objectives but which can also be used to quantify utility functions that have multiple objectives (Belton & Stewart 2002). One approach is to provide repeated pairs of combinations of options and ask which is preferred. The preferences then reveal the relative importance of each option and whether each is considered positively or negatively. *Multi-attribute Utility Theory* acknowledges that there may be many conflicting components to utility. For example, a farmer might gain utility from the presence of particular bird species, high crop output and peer recognition. These attributes could conflict when considering a conservation intervention that benefited some species but was detrimental to others, and improved status in the conservation community but degraded status amongst other farmers. To find the best compromise solution, the impact of each intervention on each group is assessed and the decision-maker weights the values for each group according to their perceived importance. Separate utilities can then be combined into one overall multi-attribute utility function. For example, van Calker *et al*. (2006) used expert and stakeholder input to develop a multi-attribute sustainability function for dairy farming. They found that the overall utility of different dairy farms varied depending on the stakeholder, with policy-makers and industrial farmers differing from consumers.

UNDERSTANDING AND PREDICTING BEHAVIOUR

Three main approaches to predicting human behaviour are likely to interest ecologists. These approaches (i) assume humans are rational optimizers; (ii) calculate the likelihood of a behaviour by evaluating an individual's motivations, the strength of belief that the behaviour will make a difference and the opinions of others on the consequences of the behavioural change; and (iii) describe macro-scale behaviour using phenomenological relationships.

Rational optimization

The simplest approach to predicting behaviour is to assume that individuals are rational, perfect optimizers and optimize utility. As in evolutionary biology, if the decisions of individuals are independent, then optimization is straightforward. Rounsevell *et al*. (2003) use a linear programming model that incorporates the costs and benefits associated with each crop type, as well as the time constraints, to predict how rotations vary between locations if farmers maximize profit.

If individual decisions depend upon the decisions of others then *game theory* is required, a technique used in both behavioural ecology (Sutherland 1996) and human systems. Breton, Zaccour & Zahaf (2006) show how game theory aids in understanding how countries respond to implementing joint environmental projects, such as the Kyoto Protocol.

Although utility optimization is widely used, it has been criticized, especially because it assumes that knowledge is freely available and that it is always possible for decision-makers to use that knowledge to construct an ordered set of choices (van den Bergh, Ferrier-i-Carbonell & Munda 2000). Techniques from the field of *bounded rationality* relax these assumptions (Rubinstein 1998), and attempt to predict behaviour when it is not possible to discriminate between all options, or to acquire information on the complete set of options available. In these circumstances, behaviour can often be modelled using *heuristics* (also known as rules of thumb), which are simple rules that achieve an approximately optimal outcome (Kahneman, Slovic & Tversky 1982). For instance, humans often use heuristics to solve game theoretical problems that occur in group situations.

Socio-psychological approaches

Socio-psychological methods predict behaviour by quantifying it as a product of values, beliefs or other psychological constructs rather than by assuming the existence of an objective optimum. This can be especially useful for behaviours that may be genuinely irrational, such as addiction, or those for which the costs and benefits are difficult to quantify in a utility framework (van den Bergh, Ferrier-i-Carbonell & Munda 2000).

The *Theory of Planned Behaviour* (Azjen 1991) builds upon the earlier *Theory of Reasoned Action* (Azjen & Fishbein 1980) and examines an individual's behavioural intention, quantifying the likelihood of behaving in a given manner. Intention depends upon a combination of attitude, subjective norm and perceived behavioural control. Attitude is assessed by determining the consequences of the behaviour and the extent to which these consequences matter to the individual. Subjective norm is the extent to which the behaviour fits with the expectations of others combined with the extent to which the individual wishes to meet these expectations. Perceived behavioural control is a product of factors that help or hinder a decision and the importance each has for the individual. Thus, according to the Theory of Planned Behaviour, whether you do something or not depends upon how good an idea you think it is, how much you care about what others think of the consequences and the extent to which you believe you are able to perform the behaviour. An advantage is the identification of belief-based barriers or drivers of behaviour that optimization assumptions do not account for. A disadvantage is that studies often rely on stated intentions rather than observed behaviour.

