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4. Strategies to conserve biodiversity

Stephen Polasky

1 INTRODUCTION

For many biologists the loss of biodiversity is perhaps the single most important environmental issue at the beginning of the 21st century (see for example, Myers, 1979; Wilson, 1992, 2002; Levin, 1999; Pimm, 2001; McKee, 2003). A 1998 survey of 400 biological scientists found that 'the rapid disappearance of species was ranked as one of the planet's gravest environmental worries, surpassing pollution, global warming . . .' and other threats (Warrick, 1998). Several studies estimate that current rates of extinction are several orders of magnitude above the average extinction rate through geologic time (Lawton and May, 1995; NRC, 1995; Pimm et al., 1995). Loss of tropical forests where a large portion of global biodiversity resides is of particular concern.

What is known about present and past rates of extinction, and estimates of future extinction rates, however, is far from conclusive. Projections of how many species are likely to go extinct over the coming century often use species-area curve relationships, which predict the total number of species as a function of the total size of the area that species could inhabit (MacArthur and Wilson, 1967). Predictions of large-scale extinction come from combining the species-area curve relationship with projections of current and future habitat loss. Using this approach Wilson (1999) estimated that 27 000 species a year are likely to go extinct. Skeptics claim that biologists have vastly overestimated the loss of biodiversity (e.g., Lomborg, 2001). Fewer than 1000 species have been documented as having gone extinct since 1500 (IUCN, 2003). Estimating extinction rates is complicated by the fact that we don't really know how many species exist and often have little record of their passing. There are approximately 1.7 million named species while most estimates of the total number of species on earth are in the range from 5 to 15 million (May, 1988, 1990; Wilson, 1988; Humphries et al., 1995). Like other large environmental changes (e.g., climate change), just how much damage human actions have already caused, or will cause if current trends continue, will not be known until the process is already well along. Of course, by then the die may be cast and the loss of a large amount of biodiversity may be irreversible.

Though species numbers and extinction rates grab much of the attention, biodiversity is a much broader concept than just the number of species and the loss of biodiversity is more than simply the number of extinctions. As Noss (1990) states: 'Conservation biologists now recognize the biodiversity issue as involving more than just species diversity or endangered species. The issue is grounded in concern about biological impoverishment at multiple levels of organization.' Biodiversity conservation involves everything from conserving genetic variability within a population, to different populations within a species, to assemblages of species within ecosystems, to ecosystem processes, and a diverse array of ecosystems.

A number of arguments for the importance of conserving biodiversity have been advanced. Many species generate direct use benefits to humans for food, clothing, pharmaceuticals and other products. Non-consumptive use values, such as wildlife viewing and ecotourism, are also significant. At higher levels of organization, ecosystems perform a number of valuable services including nutrient cycling, waste recycling, water purification, and climate regulation (Daily, 1997). Option values for conserving species exist even for species with no known current use value (Fisher and Hanemann, 1986). For example, new pharmaceuticals or other products may be found through bioprospecting (Principle, 1989; Weitzman, 1992; Polasky et al., 1993; Polasky and Solow, 1995; Simpson et al., 1996; Rausser and Small, 2000). Further, species or other components of biodiversity may have existence value (Brookshire et al., 1983; Bishop and Welsh, 1992) or intrinsic value as distinct from instrumental/utilitarian value (Norton, 1987; Ehrenfeld, 1988; Rolston, 1994).

Threats to biodiversity arise from a wide range of human actions. At the top of the list of threats is habitat loss and fragmentation (Wilson, 2002; Wilcove et al., 1998). Estimates of species loss derived from species-area relationships noted above are driven solely by habitat loss. Invasive species are another major threat, particularly on island ecosystems where native species are not well adapted to compete with newcomers. Humans also overharvest some species. There is evidence of large-scale declines in many fish species and changes in the composition of some marine ecosystems due to overfishing (e.g., Jackson et al., 2001; Myers and Worm, 2003). Other threats include pollution, particularly water pollution in aquatic ecosystems, and climate change (Parmesan and Yohe, 2003; Root et al., 2003; Pounds and Puschendorf, 2004; Thomas et al., 2004). The effects of climate change are particularly important in conjunction with habitat loss and fragmentation that may prevent the movement of species to new potentially suitable habitat as climate changes.

Conserving biodiversity will require reducing the threats from human activity. Restrictions on land use, introductions of non-native species, harvesting, pollution and emissions of greenhouse gases, all may impose substantial costs on at least some portion of society. As in other areas of environmental economics, an evaluation of conservation strategies requires assessments of difficult tradeoffs about whether the benefits of conservation exceed the costs. For biodiversity conservation, obtaining quantitative estimates of the value of conservation is particularly problematic. What is the option value for preserving genetic material? What is the existence value of preserving an endangered species? How valuable are various ecosystems services? Though some useful evidence exists, we are far from definitive answers on these and scores of other questions related to the value of biodiversity conservation.

Rather than attempting to compare the costs and benefits of conserving biodiversity, this chapter focuses on the analysis of the efficiency of conservation strategies. What conservation strategy will obtain a conservation objective, such as maximizing the number of species conserved, at least cost? Or, equivalently, what conservation strategy will maximize a conservation objective given limited resources? Such analysis is important for comparing the cost-effectiveness of alternative strategies while largely sidestepping the difficult issue of the valuation of biodiversity.

There is a direct analogy here with environmental policy that affects human health. A full cost-benefit analysis requires estimates of the value of human life, which is a controversial and unsettled area (much like the valuation of biodiversity). Even without an answer to the value of human life though, cost-effectiveness analysis shows where additional expenditures can have the greatest impact in terms of lives saved per dollar. Such analysis can be quite useful for making intelligent policy decisions. Of course, at some point, the difficult question of whether further expenditures are justified requires judgments about acceptable tradeoffs between expenditures and lives saved (or biodiversity conserved).

