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Principles of Landscape Design that Emerge from a Formal Problem-Solving Approach

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Abstract

For landscape and conservation ecology to become useful branches of applied ecology, they must be able to inform policy and management decisions, and help solve real problems within the constraints of economic reality. This means applying problem-solving tools used commonly in economics, applied mathematics and engineering. The crucial role of landscape ecologists in conservation is to provide relationships between biodiversity and attributes of the landscape that can be changed by management or policy, such as the configuration of protected or restored areas. We set out seven principles of landscape design for biodiversity based on a formal problem-solving approach. We illustrate these principles by providing a general formulation of the spatial conservation resource allocation problem and with some examples from the recent literature. We argue that good management decisions can only be made using a formal decision-theory approach, where the objectives are clearly stated, constraints such as finances are included, and the ecological information required to formulate the problems are transparent. In this way decisions can be made with full acknowledgement of the assumptions and uncertainties in the process, and we can learn from past successes and failures through a process of active adaptive management.

Keywords: conservation resource allocation; decision theory; management objectives; problem formulation; process models.

Introduction

If landscape and conservation ecology are to become useful arms of applied ecology, then they must borrow problem-solving concepts from economics, applied mathematics and engineering. Economists, applied mathematicians and engineers pose and solve real problems within the constraints of economic reality. They use rational decision-making processes, where the objectives, constraints and underlying assumptions and uncertainties are explicit. This **decision-theory approach** can and should be applied to landscape design and environmental management (Shea *et al.* 1998; Possingham *et al.* 2001). Unfortunately, landscape and conservation ecologists expend much of their effort trawling for ecological patterns and trying to disprove null hypotheses that are invariably false (Gerber *et al.* 2005). Worse still, some conservation biologists devise crude decision-support schemes that have no clear objective, ignore costs and/or rely on arbitrary weighting and scoring methods to find solutions. This risks bringing the field into disrepute and renders conservation efforts inefficient. Instead, applied landscape ecologists should strive to provide problem-solvers with statistical and/or process-based relationships between biodiversity and attributes of the landscape that can be changed by management or policy. We need to be clear that **science informs policy and management; it does not make policy and management decisions.**

In this essay we discuss a framework for posing and solving problems in landscape design for the conservation of biodiversity. Using a formal decision-theory approach, objectives are clearly stated, finances are included, and the ecological information required to formulate the problems is transparent. We provide a new and general description of the Spatial Conservation Resource Allocation Problem (SCRAP) to illustrate our point, where the question posed is: how should conservation resources be distributed in time and space to different activities to minimize the loss of biodiversity in a region? We propose seven principles of landscape design, summarized at the end of the chapter. These ideas are illustrated throughout in the context of the formulation of the spatial conservation resource allocation problem and with some examples from the recent literature.

What is the goal?

Defining the precise goal of a conservation problem is often the most important and difficult step in a rational decision-making process (Shea *et al.* 1998; Possingham & Wilson 2005), and is probably the point at which most attempts falter. Here are two examples, from the management of fire regimes and the prioritization of habitats for conservation.

First, agencies will argue about whether the goal of fire management is to have more or less frequent fires in a conservation park. This implies that the action is the goal rather than the outcome, which is a common mistake, especially where governments are involved. A better goal might be to maintain a fraction of the park in each of several post-fire succession states, or manage the park to minimize the number of extinctions. We have posed and solved a simple version of this problem (Richards *et al.* 1999) where the objective was to maximize the amount of time that each of three successional states covers 20% of the park, and where fire management decisions were the actions.

Second, in landscapes where we are attempting to prevent the destruction of the most important pieces of habitat, a frequent response by agencies is to assign each habitat patch a score based on a variety of interdependent attributes like patch size, patch shape, number of threatened species in the patch, species richness of the patch, patch location relative to other patches and so forth; an example is the Common Nature Conservation Classification System that is widely used in Queensland (Chenoweth EPLA 2000). The amalgamated score, which invariably has no well-defined physical dimensions, is then used to rank the patches forming an order of merit (e.g. EPA 2002; Root *et al.* 2003; Parkes *et al.* 2004). The score/merit of each patch can lead to multiple responses – acquire for conservation, protect from development, manage for biodiversity – in the absence of any stated goal (Mace *et al.* 2006). This kind of approach, which has been used in decision-making from wetland trading (Stein *et al.* 2000) to landscape prioritization for conservation and restoration (Fattorini 2006; Twedt *et al.* 2006), ignores two things. First it ignores the vast literature on conservation planning dating back to the mid-1980s, which attempts to formulate such problems properly (Cocks & Baird 1989). Second, the scoring approach fails to recognize that individual patches (or actions) cannot be ranked in isolation because their value depends on other decisions made in the landscape – **the whole is more than the sum of the parts.** Whether dealing with weed incursions or the dispersal of wildlife between habitat patches, the management of the whole landscape influences the value of an action on a particular land parcel.