Mattison & Norris (2007) used the Theory of Planned Behaviour to explore the likelihood of farmers producing sugar beet and oilseed rape for biofuels. The farmers surveyed felt the decision was currently beyond their control and intentions were strongly driven by the ease with which crops fit into existing rotations (attitude). While these factors explain the intention score, Mattison and Norris also projected a range of biofuel production scenarios based on different hypotheses about the intention at which farmers would grow biofuel. They found that UK score biofuel targets lay outside even the most optimistic scenario.

Aggregated models

Aggregated approaches seek to describe high-level phenomena while discarding individual-level detail. In ecology, the logistic equation models average population increase based upon intrinsic growth rate and density dependence. Although the processes leading to density dependence are complex, and largely driven by the fate of different classes of individuals, the logistic equation captures the phenomenon of density dependence using just two parameters. Macroeconomics, market analysis and social networks theory often use phenomenological models of this type.

Macroeconomic models describe the behaviour of indicators, such as total production, consumption or employment, over a section of the economy. Linking such models with ecological processes is potentially very useful because it allows the effects of broad economic factors such as trade, taxes and credit supply to be evaluated with respect to their ecological outcomes (Persson & Munasinghe 1995). Glomsrod, Monge & Vennemo (1998) apply a macroeconomic approach to deforestation in Nicaragua. Their model showed that a policy of reduced government expenditure would initially drive an increase in deforestation, but that this would eventually reduce deforestation because of a decrease in rural poverty levels and intensification of agriculture on previously deforested land.

ASSESSING WELFARE

Utility can be used as a measure of individual or social welfare and models can be used to identify actions that maximize its value. In practice, the most frequently used measure of social welfare is an aggregate sum of the utilities of all individuals (Kumar 2002; Mishan & Quah 2007). There are implicit value judgments in using such aggregate sums; for example, that increased utility experienced by some individuals can compensate for decreases experienced by others (Boardway 1974). In addition, if the same measure of utility (e.g. monetary value) is used across a heterogeneous population, the overall welfare measure will be inequitable due to differences in utility functions between individuals.

The technique of cost–benefit analysis usually considers changes in welfare on a single utility scale. When there are multiple components to utility, such as income and an environmental quality, each is considered separately. An alternative approach when there are multiple components to utility is to calculate the *Pareto efficient frontier* to examine the trade-off between them. This comprises a series of so called 'dominant' (preferred) solutions from which it is impossible to improve one component of utility without compromising another. Because these solutions can occur at different relative values of the two objectives, they form a trade-off curve (efficiency frontier). Polasky *et al.* (2005) sought to determine the best compromise between various conflicting land uses for a landscape in Oregon. For a range of landscape configurations, they calculated a biological and an economic score and used numerical optimization to find the landscapes that lay along the Pareto-efficient frontier providing the best combinations of biological and economic scores. In this case, landscapes with a high biological score could be achieved for a small reduction in the economic score.

The application of integrated social and ecological models in agro-ecology

Here we examine the ways that social science methods have been applied to ecological contexts. Our focus is specifically on models from agriculture and silviculture, because these subjects involve a pervasive interaction between humans and their natural environment and form a reasonably coherent research area. Other key areas of interaction include wild harvesting (e.g. fisheries), epidemiology and urban expansion. Many of our results will also be applicable to these fields.

We attempted to find all quantitative models published in the past 5 years (April 2003 to April 2008) that included both social and ecological variables. Social concepts are defined in our taxonomy, and to qualify as ecological, the model must have been applied to wild organisms. We performed our search by scanning the tables of contents of 27 ecological, social and interdisciplinary journals. The resulting list of 36 integrated modelling studies came from 12 journals, and just three of these, *Ecological Economics*, *Biological Conservation*, and *Ecological Modelling* contained half (18) of the papers found. We analysed all of the models in detail, classifying each according to the social concepts employed, conservation management implications, how outputs were validated, and whether there was feedback between social and ecological systems. A full list of papers and model details is available in Supporting Information, Appendix S1. With few exceptions, the social aspects of models were based on quantification of social welfare or utility, but the methodologies varied considerably. The first two sections of our review examine these methodologies. We then briefly examine the management implications raised by the models and the ways that their outputs were validated.

WELFARE BASED POLICY ASSESSMENT

Almost all models used utility as a measure of welfare (to assess desirability of outcomes), even if there was no explicit statement about this. We classified these models into four categories summarizing the spatial scale, the role of human decision-making and the types of recommendation in the models (see Fig. 3). The categories are: *Private Profit* (pure profit models at the single landholder level), *Private Conservation* (models of private profit and provision of public conservation benefits at the single land-holder level), *Collective Management* (landscape models without consideration of individual behaviour) and *Constrained Policy* (landscape models, taking individual behaviour into account).