In the next section, two general approaches to cost-effectiveness analysis are described, one by Weitzman (1998) that focuses on a species-by-species approach, and one by Polasky and Solow (1999) that allows for analysis of a habitat approach. In discussing the details of cost-effective conservation strategy, it is useful to break the analysis into two component parts. The first part involves an analysis of efficient conservation plans assuming a planner with access to all available information and the ability to fully implement plans. Literature on efficient conservation planning is reviewed in Section 3. The second part involves an analysis of decentralized decisionmaking, asymmetric information, and other implementation issues, which may preclude attaining an efficient outcome. Implementation and policy issues are discussed in Section 4. The analysis in Sections 3 and 4 will focus on habitat conservation strategies because habitat loss is the dominant threat to biodiversity conservation and because issues related to habitat loss involves novel analyses that are not commonplace in economics. On the other hand, analysis of harvesting and pollution issues is very well developed within resource and environmental economics with little new to add here. Though interest has increased dramatically over the past couple of years, there is not yet an extensive literature within economics on invasive species (see Polasky et al., 2005a for a recent review). For the sake of brevity and focus, only terrestrial conservation will be discussed in this chapter. The rapidly growing literature on conservation in marine systems and marine reserves will not be covered here (see Botsford et al., 2001; Sanchirico and Wilen, 2001, 2002; Polasky et al., 2005a; and various articles in a special issue of *Ecological Applications* in February 2003 for discussion of marine conservation issues).

2 GENERAL FRAMEWORKS FOR COST-EFFECTIVE CONSERVATION

'The preservation of biodiversity is plagued by the absence of a workable cost-effectiveness framework . . .' (Weitzman, 1998, p. 1279)

Public and private groups with a mandate to conserve biodiversity have limited resources relative to what is necessary to accomplish the task. These groups must set priorities and make difficult choices. In essence, conservation groups face the classic economic problem of allocating scarce resources. For this reason economic analysis should play a more active role in biodiversity conservation. This section reviews two cost-effectiveness frameworks with which to analyse biodiversity conservation plans: Weitzman's 'Noah's Ark Problem,' and Polasky and Solow's 'Conservation with Scarce Resources'.

A. Weitzman: 'The Noah's Ark Problem'

In 'The Noah's Ark Problem,' Weitzman sets out to define a cost-effective approach to conservation that is both intuitive and rigorously derived from basic principles. Weitzman models a resource constrained Noah who doesn't have an ark big enough to fit all species. The conservation problem for the space constrained Noah is to choose species survival probabilities to maximize expected utility from species conservation subject to a budget constraint. Weitzman assumes that the cost of increasing survival probabilities is a linear function. He further assumes that utility consists of the direct value from the existence of the species, and the 'distinctiveness' value, which measures the difference between a species and its closest genetic neighbor. The distinctiveness value captures the degree to which each species adds unique genetic information to the set of surviving species.

Given this setup, Weitzman proves that the optimal conservation policy is an 'extreme policy' in which each species is either conserved to the maximum degree possible or not conserved at all (with the possible exception of a single fractionally conserved species). This result occurs because the objective function is convex and the constraint is linear in probabilities. The expected direct value from existence is simply the sum of direct utility value for each species times its survival probability, which is linear in probabilities. The distinctiveness value is convex in probabilities. The intuition for this result can be most easily seen from an example with two related species. Suppose the distinctiveness value of conserving only one of two species is the same regardless of which of the species is conserved, $V = V(1) = V(2)$. Let the distinctiveness value of conserving both species be $V(1,2)$. Because of the relatedness of the species, adding the second species does not increase the distinctiveness value by as much as adding the first species so that: $V(1,2) < V(1) + V(2) = 2V$. Now consider a problem in which the survival probabilities cannot exceed one: $P_1 + P_2 \leq 1$. Assuming the constraint is binding, with $P_1 = p$ and $P_2 = 1 - p$, the expected distinctiveness payoff is equal to:

$$\begin{aligned} &P_1P_2V(1,2) + P_1(1-P_2)V(1) + (1-P_1)P_2V(2) \\ &= p(1-p)V(1,2) + p^2V + (1-p)^2V \\ &= p(1-p)[V(1,2) - 2V] + V \end{aligned}$$

Since $V(1,2) - 2V < 0$, this expression is maximized by setting $p = 0$ or $p = 1$. In words, it is better to conserve one species for sure rather than having a chance of conserving both species with an equal chance of conserving no species. Compared to conserving a single species, the loss from having no species exceeds the gain from adding a second species.

Under Weitzman's approach, which species should be conserved is determined by a simple ranking criterion:

$$R_i = (D_i + U_i)\frac{\Delta P_i}{C_i}$$

where R_i is the ranking criterion score for species i, D_i is the distinctiveness value added by species i, U_i is the direct existence value of species i, ΔP_i is the change in the probability of survival of species i, and C_i is the cost of increasing the survival probability of species i. The conservation budget should be allocated to conserve the highest ranking species.

Weitzman's conservation allocation rule is exceedingly simple and intuitive. If the real conservation problem were the same as the constrained 'Noah's Ark Problem' there would be little left to discuss, with the exception of how large a budget should be given to Noah. There are at least two reasons that real conservation problems are not the same as the 'Noah's Ark Problem'. First and foremost, conserving species typically requires conserving the habitat of the species. Many species live in the same habitat so that one cannot isolate the effect of a strategy on the survival probability of a single species. In economic terms, habitats exhibit 'joint production' providing increased survival probabilities for a number of species simultaneously. Second, the costs of increasing survival probabilities are unlikely to be linear. It is reasonable to expect that there may be some critical range size for which the marginal change in species survival probabilities is high per unit of additional area, and beyond which the marginal change falls.