Broadly speaking there are two kinds of approaches to systematic landscape design. The first is to meet a variety of biodiversity targets while minimizing net expected costs – for example, the minimum set problem in conservation planning (Cocks & Baird 1989; Pressey *et al.* 1989; Possingham *et al.* 1993; Meir *et al.* 2004). The second is maximizing biodiversity benefit within a fixed budget – for example, the maximal coverage problem of conservation planning (Underhill 1994; Nicholson & Possingham 2006; Wilson *et al.* 2006). Both have the advantage that they do not attempt to mix the two currencies, biodiversity benefit and money. Amalgamating biodiversity benefit and money into a single currency requires the use of either valuation methods (Bandara & Tisdell 2004), or explicit political trade-offs, both of which are contentious.

Both the minimum set and maximal coverage problems include an implicit trade-off between money and biodiversity. The implicit trade-off in the minimum set problem is the level of the biodiversity target (e.g. 30% of the distribution of every species must be conserved), while it is the size of the budget in the case of the maximal coverage problem. Either way, **inclusion of finances is essential,** whether through the objective (minimize total cost) or through a resource constraint (annual budget). Ignoring money in landscape design is like advising someone how to do their weekly shopping on the assumption that every item in the supermarket costs the same amount.

In this chapter we will focus our discussion on problems where we maximize biodiversity benefit within a fixed budget. The thinking is illustrated with a fairly general formulation that we call the Spatial Conservation Resource Allocation Problem (SCRAP).

The Spatial Conservation Resource Allocation Problem (SCRAP)

Consider a landscape that is divided into planning units (polygons that cover the entire two-dimensional region of interest). In each planning unit in the landscape we can take a variety of actions; for example, acquire land for conservation, acquire some component of the development rights, fence, control feral predators and/or reduce the probability of fire.

We will assume that there is a budget of B_t dollars available per year, t, of the management time horizon T. Let the amount of money spent on action k

in planning unit i in year t be b_{ikt}. Each year the total cost of all actions must be less than or equal to the budget. Assume that the set of these actions across all the planning units, $\mathbf{b}_{kt}$, changes the state of the system in year t, $\mathbf{y}_t$, and both the set of actions and the system state can affect the chance of persistence of any species, j, in the system. Given these definitions a general formulation of the SCRAP with species persistence as the prime objective is

$$\text{Maximize} \quad \sum_i P_{jT}(\mathbf{y}_0, \mathbf{b}_{k1},...,\mathbf{b}_{kT}) \tag{45.1}$$

where P_{jT} is the probability that species j is extant by year T (a terminal time at which our plan is evaluated), given the initial state of the system and the sequence of actions each year. The sum of persistence probabilities gives the expected number of species surviving at the end of the management time horizon.

This objective is maximized subject to a series of constraints and equations describing the dynamics of the system (ecosystem and population models). The **constraints** include the annual budget,

$$\sum_i \sum_j b_{ikt} \leq B_t \quad \text{for all years } t = 1, ..., I, \tag{45.2}$$

and constraints on the resources allocated to what actions can occur in different planning units (e.g. selective logging could be ruled out in estuaries).

The **system dynamics** are equations describing the way in which the **system state**, $\mathbf{y}_t$, changes due to actions, system parameters and external stochastic forces. For example, with our fire management problem (Richards *et al.* 1999), the system state is the fraction of the park in each successional state, and this state changes stochastically depending on the process of succession and random fire events, both modified by management actions. The system state each year, $\mathbf{y}_t$, can include not just attributes of the population of every species (including its genetic structure) but also aspects of the environment in which the species live.