Private Profit models describe human welfare at the level of a single self-interested decision-maker. Most of these evaluated ecological outcomes purely in terms of their effect on profit. Examples came from forestry (Wam *et al*. 2005) and from range management (Quaas *et al*. 2007) where domestic animals are dependent on the ecological services provided by wild pastures. Since the ecological aspects of these systems are directly related to production (moose, cattle or wood harvesting), their value can be expressed in terms of the market value of production.

The *Private Conservation* model category focussed on single landholders but split utility into two parts, a private profit component and an ecological benefits component. Since the ecological benefit largely consisted of benefits to society (van Wenum, Wossink & Renkema 2004; Tichit *et al*. 2007), we considered it to be an external to the farmer's decision-making process (Fig. 3): in economic terminology it was an *externality*.

Both the *Collective Management* and *Constrained Policy* templates are concerned with aggregate or 'group' measures of utility rather than individual ones, but *Collective Management* models do not explicitly make predictions of individual behaviour. *Collective Management* models typically consider a landscape where there is spatial variation in contributions to welfare from profit and from ecological services. *Collective Management* models calculate the change in welfare associated with a change in landscape configuration or a change in policy affecting profitability of different land uses. In some cases, numerical optimization was used to find land-use configurations that optimized contributions to welfare from different landscape objectives – usually profit and biodiversity (Polasky *et al*. 2005).

The *Constrained Policy* approach makes an explicit distinction between decisions made by individual landholders and those made at the policy level (Berger & Bolte 2004; Gottschalk

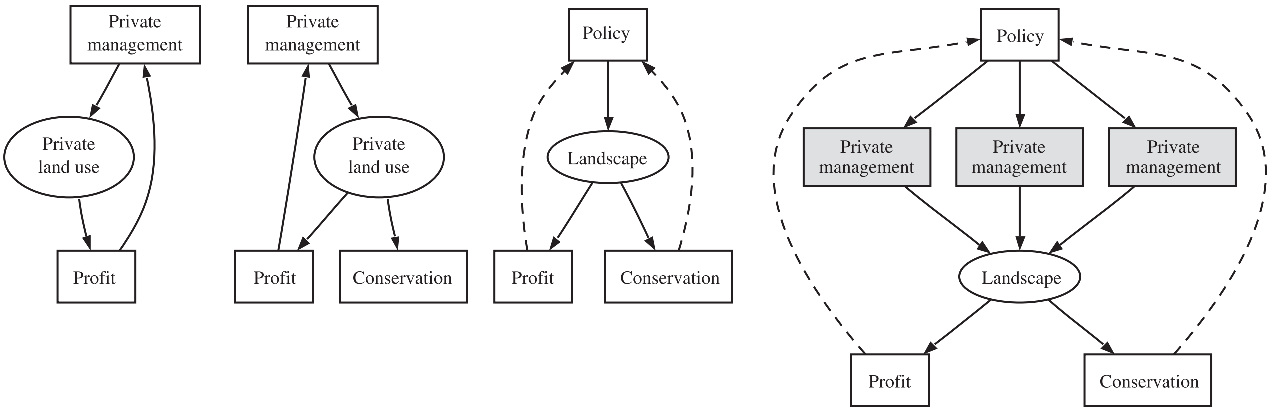

Fig. 3. Social organization in the different model categories. From left to right they are Private Profit, Private Conservation, Collective Management and Constrained Policy. The topmost box in each case represents the social level at which welfare is calculated, with arrows indicating the flow of effects in the model. Shaded boxes indicate aspects of the model involving behavioural prediction. The ovals indicate the spatial scale at which land-use was calculated. Lines from profit and conservation objectives back to the decision-making parts of the model indicate the possibility of feedback. Dashed lines indicate feedback loops that are not present in all models.

et al. 2007). Land-use patterns arise as a consequence of decisions made by individual landholders whose behaviour is influenced by policy. The policy aspect of these models can be implemented by formulating discrete scenarios. For example, Berger & Bolte (2004) determined the land-use patterns resulting from three policies designed by stakeholders. An alternative to analysing scenarios is to calculate optimal policy decisions. Drechsler *et al.* (2007b) optimized a subsidy framework for the timing of mowing to maximize conservation outcomes for a fixed conservation budget. In that study, variable costs of conservation between farmers were calculated to determine the uptake of mowing schemes at different payment levels. A single fixed payment across the landscape was assumed, reflecting a constraint to the flexibility of policy. Such policy constraints inevitably lead to inefficiencies, but are interesting because they reflect practical realities.