B. Polasky and Solow: 'Conserving Biological Diversity with Scarce Resources'

Polasky and Solow describe a simple general approach to conserving biodiversity under a budget constraint. The conservation problem is to choose affordable conservation strategy s from the set of potential conservation strategies S to maximize expected biodiversity conserved:

$$\text{Max} \sum_{x \in X} D(x) P_s(x)$$
$$s.t.\ C(s) \leq B$$

where x is a particular outcome from the set of possible outcomes, X, $D(x)$ is the biodiversity measure of outcome x, $P_s(x)$ is the probability of outcome x under conservation strategy s, $C(s)$ is the cost of strategy s, and B is the size of the conservation budget. Formally, this problem is virtually the same as maximizing expected utility under a budget constraint, with $D(x)$ playing the role of the utility function.

What makes this problem different from a standard economic expected utility maximization problem comes from defining the measure of biodiversity, $D(x)$, and translating a conservation strategy into a probability distribution over possible biological outcomes, $P_s(x)$. There are a range of potential biodiversity measures that could be used, including species richness (the number of species), measures of species diversity that includes a premium for taxonomic distinctiveness (Vane-Wright et al., 1991; Faith, 1992, 1994; Weitzman, 1992; Solow et al., 1993; Solow and Polasky, 1994), measures of relative abundances of species (see Magurran, 1988, 2004), or

measures of ecosystem properties such as productivity (Naeem et al., 1994, 1995; Tilman et al., 1996; Hector et al., 1999), stability (Tilman and Downing, 1994; McGrady-Steed et al., 1997; Naaem and Li, 1997), resilience (Hollings, 1973; Perrings et al., 1995; Carpenter et al., 1999; Walker et al., 1999; Scheffer et al., 2001) or the value of ecosystem services (Costanza et al., 1997; Daily, 1997; Daily et al., 2000).

There are also different approaches for determining how conservation strategies impact on the probability of various potential outcomes occurring depending in part on what objective is used for $D(x)$. Understanding the likelihood of outcome x occurring under strategy s, $P_s(x)$, requires understanding biological cause-and-effect relationships in relation to management actions. In other words, it requires integrating biological knowledge into an economic decisionmaking framework. There is a long tradition of this kind of integration in bioeconomic models of optimal harvesting (e.g., Clark, 1990) but this integration is in its infancy for modeling conservation of habitat or invasive species controls. Biologists are not often used to thinking in terms of marginal analysis useful in analysing policy alternatives. Economists often ignore or simplify biological relationships in economic models. Integration of biological relationships with decision-making approaches from economics will ultimately make such approaches of greater value.

Integrated bioeconomic models of conservation strategy may turn out to be a rich avenue of research, but such models are unlikely to give neat analytic solutions as in Weitzman (1998). Even when a measure of species diversity is used, as in Weitzman (1998), the conservation strategy is rarely specific to a single species. As stressed in the introduction, the key threats to species conservation are loss of habitat and invasive species. Conserving habitat, or protecting an ecosystem from invasion, typically provides protection for multiple species within an ecosystem (joint production). Joint production along with spatial and dynamic relationships make conservation problems complex. While the Noah's Ark rule works for a space constrained Noah, modern Noahs that seek to conserve biodiversity through habitat protection or invasive species control have a tougher challenge in determining cost-effective conservation strategies. This challenge is taken up in the next section.

3 COST-EFFECTIVE CONSERVATION STRATEGIES

This section reviews work from conservation biology and economics on strategies to conserve biodiversity via habitat preservation and control of invasive species. As compared to the general frameworks of Section 2, this section is more specific about the details of conservation strategies. Of the two issues, habitat conservation and invasive species control, far more literature to date has been directed toward habitat conservation. The bulk of this section will review habitat conservation strategies. The small literature on controlling invasive species, which has been starting to grow more rapidly within the past couple of years, will be covered at the end of the section.

A. Habitat Conservation: the Reserve Site Selection Problem

The widespread conversion of land to human dominated uses in many areas has left small fragmented islands of more natural habitat capable of supporting a wide range of biodiversity. Vast areas of temperate forests and grasslands have been converted to farmlands, pasture and urban development. While some temperate areas have shown slower rates of conversion in recent years or even regrowth of forests, habitat destruction in tropical developing countries has continued unabated. In response to the loss of habitat, conservation biologists have proposed establishing a system of formal protected areas to preserve key remnants of remaining natural habitat. It is estimated that protected areas now cover 11.5 per cent of land globally (Chape et al., 2003). Yet, recent studies show that current protected areas are inadequate to conserve all of biodiversity (Rodrigues et al., 2004). With an expanding human population and unmet human needs, particularly in tropical developing countries, there will be limits on how much land will be devoted to meet conservation objectives.

What lands should be set aside as nature reserves to conserve biodiversity given the other pressing demands on land use is a classic economic problem. In fact, this is an excellent problem with which to explain basic economic concepts such as opportunity cost or the optimal allocating of scarce resources under a budget constraint to biologists who may be unfamiliar (or skeptical) about the relevance of economic tools. There is a large literature, written mostly by conservation biologists though economists have become more active in recent years, on setting priorities for habitat conservation.