The SCRAP assumes that the overall objective is to maximize the number of species that persist over some long timeframe. We may choose to generalize this objective by including other aspects of the environment that people would like to conserve, like some measures of water flow, and provide rewards for every year that these services (including species existence services) are provided. The possibilities are endless, although adding complexities then requires knowledge

about how the actions and random events affect ecosystem services, and interactions between all these services. **Describing and predicting how the state of an ecosystem determines outcomes like extinction probabilities and the flows of ecosystem services is the realm of scientists** – this is the task to which landscape ecologists should apply themselves, a point we return to below.

While formulating the general resource allocation problem is intellectually engaging, it is of little practical value unless we can be more specific. Here we provide specific examples of the SCRAP.

Example 1: The classic 'maximum coverage' conservation planning problem

The classical conservation planning problem (Kirkpatrick 1983; Cocks & Baird 1989) and its relatives represent the only widespread and generally accepted use of decision theory in conservation biology. The simplest version of the objective is to ensure that a fixed fraction of a variety of conservation features (e.g. habitat types and species) is conserved in protected areas for the minimum total cost. The related problem is the maximal coverage problem where, given a fixed budget, we try to maximize biodiversity gains, such as ensuring that as many features as possible meet a fixed conservation target. These problems are time-independent, ignoring both the dynamics of the species and the landscape during the process of acquisition, and only one action is possible: reserve acquisition. The performance of conservation action is evaluated solely on the final reserve system.

Overall this is a well-posed and sensible approach. Two concerns with this version of the SCRAP are the lack of dynamics and the fact that the performance of the system is solely a function of the number of species and/or habitat that meet a prespecified target. In reality we have no guarantee that meeting an arbitrarily set target ensures persistence. Most targets are based on political pragmatism rather than ecological principles (Kirkpatrick 1998; Cabeza & Moilanen 2001; Reyers *et al.* 2002). However, with some thought we are getting better at defining 'adequate' targets; for example, setting species-specific targets for the area or population size required for viability, adjusting targets according to levels of threat faced and considering large-scale movement and evolutionary dynamics (e.g. Cowling *et al.* 1999; Nicholls 1999; Burgman *et al.* 2001; Cowling & Pressey 2001; Verboom *et al.* 2001; Pressey *et al.* 2003). Ignoring time simplifies the problem but leaves the manager unclear about how to prioritize between options and when to take actions. Meir *et al.* (2004) describe a dynamic version of the problem where planning units can be destroyed and the acquisition process needs to meet the constraints of an annual budget.

Example 2: The optimal habitat restoration problem

The optimal habitat restoration problem is a minor variant on the maximal coverage problem, but now the expenditure is not on acquisition but restoration. If we know the 'cost' (where cost may have an economic, social and political dimension) of restoring any planning unit and the benefits of those planning units to meeting species targets then we can find optimal plans for restoring habitat (Westphal *et al.* 2003; Westphal & Possingham 2003).

Ideally we would know how landscapes affect viability, but in the absence of this information, we may rely instead on statistical models relating landscapes variables to habitat suitability – for example planning unit i has an x% chance of having species j if it is restored to a certain habitat quality and it is embedded in landscape y. This sits comfortably with many ecologists as it utilizes the methods of statistical habitat models. The problem is that we do not know how statistical habitat models translate into species viability.

Example 3: The optimal patch protection problem

In an attempt to include species persistence in landscape planning, we used models of metapopulation viability to find optimal reserve systems that maximize the long-term persistence of multiple species (Nicholson & Possingham 2007; Nicholson *et al.* 2006). The objective relates directly to extinction risk, a function of the ecology of the species and the amount, quality and spatial configuration of habitat in the landscape (Frank & Wissel 2002). Although able to deal with some spatial dynamics, the current formulation in Nicholson *et al.* (2006) does not include the temporal dynamics of the problem described in the SCRAP formulation. Furthermore, the choice of objective function – for example, maximize number of species that persist, versus minimize the probability any species goes extinct – can alter the outcome, and therefore needs to be carefully considered (Nicholson & Possingham 2006).