PREDICTIONS OF BEHAVIOUR

Only eight studies attempted to predict human behaviour, and of these, four used a rule-based or bounded-rationality approach, three used utility optimization and one used an aggregated model. All of the rule-based models were implemented as decision-trees in computer simulations while the rule-sets were based on interviews with farmers (Retzer & Reudenbach 2005; Boone *et al.* 2006). One of the studies (Berger & Bolte 2004) used a multi-criteria decision analysis technique, called TOPSIS (Hwang & Yoon 1981) as a rule-set. In this case, the variables supplied to the TOPSIS technique would have meant that it acted as a heuristic for risk-averse profit maximization. The other three studies also used rule-sets that were based around profit or production variables as well as some element of risk. For example Retzer & Reudenbach (2005) assumed that graziers maximized livestock while minimizing losses, whereas Boone *et al.* (2006) assumed that farmers attempted to meet caloric needs while maintaining required levels of livestock and cash. Although ultimately the rule-sets in all of these studies can be broadly interpreted as heuristics, none was built upon formal psychological theories of such behaviour (e.g. Rubinstein 1998; Brocas & Carrillo 2004).

Of the three studies that used utility optimization to predict behaviour, one based utility entirely on profit (Gottschalk *et al.* 2007), one included conservation attitude as a stochastic contribution to utility (Drechsler *et al.* 2007b), and one included quantified non-market values, such as recreation and conservation, in the utility calculation (Kurttila & Pukkala 2003).

MODEL CERTAINTY AND APPLICATION

The range of applications spanned by integrated models was broad and included analysis of human wildlife conflict, assessments of the costs of conservation strategies, optimal formulation of agricultural subsidies or of land-use to deliver biodiversity outcomes at least cost, and assessment of the sustainability of agricultural practice. Integrated models adopted a broad range of philosophies from the very abstract and general (e.g. Quaas *et al.* 2007 to the very specific (e.g. Boone *et al.* 2006). Abstract modelling papers tended to focus on communicating model concepts through exploration of model behaviour and never included an explicit comparison with empirical data. In the case of more specific and directly applied models, we found a number of examples where validation against independent data was performed, but overall the emphasis on model validation and evaluation of uncertainty was low. Many integrated models were quite complex, being built of several components but in many cases only one aspect of the model was validated, and where independent data were presented, the model fit to this was often not statistically quantified.

We found that ecological principles were distributed relatively equally across the different types of social approach used and that many studies also modelled bidirectional feedbacks between the two systems. This is elaborated in further detail in Supporting Information, Appendix S2.

Discussion

Many models exist that quantify either ecological or social aspects of land-use systems, but models that integrate the two are far less common. For instance, a recent 910-page volume on modelling agricultural policies mentions no fewer than 50 models for Europe alone (Arfini 2005) and there are many ecological models for wild organisms in agricultural environments. Yet in our comprehensive survey, we identified just 36 examples of quantitative integrated models. Although this analysis focuses specifically on agriculture, many of the issues around integrating models in that discipline are likely to apply to other ecological fields such as fisheries, epidemiology and urban expansion.

One advantage of developing integrated models is that important effects may be revealed that are not present in models of ecology or socio-economics alone. For example, Watkinson *et al.*'s (2000) model of arable weeds and farmland birds concluded that a socio-economic effect [uptake of genetically modified (GM) technology] played an important role in predicting the impact of GM technology on birds. The critical issue was whether the farmers with weed-rich fields are especially likely to adopt GM technology (as they would have greatest benefit) or especially unlikely to adopt (for example, because they are organic farmers or are late in adopting the latest technology). Thus, the interaction between an ecological variable (field weed population) and a social one (attitude to GM crops) was crucial to the outcome.

Integrated models may also play a role in facilitating knowledge transfer between ecology and socio-economics by bringing researchers from those disciplines together to work on a common project. Whether such exchanges will produce results that are useful to policy-makers depends on both model accuracy and on the ability of models to frame their outputs in terms that are directly policy relevant. Since policy is often made at a broad scale, the 'Constrained Policy' and 'Collective Management' templates outlined in our review are likely to be most useful in this regard.