A simple yet instructive approach to systematic conservation planning is the 'reserve site selection' problem. In the standard formulation of this problem, a conservation planner chooses sites to include in a conservation reserve network to represent the maximal number of species within the reserve network, subject to a constraint on the total number of sites that can be included:

$$\text{Max} \sum_{i=1}^{m} y_i \tag{1}$$

$$s.t. \sum_{j \in N_i} x_j \geq y_i \tag{2}$$

$$\sum_{j=1}^{n} x_j \leq k, \tag{3}$$

where y_i is an indicator variable for species i survival ($y_i = 1$ if species i survives and $y_i = 0$ as species i goes extinct) for all $i \in I$, where I is the set of all species; x_j as an indicator variable for whether site j is selected ($x_j = 1$ if site j is selected in the reserve network and $x_j = 0$ if site j is not selected) for all $j \in J$, where J is the set of all potential reserve sites; N_i is the set of sites in which species i occurs; and k is the number of sites that may be included in the reserve network. This is an integer programming problem that is called the 'maximal coverage problem' in operations research (Church and Revelle, 1974; Underhill, 1994; Camm et al., 1996). Even reasonably large-sized problems with hundreds of sites can be solved using branch-and-bound algorithms (Church et al., 1996; Csuti et al., 1997; Pressey et al., 1997; Ando et al., 1998).

In some respects, the reserve site selection problem resembles the Noah's Ark problem, with the collection of reserves sites constituting the conservation network playing the role of the ark. There is, however, one important difference between the two problems. In the Noah's Ark problem each species is chosen individually. In the reserve site selection problem, sites that contain numerous species are chosen. Because of the potentially complicated pattern of overlap in species there isn't a simple method for finding an optimal solution. In choosing sites, complementary sites that add different species than are represented in other selected sites are more valuable than sites with higher species richness but more overlap (Pressey et al., 1993).

By solving the reserve site selection problem for different levels of the constraint on the number of sites that can be included in the reserve network one can trace out an accumulation curve showing the number of species that can be represented for various sized conservation networks. Csuti et al. (1997) solved for an accumulation curve for terrestrial vertebrates in Oregon. The accumulation curve is initially quite steep as numerous species co-occur in the same biologically rich sites, but declines quickly after the first few sites. Over 90 per cent of all terrestrial vertebrate species in Oregon were included in a reserve network of five sites and over 95 per cent were included in ten sites. In total 23 sites were needed to include all species, with the last sites adding only one or two species as a time.

The reserve site selection problem can be translated into a budget constrained rather than site constrained problem by incorporating the cost of including site j in the reserve network. Doing so, the constraint in equation (3) can be rewritten as: $\sum c_j x_j \leq B$, where c_j is the cost of including site j in the reserve network and B is the total conservation budget (Ando et al., 1998; Polasky et al., 2001). In this case, the accumulation curve becomes a cost curve showing the minimum total cost necessary to represent a given number of species in the network. Ando et al. (1998) used data on average value of agricultural land and endangered species by county to solve for both a budget constrained and a site constrained reserve network. By choosing sites that have a high species represented per dollar ratio rather than the biologically rich sites, the budget constrained solution resulted in the same number of endangered species represented in selected sites at one-third to one-half the cost of the site constrained approach. Under the budget constrained approach, sites with lower land costs in the interior mountain states are included more often whereas the site constrained solution includes more sites with high land costs such as coastal Southern California. Both Ando et al. (1998) and Polasky et al. (2001) found low costs for conserving the majority of species with steeply rising costs as solutions approach complete representation.

At a global scale, Balmford et al. (2003) used cost-effectiveness analysis to find that large efficiency gains could be made by redistributing conservation efforts toward tropical developing countries where the costs of protection are low and the benefits of protection are high. They found a 'gross mismatch' between beneficial conservation projects that are concentrated in tropical developing countries and current conservation spending that is heavily skewed toward temperate developed countries. They find that cost differences among sites range over several orders of magnitude, which is greater than the variance of biological benefits. Balmford et al. (2003) conclude that failure to incorporate costs into conservation planning will result in missed opportunities for greater conservation efficiency. This study highlights important differences in costs and benefits of conservation across countries. However, the study was limited in the type of cost data it used, focusing on management costs while not incorporating land purchase or opportunity costs that are likely to make up a large fraction of conservation costs. More work analysing cost-effective conservation strategies at an international scale would be highly beneficial.

In almost all applications, information about costs or benefits of conservation is incomplete. Several studies have analysed reserve site selection when information about species ranges is incomplete. Polasky et al. (2000) use heuristic methods to maximize expected species representation in a reserve network given only probabilistic information about species ranges. Camm et al. (2002) solved the expected species representation problem using linear approximations to achieve a solution arbitrarily close to the optimal solution using linear programming techniques. Overall, solutions to expected species representation problem tend to be similar to solutions assuming presence/absence data. One difference between the approaches, however, is that two nearby sites can be included under the probabilistic approach, where doing so increases the probability of representation for some set of species, but not under the presence/absence approach. Incorporating uncertainty opens several important dimensions for research including species persistence probabilities, threats of habitat conversion and stochastic events, some of which are discussed in the next subsection.

B. Modeling Land Use and Species Persistence: Beyond Reserves and Beyond Representation

Ultimately what is of importance is the long-term survival of biodiversity. Just because species are currently represented inside a reserve network does not necessarily guarantee their persistence over time. Reserve sites may be too small to sustain a viable population. There may be drought, disease or other stochastic events that wipe out local populations. Conditions in reserves may become less hospitable to species over time due to natural succession or climate change. Similarly, species outside of the reserve system will not necessarily perish. Many species can tolerate some level of human activity and habitat disturbance. Some species thrive in human dominated landscapes, though these species are not necessarily species that humans prefer (e.g., rats and pigeons). Further, not all land outside of reserves will be heavily impacted by human activities (at least in the near term). For these reasons, conservation analysis must progress beyond mere representation (as in the reserve site selection approach) and address likely persistence.

Another fact pushing analysis beyond consideration of reserves is the fact that the vast majority of land lies outside of protected areas. For conservation of biodiversity to be successful roughly 90 per cent of land that lies outside of formal protected areas must contribute to conservation goals. Many analysts have pointed out the need to move beyond reserves and analyse the likely conservation outcomes as a function of what is happening on the entire landscape (e.g., Franklin, 1993; Miller, 1996; Reid, 1996; Wear et al., 1996; Daily et al., 2001; Polasky et al., 2005b; Rosenzweig, 2003). Miller (1996, p. 425) summed up a landscape approach as follows: 'biodiversity will be retained to the extent that whole regions are managed cooperatively among protected areas, farmers, foresters and other neighboring land users.'