Huge challenges remain in furthering this approach, in particular for more complex landscapes and combinations of species. The metapopulation models for species viability are approximations of idealized systems. We have no simple way of converting more complex landscapes into probabilities of persistence for use in a reserve optimization framework. Even when using

a metapopulation approach, with its relatively simple view of the landscape and species, we rarely have sufficient information on species to put values to many of the parameters. Solving these challenges relies on moving from statistical relationships of patterns of presence/abundance into process-based viability models. In these mechanistic models the fundamental population processes of birth, death and movement will be complex functions of the landscape – easy to say but daunting to address empirically (Tyre *et al.* 2001).

What use is a landscape ecologist?

A discussion of finding better functional forms relating management actions to biodiversity benefits leads us back to the role of landscape ecology. Good objectives are directly related to biodiversity outcomes – for example, extinction probabilities. Bad objectives are vague and uncertain surrogates for biodiversity – for example, landscape metrics or combinations of landscape metrics. The reason people don't always use the good objectives is because they are hard to calculate and parameterize. Crude surrogates, like the 'habitat hectares' metric (McCarthy *et al.* 2004; Parkes *et al.* 2004) that represents a rough measure for the condition and extent of habitat, are easier to measure and optimize. Many functions are available for describing the relationship between biodiversity benefit and increasing cost or effort (Hof & Raphael 1993; Arponen *et al.* 2005). The difficulty lies in relating this benefit function to meaningful outcomes. When the benefit of an action is estimated without a basis in ecological theory, and when we use a rough surrogate for persistence, such as the amount of habitat or a landscape metric, we run the risk of optimizing something that is not directly related to the outcomes we want, such as species persistence.

We believe that the primary task of a useful landscape ecologist is to determine the relationships between measurable surrogates and real biodiversity outcome measures – for example, turn landscape pattern metrics and habitat condition metrics into probabilities of presence (e.g. Lindenmayer *et al.* 2002; Wintle *et al.* 2005), and, better still, probabilities of persistence. To be succinct, **landscape ecologists should be trying to define the functional forms of P**, or equally well-defined and logical objectives (eqn. 45.1) in the SCRAP.

Solving the problem

Although the problems that we have formulated may seem very large and complex, finding a range of good solutions poses few difficulties. The SCRAP is technically solvable using stochastic dynamic programming, although simplifying assumptions are required for exact solutions to be found due to computational constraints. Alternatively, approximate near-optimal solutions may be obtained using modern optimization tools, such as genetic algorithms and simulated annealing. Many of these search algorithms have already been used to solve conservation planning problems (e.g. Moilanen & Cabeza 2002; Westphal *et al.* 2003; Nicholson *et al.* 2006). Finding the very best solution to these problems is fortunately irrelevant for most real-world problems. Despite the unrealistic calls for more exact optimization (Rodrigues & Gaston 2002; Fischer & Church 2005), there is so much uncertainty in the models, data and politics of any real problem that a variety of good alternatives is far more useful to decision-makers and managers. Future challenges include devising good methods to find and communicate good solutions to the SCRAP, in particular robust and accessible rules of thumb (Regan *et al.* 2006; Wilson *et al.* 2006).

Dealing with uncertainty

Practitioners often assume that large amounts of uncertainty about parameters and processes make a formal decision-making process useless. Ironically it is under great uncertainty that decision-making tools can provide the most substantial advantages over raw human intuition. By borrowing methods from applied mathematics, economics and engineering, we are continually expanding the toolbox for dealing with ecological uncertainties (e.g. Drechsler 2004; Regan *et al.* 2005; Nicholson & Possingham 2007). It is worth noting that when considering uncertainty our objective may change. For example, rather than maximizing the expected outcome (like maximizing expected profit when investing in the stock market), we may wish to minimize the uncertainty about the outcome (Milner-Gulland *et al.* 2001), or maximize robustness to uncertainty by choosing the management strategy that delivers a particular performance outcome subject to the greatest amount of uncertainty (Ben-Haim 2001).

Learning and active adaptive management

Merely maximizing some biodiversity objective ignores one final issue – learning. Given the uncertainty about many parameters and processes embedded in the problem formulation, there must be merit in reducing some of that uncertainty through learning. If the problem is properly formulated we can simultaneously

learn and act to achieve optimal outcomes. This is the concept of active adaptive management, a process for learning while doing.