Ensuring model accuracy depends on both ecological and economic model accuracy as well as the ability of researchers to quantify the variables that link the two. In the case of socio-ecological models these linking variables often consist of quantified human welfare values related to nature (see Fig. 2), or the impacts of human activity on ecological systems. In our review, we found that estimates of human welfare were typically confined to market quantities. In order to move beyond this narrow viewpoint, researchers will need to make greater use of a wider range of socio-economic techniques such as those outlined in Fig. 2. None of these techniques is unbiased, but focussing purely on market-valued quantities only acts to re-enforce one particular form of bias. If possible, results from different techniques should be compared (Azevedo, Herriges & Kling 2003).

In our review of integrated models, we identified two key areas where improvements must be made if this work is to be useful for policy-makers. Firstly, a greater variety of techniques quantifying human values should be used. Secondly, there needs to be a greater emphasis on assessing the credibility of integrated models. Ecologists with a sound understanding of social science modelling techniques will be in a position to tackle both these problems. The specialist data requirements (for both parameterization and validation) of integrated models necessitate excellent communication between researchers in the early stages of a project. This can only occur when both ecologists and social scientists understand the other discipline. Our taxonomy of approaches and review of models will assist ecologists in this regard.

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References

Adamowicz, W.L., Louviere, J. & Williams, M. (1994) Combining stated and revealed preference methods for valuing environmental amenities. *Journal of Environmental Economics and Management*, **26**, 271–292.

Arfini, F. (2005) *Modelling Agricultural Policies: State of the Art and New Challenges*. Proceedings of the 89th Seminar of the European Association of Agricultural Economists. Monte Universitá Parma, Parma, Italy.

Azevedo, C.D., Herriges, J.A & Kling, C.L. (2003) Combining revealed and stated preferences: consistency tests and their interpretations. *American Journal of Agricultural Economics*. **85**, 525–537.

Azjen, I. (1991) The theory of planned behaviour. *Organizational Behaviour and Human Decision Processes*, **50**, 179–211.

Azjen, I. & Fishbein, M. (1980) *Understanding Attitudes and Predicting Social Behaviour*. Prentice Hall, Englewood Cliffs, NJ, USA.

Bateman, I., Carson, R.T., Day, B., Hanemann, W.M., Hanley, N., Hett, T., Jones, A., Loomes, G., Mourato, S., Ozdemiroglu, E., Pearce, D.W., Sugden, R., Swanson, J. (2002) *Economic Valuation with Stated Preferences Techniques. A Manual*. Edward Elgar, Cheltenham, UK.

Beckerman, W. & Hepburn, C. (2007) Ethics of the discount rate in the stern review on the economics of climate change. *World Economics*, **8**, 187–210.

Belton, V & Stewart, T.J. (2002) *Multiple Criteria Decision Analysis: An Integrated Approach*. Kluwer Academic Publishers, Dordecht, The Netherlands.

Berger, P.A. & Bolte, J.P. (2004) Evaluating the impact of policy options on agricultural landscapes: an alternative-futures approach. *Ecological Applications*, **14**, 342–354.

Boardway, R.W. (1974) The welfare foundations of cost–benefit analysis. *The Economic Journal*, **84**, 926–939.

Boone, R.B., Galvin, K.A., Thornton, P.K., Swift, D.M. & Coughenour, M.B. (2006) Cultivation and conservation in Ngorongoro conservation area, Tanzania. *Human Ecology*, **34**, 809–828.

Breton, M., Zaccour, G. & Zahaf, M. (2006) A game-theoretic formulation of joint implementation of environmental projects. *European Journal of Operational Research*, **168**, 221–239.

Brocas, I. & Carrillo, J.D. (2004) *The Psychology of Economic Decisions*, Vol.1. Rationality and Well-Being. Oxford University Press, New York.

Courchamp, F.,Angulo, E., Rivalan, P., Hall, R.J., Signoret, L., Bull, L. & Meinard, Y. (2006) Rarity value and species extinction: the anthropogenic Allee effect. *PLoS Biology*, **4**, 2405–2410.