The most well-developed approach to modeling species persistence is population viability analysis (Soule, 1987; Boyce, 1992; Beissinger and McCullough, 2002). Population viability analysis incorporates demographic, genetic and environmental stochasticity to predict the likelihood of survival of a species with a given static population. This analysis can be combined with a landscape analysis that provides the distribution of habitat. Several economic studies have combined spatially explicit biological models with human land use decisions to find tradeoffs between species persistence and the value of economic production activities. In particular, a number of papers have traced out production possibility frontiers for the value of timber production and species persistence for a single or small number of forest dwelling species (e.g., Montgomery et al., 1994; Haight, 1995; Hof and Bevers, 1998; Marshall et al., 2000; Rohweder et al., 2000; Calkins et al., 2002; Nalle et al., 2004). Montgomery et al. (1994) combined a population biology model for the spotted owl with economic models of the value of timber harvest to estimate a marginal cost curve for increasing owl survival probabilities. The results showed that marginal costs of increasing owl survival were low for survival probabilities below 90 per cent but increased sharply for survival probabilities above 90 per cent.

Land use decisions simultaneously affect a large set of species so conservation planning would ideally move beyond a species-by-species approach. Several papers have expanded the landscape level analysis to include a large number of species (Montgomery et al., 1999; Lichtenstein and Montgomery, 2002; Polasky et al., 2005b). Doing so necessitates a change in approach because detail intensive population biology modeling becomes unwieldy with a large set of species. Montgomery et al. (1999) used the percentage of habitat conserved under various land use decisions in Monroe County, Pennsylvania to construct probabilities of survival for 147 bird species that currently inhabit the county. Polasky et al. (2005b) used a spatially explicit model of the consequences of alternative land use decisions on the persistence of various species and the value of agricultural and forestry production, based on conditions in the Willamette Basin in Oregon. In their analysis of efficient land use patterns, they found that a large fraction of species conservation could be obtained at low cost. For example, they found it was possible to obtain 96 per cent of the maximum species persistence value while also obtaining 93 per cent of the maximum commodity production value. Trying to increase either objective from this point, however, resulted in dramatic reductions in the value of the other objective. In comparison, running a reserve site selection analysis that assumes no biological value for lands outside of reserves shows both lower scores and more continuous tradeoffs between biological and economic objectives.

Virtually all of the landscape analyses done to date have been static. This is perhaps not surprising since these analyses require integrating spatially explicit biological and economics models. Adding dynamics on top of this is a daunting task. However, there are important dynamic elements to

conservation that cannot be ignored. Costello and Polasky (2004) analyse a dynamic reserve site selection problem in which each site currently unprotected has a probability of being developed during that period. A conservation agency would like to protect as many sites as possible as early as possible but faces constraints on when funds are available. The model is solved using stochastic dynamic programming, which limits the size of problems that can be handled. However, a heuristic solution that involves selecting sites that combine high biological value added per unit cost plus face a high development threat performed quite well for a set of small-scale problems (whether this remains true for large-scale problems is unclear). This result provides some support for the 'hotspots strategy' that gives high priority to conserving places of high biodiversity or high endemism and facing immanent threats (e.g., Myers, 1988; Mittermeier et al., 1998; Myers et al., 2000). The hotspot approach was recently criticized by Kareiva and Marvier (2003), though some of the criticisms, such as ignoring cost and issues of complementarity among sites, have been addressed in the literature discussed above.

In general, unforeseen changes originating either from the economic side (development activity or changes in relative prices), from the biological side (biological invasions, disease outbreaks) or from changes in the physical environment (hydrology, climate change), mean that once-and-for-all conservation decisionmaking is inappropriate. Conservation decisionmaking should adapt to changing conditions and be forward looking, trying to anticipate what may lie ahead. This latter point is especially important with irreversible outcomes, such as extinction. The path breaking work by Arrow and Fisher (1974) highlights the value of avoiding irreversible outcomes prior to the resolution of uncertainty, finding that there is an option value to maintaining flexibility. Recent work by economists and ecologists has also emphasized the importance of preserving the resilience of ecosystems and of avoiding potentially damaging and difficult-to-reverse shifts between alternative ecosystem states (e.g., Perrings et al., 1995; Carpenter et al., 1999; Walker et al., 1999; Dasgupta and Mäler, 2003).

Solving for optimal solutions in spatially explicit integrated biological and economic models, with either dynamics or uncertainty, can be quite difficult. While optimal solutions are a useful benchmark they are not a prerequisite for analysis to be useful to decisionmakers. In fact, given how suboptimal most land use and conservation policy is at present, most well-grounded analysis can only help to improve the situation. One quite useful approach is to blend the type of analysis discussed in this section with predictions of likely land use that will result under various policy scenarios. The next section takes up issues concerning policy and implementation issues related to conserving biodiversity.

4 CONSERVATION POLICY AND IMPLEMENTATION ISSUES

The previous section focused on the question of optimal conservation plans. This section focuses on the question of how conservation plans might be implemented. The fundamental problem raised by biodiversity conservation is the mismatch between the scale at which benefits accrue and the scale at which land use decisions are typically made. Conservation often generates widespread benefits including global public goods such as species existence value or carbon sequestration. But most decisions affecting habitat are local land use decisions made by individual landowners, small communities or local governments. Because local decisionmakers do not receive the full benefits of conservation they will not typically have adequate incentives to conserve. Like most issues in environmental economics the question is how to internalize the externalities (in this case the positive externalities from widespread conservation benefits). To some extent, there may be ways that local decisionmakers can capture at least some of the benefits of conservation. The first part of this section addresses the degree to which biodiversity conservation can pay for itself. To the extent that local decisionmakers receive adequate returns from conserving biodiversity, there is no need for explicit conservation policy at a higher scale of governance (national or international). However, to the degree that conservation benefits cannot be internalized directly, there is a role for explicit policies to promote conservation. The second part of the section will look at conservation policies at the national level, in particular the US Endangered Species Act. The final part of this section will look at conservation policies at the international level, in particular the Convention on International Trade in Endangered Species and the Convention on Biological Diversity.