Active adaptive management contrasts with passive adaptive management, where learning occurs serendipitously (Shea *et al.* 1998). While the concept of active adaptive management has been around for a long time (Walters 1986), the idea is rarely applied to conservation problems (Gerber *et al.* 2005). Active adaptive management is further complicated by cultural and social issues surrounding the experimental management of national parks and state forests. Specifically the approach embraces uncertainty, which does not suit people, politicians and agencies, who may prefer a clear and invariant plan. A great deal more research is required in this area, in particular in the development of robust rules-of-thumb that allow managers to allocate sufficient effort to monitoring outcomes to enable learning.

Final words

It is a truism to say that you cannot simultaneously maximize two different objectives. Win–win decision-making is wonderful but rare. However, a decision-theory approach, with explicit costs and objectives, allows us to examine the trade-offs between alternative currencies and find transparent compromise solutions.

Land managers and management agencies will usually consider other issues in addition to economics and biodiversity. For example, regions interested in riparian restoration will consider the impact of actions, such as fencing, habitat restoration and altering agricultural practices, on the amount of nutrients and sediments flowing into streams, estuaries and water storage systems. Biodiversity itself, being such a broad concept, implies multiple objectives that cannot be amalgamated into a single objective function. So far we have focused on the case of minimizing the expected loss of species. Other biodiversity objectives can include: (i) maximize the opportunity for the continued evolution of existing species and the evolution of new species; (ii) maximize the delivery of ecosystem services, such as salinity mitigation and carbon sequestration; and (iii) maximize the number of different ecological communities that are 'viable'.

While constructing better spatially explicit process models of ecosystems will challenge us technically for a good time, considering multiple criteria concurrently requires even more careful thought about objectives and constraints in the problem formulation. Where there are several objectives that cannot easily be turned into the same currency (e.g. soil loss, nutrient pollution and biodiversity) there are a variety of well-known decision-making methods, like multicriteria decision analysis, that can provide objective and repeatable advice to decision-makers (Drechsler 2004; Mendoza & Martins 2006). Researchers have already produced software for supporting such decision-making (e.g. Janssen *et al.* 2005; Hajkowicz 2006). The situation is further confused by the fact that the objectives interact (e.g. some ecosystem services, like pollination, may be part and parcel of ensuring species' persistence and future evolutionary potential). Developing sensible, reliably quantifiable and more sophisticated biodiversity objectives is a challenging topic for future research. The associated socioeconomic problem is equally intricate. If we choose to include a complex suite of objectives, such as species' persistence, ecosystem services and landscape aesthetics, we need to be able to construct an appropriately weighted objective function – for example, how much is the persistence of the orange-bellied parrot for another year worth relative to more natural flow regimes in a river?

At this point the astute reader might ask the question – surely there is a great deal of subjectivity about how problems are formulated? Does this mean that our approach is just as arbitrary as the scoring and rule-based methods we have criticized so vehemently earlier in this chapter? It is true that problem formulation is a complex social construct. By necessity it is a translation of the aspirations of, and constraints on, society into a suite of mathematical equations. Once done it is transparent. It is not modelling – it is translation (like translating English into French). By using a formal decision-theory approach we can test the sensitivities of alternative goals and measures of the benefits of management actions; this is impossible when there is no explicit statement of the goals. Our challenge is to translate into mathematics as best we can our hopes, dreams and desires for landscapes that are functional and biodiverse. There are innumerable papers that fail to achieve this because they simply don't bother to pose their problem properly using objectives, constraints, decision variables and state dynamics.

Acknowledgements

The chapter contains ideas and labour shamelessly stolen from innumerable colleagues in the spatial ecology lab (http://www.uq.edu.au/spatialecology/) and elsewhere. We also thank Peter Baxter and David Lindenmayer for their valuable comments on earlier drafts.

Seven principles for managing and designing landscapes for biodiversity conservation

1. To inform directly landscape design one must use a decision-theory approach with clear and quantifiable goals.
2. Ecological principles inform practice but do not alone determine practice.
3. In landscape design the whole is more than the sum of the parts.
4. Landscape design problems should accommodate financial costs of different actions.
5. Solving the landscape design problem, once properly posed, requires decision support tools that are readily available from economists, mathematicians and engineers.
6. Uncertainty about parameters and processes is no excuse for not using formal decision-making tools and concepts.
7. Active adaptive management is management with a plan for learning, integrating research with management and monitoring, and enabling us to learn from our successes and failures.

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