Drechsler, M., Grimm, V., Mysiak, J. & Wätzold, F. (2007a) Differences and similarities between ecological and economic models for biodiversity conservation. *Ecological Economics*, **62**, 232–241.

Drechsler, M., Watzold, F., Johst, K., Bergmann, H. & Settele, J. (2007b) A model-based approach for designing cost-effective compensation payments for conservation of endangered species in real landscapes. *Biological Conservation*, **140**, 174–186.

Ewers, R.M. & Rodrigues, A.S.L. (2006) Speaking different languages on biodiversity. *Nature*, **443**, 506–506.

Fox, H.E., Christian, C., Nordby, J.C., Pergams, O.R.W., Peterson, G.D. & Pyke, C.R. (2006) Perceived barriers to integrating social science and conservation. *Conservation Biology*, **20**, 1817–1820

Glomsrod, S., Monge, M.D. & Vennemo, H. (1998) Structural adjustment and deforestation in Nicaragua. *Environment and Developmental Economics*, **4**, 19–43.

Gottschalk, T.K., Diekotter, T., Ekschmitt, K., Weinmann, B., Kuhlmann, F., Purtauf, T., Dauber, J. & Wolters, V. (2007) Impact of agricultural subsidies on biodiversity at the landscape level. *Landscape Ecology*, **22**, 643–656.

Hwang, C.L. & Yoon, K. (1981) *Multiple Attribute Decision Making: Methods and Applications*. Springer, Berlin .

Kahneman, D., Slovic, P. & Tversky, A. (1982) *Judgment under Uncertainty: Heuristics and Biases*. Cambridge University Press, Cambridge, UK.

Kumar, S. (2002) Does 'participation' in common pool resource management help the poor? A social cost–benefit analysis of joint forest management in Jharkhand, India. *World Development*, **30**, 763–782.

Kurttila, M. & Pukkala, T. (2003) Combining holding-level economic goals with spatial landscape-level goals in the planning of multiple ownership forestry. *Landscape Ecology*, **18**, 529–541.

Lawton, J.H. (2007) Ecology, politics and policy. *Journal of Applied Ecology*, **44**, 465–474.

Lowenstein, G. & Adler, D. (1995) A bias in the prediction of tastes. *The Economic Journal*, **105**, 929–937.

Mascia, M.B., Brosius, J.P., Dobson, T.A., Forbes, B.C., Horowitz, L., McKean, M.A., Turner, N.J. (2003) Conservation and the social sciences. *Conservation Biology*, **17**, 649–650.

Mattison, E.H.A. & Norris, K. (2007) Intentions of UK farmers toward biofuel crop production: implications for policy targets and land use change. *Environmental Science & Technology*, **41**, 5589–5595.

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

Mishan, E.J. & Quah, E. (2007) *Cost–Benefit Analysis*, 5th edn. Routledge, Oxford, UK .

Morandin, L.A. & Winston, M.L. (2006) Pollinators provide economic incentive to preserve natural land in agroecosystems. *Agriculture, Ecosystems & Environment*, **116**, 289–292.

Naidoo, R., Balmford, A., Ferraro, P.J., Polasky, S., Ricketts, T.H. & Rouget, M. (2006) Integrating economic costs into conservation planning. *Trends in Ecology & Evolution*, **21**, 681–687.

Parsons, G.R. (2003) The travel cost model. *A Primer on Nonmarket Valuation* (eds P.A. Champ, K.J. Boyle & T.C. Brown), pp. 269–330. Springer, Berlin.

Peck, S.L. (2004) Simulation as experiment: a philosophical reassessment for biological modelling. *Trends in Ecology & Evolution*, **19**, 530–534.

Persson, A. & Munasinghe, M. (1995) Natural resource management and economy-wide policies in Costa Rica: a computable general equilibrium (CGE) modeling approach. *The World Bank Economic Review*, **9**, 259–285.

Polasky, S., Nelson, E., Lonsdorf, E., Fackler, P. & Starfield, A. (2005) Conserving species in a working landscape: land use with biological and economic objectives. *Ecological Applications*, **15**, 1387–1401.

Quaas, M.F., Baumgartner, S., Becker, C., Frank, K. & Muller, B. (2007) Uncertainty and sustainability in the management of rangelands. *Ecological Economics*, **62**, 251–266.

Retzer, V. & Reudenbach, C. (2005) Modelling the carrying capacity and coexistence of pika and livestock in the mountain steppe of the South Gobi, Mongolia. *Ecological Modelling*, **189**, 89–104.