A. Marketing Biodiversity: Can Biodiversity Conservation Pay for Itself?

Though some of the benefits from conserving biodiversity generate global public goods, other conservation benefits generate private goods that may be sold in markets potentially generating returns to local landowners. Given the problems that arise with conservation policy, to be discussed below, making 'conservation pay' thereby generating direct incentives to landowners and other local decisionmakers to conserve is an attractive option (Heal, 2000; Daily and Ellison, 2002; Pagiola et al., 2002). In fact, some conservation biologists and economists have made the point that conservation will likely occur only to the extent that it is used and generates returns, i.e., 'use it or lose it' (Jantzen, 1992; van Kooten and Bulte, 2000).

Perhaps the most straightforward way in which species conservation can generate returns to landowners and local communities is through ecotourism. Nature based tourism is one of the fastest growing segments of the overall tourism industry, which generated estimated revenues of \$463 billion in 2001 (World Tourism Organization, 2002). Areas with unique resources or charismatic megafauna (e.g. Krueger National Park, Serengeti National Park, Yellowstone National Park) have the potential to generate large amounts of revenue. Examples of successful development of nature based tourism include Costa Rica, where about 1 million tourists spent approximately \$1 billion in 2000 (Daily and Ellison, 2002, p. 178), and South Africa, which generated over \$2 billion in nature based tourism revenue in 2000 (World Tourism Organization, 2002). Other examples of ecotourism are analysed and summarized in Maille and Mendelsohn (1993), Alyward et al. (1996), Wunder (2000) and Lindberg (2001).

Ecotourism, however, raises its own set of problems. As ecotourism becomes more successful it brings more tourists and more economic activity to the area, thereby increasing the danger of damaging the very ecosystems that provide the attraction (Liu et al., 2001). A second major issue with ecotourism is that of who captures rents that might be generated from ecotourism. Large revenue figures, as quoted in the previous paragraph, do not say anything about the size of rents created from ecotourism because costs are not included. Presuming there are positive rents once costs are subtracted from revenues, what share of rents goes to international companies, national governments, and local communities? An important issue, in terms of both equity and providing the right set of incentives, is to insure that local communities receive an adequate return from ecotourism. Receiving an adequate return is especially important in cases where wildlife damages crops or livestock. Much of the push for community based conservation was the recognition that local communities will not have enough incentive to conserve unless they are given access to ecotourism revenues or other benefits generated by conservation (Barbier, 1992; Wells and Brandon, 1992; Western and Wright, 1994). Giving local communities more control over resources and a larger slice of the revenue stream, however, requires that national governments and companies cede some control over resources and revenues to local communities. Issues of power and control over resources are major stumbling blocks that often prevent adequate sharing with local communities.

Bioprospecting is another potential way in which biodiversity conservation may generate market rewards that could provide incentives for conservation. Bioprospecting is the systematic search for useful genetic material from plant or animal species for development of valuable pharmaceuticals or other products. An agreement in 1991 between Merck and Costa Rica's Instituto Nacional de Biodiversidad (INBio) provided $1 million to INBio. Despite the initial excitement, bioprospecting has failed to provide much if any spur to conservation. There have been no other major deals signed since the Merck–INBio deal. Questions have been raised as well about how much difference the deal has made in slowing deforestation or other forms of habitat loss in Costa Rica. During the 1990s, Costa Rica had one of the highest rates of deforestation in Latin America with average annual losses of forest area exceeding 3 per cent (World Resources Institute et al., 2000).

Whether there are likely to be significant rents generated by bioprospecting, and how to share these rents are open questions. Simpson et al. (1996) showed that economic returns from bioprospecting are likely to be quite small, far too small to generate adequate incentives for conservation (though a different view is given in Rausser and Small, 2000). If there are significant rents from bioprospecting, it is unclear how those rents should be allocated between local communities and countries in which the biological resources are located and companies or countries supplying intellectual discoveries that turn these biological resources into valuable products. The rent allocation issue has been the subject of a heated debate between developing countries and developed countries (particularly the US). There is a strongly held feeling by some in developing biodiversity-rich countries that the application of the intellectual property rights under the aegis of the World Trade Organization's Agreement on Trade Related Aspects of Intellectual Property Rights (TRIPs) would result in global corporations profiting from biological resources and traditional knowledge without giving local communities their fair share. This has led some to refer to bioprospecting as 'biopiracy'. On the other side, the US failed to ratify the Convention on Biological Diversity largely because it felt there was not adequate protection of intellectual property rights the way the Convention was drafted. In economic terms, if adequate returns are not given to the host community or country supplying the biological resources, there will be insufficient incentives to conserve habitat on the ground. On the other hand, if adequate returns are not given to the company supplying the innovation, there will be insufficient incentives to develop new products. On the latter issue, several analysts have pointed out the often large gap between the private returns to a firm commercializing a new product and the social returns from such an innovation (Mendelsohn and Balick, 1995; Koo and Wright, 1999). Other than vertically integrating the suppliers of biological resources with the suppliers of intellectual resources, there does not appear to be an easy solution to the incentives problem nor to the bitter political dispute over what is a fair sharing of the rents.