Rounsevell, M.D.A., Annetts, J.E., Audsley, E., Mayr, T. & Reginster, I. (2003) Modelling the spatial distribution of agricultural land use at the regional scale. *Agriculture, Ecosystems and Environment*, **95**, 465–479.

Rubinstein, A. (1998) *Modeling Bounded Rationality*. MIT Press, Cambridge, UK.

Stern, N (2007) *The Economics of Climate Change: the Stern Review*. Cambridge University Press, Cambridge, UK.

Stevens, T.H., Echeverria, J., Glass, R.J., Hager, T. & More, T.A. (1991) Measuring the existence value of wildlife: what do CVM estimates really show? *Land Economics*, **67**, 390–400.

Sutherland, W.J. (1996). *From Individual Behaviour to Population Ecology*. Oxford University Press, Oxford, UK

Sutherland, W.J. (2006) Predicting the ecological consequences of environmental change: a review of the methods. *Journal of Applied Ecology*, **43**, 599–616.

Sutherland, W.J., Bailey, M.J., Bainbridge, I.P., Brereton, T., Dick, J.T.A., Drewitt, J., Dulvy, N.K., Dusic, N.R., Freckleton, R.P., Gaston, K.J., Gilder, P.M., Green, R.E., Heathwaite, A.L., Johnson, S.M., Macdonald, D.W., Mitchell, R., Osborn, D., Owen, R.P., Pretty, J., Prior, S.V., Prosser, H., Pullin, A.S., Rose, P., Stott, A., Tew, T., Thomas, C.D., Thompson, D.B.A., Vickery, J.A., Walker, M., Walmsley, C., Warrington, S., Watkinson, A.R., Williams, R.J., Woodroffe, R. & Woodroof, H.J. (2008) Future novel threats and opportunities facing UK biodiversity identified by horizon scanning. *Journal of Applied Ecology*, **45**, 821–833.

Taylor, L.O. (2003) The hedonic method. *A Primer on Nonmarket Valuation* (eds P.A. Champ, K.J. Boyle & T.C. Brown), pp. 331–393. Springer, Berlin.

Tichit, M., Doyen, L., Lemel, J.Y., Renault, O. & Durant, D. (2007) A co-viability model of grazing and bird community management in farmland. *Ecological Modelling*, **206**, 277–293.

van Calker, K.J., Berentsen, P.B.M., Giesen, G.W.J. & Huirne, R.B.M. (2006) Development and application of a multi-attribute sustainability function for Dutch dairy farming systems. *Ecological Economics*, **57**, 640–658.

van den Bergh, J.C.J.M., Ferrier-i-Carbonell, A. & Munda, G. (2000) Alternative models of individual behaviour and implications for environmental policy. *Ecological Economics*, **32**, 43–61.

van Wenum, J.H., Wossink, G.A.A. & Renkema, J.A. (2004) Location-specific modeling for optimizing wildlife management on crop farms. *Ecological Economics*, **48**, 395–407.

Venkatachalam, L. (2004) The contingent valuation method: a review. *Environmental Impact Assessment Review*, **24**, 89–124.

Wam, H.K., Hofstad, O., Navdal, E. & Sankhayan, P. (2005) A bio-economic model for optimal harvest of timber and moose, *Forest Ecology and Management*, **206**, 207–219.

Watkinson, A.R., Freckleton, R.P., Robinson, R.A. & Sutherland, W.J. (2000) Predictions of biodiversity response to genetically modified herbicide-tolerant crops. *Science*, **289**, 1554–1557.

Wätzold, F., Drechsler, M., Armstrong, C., Baumgärtner, S., Grimm, V., Huth, A., Perrings, C., Possingham, H., Shogren, J., Skonhoft, A., Verboom-Vasiljev, J. & Wissel, C. (2006) Ecological–economic modeling for biodiversity management: potential, pitfalls, and prospects. *Conservation Biology*, **20**, 1034–1041.

White, P.C.L., Jennings, N.V., Renwick, A.R. & Barker, N.H.L. (2005) Questionnaires in ecology: a review of past use and recommendations for best practice. *Journal of Applied Ecology*, **42**, 421–430.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. List and details of reviewed papers on social and ecological modelling

Appendix S2. Use of ecological concepts and information flow

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