B. National Conservation Policy

In some fortuitous circumstances, conservation may generate sufficient returns to private parties to make conservation pay. Biodiversity conservation, however, creates many goods and services that generate benefits that accrue far beyond local decisionmakers, including global public goods. For this reason, there is a clear rationale for conservation policy by governments as well as actions by non-governmental conservation organizations.

At the national level, there are a number of policies related to conservation. One of the most important, and controversial, is the US Endangered Species Act (ESA). The ESA has proved to be a powerful tool for conservationists and a magnet for criticism for groups promoting private property rights and deregulation (Brown and Shogren, 1998). The two central provisions of the ESA are contained in Section 7, which prohibits federal government actions that cause 'jeopardy' (i.e., risk of extinction) to listed species, and Section 9, which prohibits public and private parties from 'taking' listed species. 'Taking' includes causing harm to species, where harm includes adverse habitat modification from otherwise legal land uses. The ESA has caused changes in timber harvesting plans in the Pacific Northwest to protect the spotted owl and the Southeast to protect the red cockaded woodpecker, in residential and commercial development plans in Southern California and elsewhere.

One criticism of the ESA among economists is that it fails to create positive incentives to conserve, and worse, may create perverse incentives that actually result in less protection for listed species (Innes et al., 1998). Because a landowner may face land use restrictions and is not guaranteed compensation for lost value, there is an incentive to prevent listed species from becoming established or preventing their discovery (Mann and Plummer, 1995, Polasky and Doremus, 1998). There also may be incentives to race to develop in order to beat the imposition of the ESA (Innes, 1997). A second criticism of the ESA is that it does not weigh costs and benefits of actions but is an absolute prohibition against harming listed species.

Several initiatives were begun in the Clinton Administration in an attempt to make the ESA more flexible and less onerous on private landowners. Landowners subject to ESA prohibitions were encouraged to file Habitat Conservation Plans (HCP). Landowners with an approved HCP would be guaranteed 'no surprises', so that costs of further prohibitions to protect the listed species covered by the HCP would be the government's responsibility not the landowner's, and 'safe harbors', so that a landowner that improved habitat for a listed species would not be penalized.

More fundamental reforms in conservation policy call for more market based regulatory approaches and more voluntary (less proscriptive) approaches. One such proposal is to institute a system of transferable development rights (Field and Conrad, 1975; Mills, 1980; Panayotou, 1994; Merrifield, 1996; Renard, 1999; Thorsnes and Simons, 1999; Weber and Adamowicz, 2002). Such transferable development rights (TDRs) would operate much like a marketable pollution permits or individual transferable quotas (ITQs) in fisheries management. As with marketable pollution permits and ITQs, TDRs face questions about the efficient number of permits to issue as well as questions about how to allocate the permits. A problem that is arguably more severe with TDRs than with marketable pollution permits or ITQs is figuring out what constitutes an equivalent trade. For conservation purposes, the spatial pattern, extent and quality of habitat matter. Allowing trades on an equal area basis is not likely to be a sensible policy. Instead, trading ratios should be established based on the relative contribution of particular land parcels to conservation goals. However, the contribution of particular parcels is not constant but in general depends upon the overall pattern of land use in a region. How to design a reasonably effective yet workable TDR scheme given land heterogeneity and interdependent values is an open question.

Another approach to conservation is to use direct payments for conservation. Proponents of this approach argue that the direct approach is the most efficient means of promoting conservation (Ferraro and Kiss, 2002; Ferraro and Simpson, 2002). Ferraro and coauthors argue that by directly targeting and paying for conservation, this method can deliver more in terms of conservation per dollar spent than indirect schemes that like integrated conservation and development projects. Direct payment schemes also have the advantage of being voluntary rather than coercive. Costa Rica has instituted a system of payments for ecosystem services: mitigation of greenhouse gas emissions, watershed protection, biodiversity conservation, and scenic beauty. Many developed countries have some form of 'green payments' in their agricultural policies that pay farmers who adopt environmentally friendly management practices or land uses (OECD, 2001). Smith and Shogren (2002) outline a voluntary payments scheme for biodiversity conservation similar to the US Conservation Reserve Program, in which farmers receive payments to retire land from active production. Many local communities in the US have recently passed bond issues to raise money for the purchase of open space. In 2000, 174 out of 210 open space bond issues were passed, raising $7.5 billion for the purchase of open space (The Trust for Public Lands). A number of non-governmental organizations, such as the Nature Conservancy, raise substantial amounts of money used to purchase important habitat for conservation.

Many policies affect conservation outcomes besides those that are expressly about conservation. Any policy that affects land use decisions, including agricultural and forestry policy, the placement of infrastructure and local zoning ordinances potentially has impacts upon conservation. Often these policies promote activity that is harmful to conservation. Coordinating policies and removing subsidies for activities harmful to the conservation of biodiversity remain high on the wish list of reforms by conservationists in most countries.

C. International Conservation Policy

Some of the benefits of biodiversity conservation may spill beyond national boundaries, and indeed may be global in scale, such as existence value for species. Adequately dealing with such benefits requires international policy and international institutions. At the international level, there are two major conventions related to biodiversity conservation, the Convention on International Trade in Endangered Species (CITES) and the Convention on Biological Diversity (CBD), as well as other programs and policies.

CITES has provisions that allow prohibitions on international trade in endangered species (for species listed under its Appendix I) and regulation of international trade (for species listed under its Appendix II). Because it is focused on trade, CITES deals primarily with high-profile species that are harvested either for food, medicines, pets or trophies. CITES does not directly address issues of habitat loss and fragmentation or control of invasive species. The most high-profile and controversial action taken under CITES was the ban on trade in ivory that began in 1989. Prior to the ban, rampant poaching of African elephants had caused an approximate 50 per cent decline in elephant populations (Barbier et al., 1990). Poaching and population decline was a severe problem in East Africa, while Southern African countries (Botswana, Malawi, Namibia, South Africa, Zimbabwe) had relatively healthy elephant populations. The Southern African countries opposed the ban arguing that selling ivory provided a large financial reason for conserving elephants and the resources to prevent poaching. Many economists predicted that the ban would be ineffective, driving trade into the black market where high prices would provide large incentives to keep poaching (Barbier and Swanson, 1990). In practice, the ivory ban has been largely a success story. Poaching of elephants declined and elephant populations in East Africa recovered. The ban appears to have been successful because it acted not only to restrict supply but also to reduce demand. The major demand for ivory is largely for display purposes. Banning ivory caused a large decline in this demand because most would-be consumers do not wish to purchase and display an illegal item. Van Kooten and Bulte (2000) provide a useful summary of economic arguments about the ivory ban.

Trade bans on products from other endangered species have not proven to be as effective as the ivory ban. Black market trade in many items banned under CITES is alive and well (Webster, 1997). The work of Brown and Layton (2001) on the rhinoceros provides one economic argument for why CITES may be ineffective for curtailing poaching of some species. Demand for rhino is fueled by use of ground rhino horn in traditional Asian medicine. Medicinal demand is unlikely to decline even with the imposition of a trade ban. The case of rhino horn is more likely to resemble the case of illegal drugs, which is a large and thriving industry despite being illegal, than it is to resemble the case for ivory.

The Convention on Biological Diversity was one of two high-profile conventions discussed at the Earth Summit in Rio de Janeiro in 1992, the other being the Convention on Climate Change. The goals of the CBD are to conserve biodiversity, sustainably use biodiversity, and equitably share the benefits from use of genetic resources. This last goal has been at the center of debates over sharing rents on bioprospecting as discussed above. The CBD provides guidance to national governments on biodiversity issues, however it is up to national governments themselves to take action. The CBD itself has no enforcement power. Unlike the Climate Change Convention, which spawned a set of ongoing international negotiations to address emissions of greenhouse gases, the CBD has spawned limited interest and little action since its inception.

There is at present a striking negative correlation between the global distribution of economic wealth and the global distribution of biological wealth. As Balmford et al. (2003) point out, most of the cost-effective targets for conservation occur in developing countries while most of the conservation resources are in developed countries. Therefore, an effective and efficient conservation strategy may require large income transfers from economically rich but relatively biodiversity-poor temperate countries to biodiversity-rich but economically poor tropical countries. There are a number of mechanisms under which this might occur. One approach is to arrange for direct conservation payments through bilateral agreements or through multilateral institutions like the Global Environmental Fund. A second approach is to regulate international trade, either through CITES, or through other trade agreements. A third approach is to try to foster international markets for the sustainable use of biodiversity. This can be done either by finding marketable products that rely on the sustainable use of biodiversity (as discussed above in subsection A), or through the creation of environmental markets via TDRs, carbon sequestration credits, or other means. See Barbier (2000) for a summary of international policy approaches to address biodiversity conservation.

5 SUMMARY

Great strides have been made over the past decade in combining economic and biological analysis in integrated models of biodiversity conservation. However, there remain many unanswered questions and much more work remains to be done. For cost-effectiveness analysis, we need better models of how strategies affect long-term persistence of elements of biodiversity and the production of ecosystem services. This will require advances in ecology and economics and the links between the two fields. Existing approaches generally do not do an adequate job of incorporating dynamics and uncertainty. Going beyond cost-effectiveness analysis, economists and others will need to be able provide reliable evidence of the value of conserving biodiversity to be able to justify expenditures on conservation. As mentioned in the introduction, the valuation of biodiversity presents some difficult challenges. Another important area where further work is necessary is how to provide the right set of incentives for conservation to landowners and other decisionmakers. Designing incentive schemes that incorporate heterogeneity and spatial relationships but can be administered in a simple manner can be demanding. Designing and implementing incentive schemes are particularly demanding in developing countries where there is the additional handicap of limited existing institutions. Additional work on conservation policy and implementation issues is particularly important in developing countries, which contain a large share of biodiversity, have rapidly growing populations, and have urgent needs for economic development. Given the rapid pace of change and the enormity of the threats to biodiversity, there is a pressing need for research that can provide insights and information useful for conservation.

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5. Corporate sustainability

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1 INTRODUCTION

This chapter provides an overview of and discussion about current approaches to corporate sustainability. Considerable interest has been generated in the notion of corporate sustainability for a number of reasons. Corporations provide a practical, direct point of contact for the implementation of government policy. Many large corporations control more resources than many sovereign nations. Managements of corporations that seek to gain a competitive advantage are beginning to appreciate the necessity for promoting corporate sustainability initiatives as a way of differentiating themselves from competitors as well as of reducing costs of undertaking business and risks associated with operations. For example, since the concept of ecologically sustainable development appeared (Commission for the Future, 1987) and the related 'precautionary principle' was introduced (Commonwealth of Australia, 1990, p. 9), environmental risk has become a growing concern (Schaltegger et al., 2003, pp. 195–203). There is now greater focus on the management of environmental risks through voluntary means. A further factor promoting corporate interest in sustainability is that when problems occur, such as severe or persistent corporate impacts on the environment, communication with stakeholders is an important way of trying to minimize damage before or after the event. A distinction should be made between sustainability and sustainable development. The former is taken here to represent the goal or end point of the process of sustainable development.

The term 'corporate sustainability' links the general approach to sustainability with sustainability at the corporate level. Section 2 examines the question as to what motivates managers to address sustainability issues within the corporate milieu. Section 3 briefly introduces some of the main aspects of the general concept of sustainability and its links with the corporate level. The fourth section examines the characteristics of and challenges for corporate sustainability in greater depth and considers what is currently included in corporate sustainability. Section 5 discusses the directions corporate