Incorporating the Effects of Socioeconomic Uncertainty into Priority Setting for Conservation Investment

MARISSA F. McBRIDE,*† KERRIE A. WILSON,† MICHAEL BODE,‡ AND HUGH P. POSSINGHAM†

*Environmental Science, School of Botany, University of Melbourne, Melbourne, Victoria 3010, Australia, email m.mcbride@pgrad.unimelb.edu.au

†The Ecology Centre, School of Integrative Biology, The University of Queensland, Brisbane, Queensland 4072, Australia

‡School of Physical Sciences, The University of Queensland, Brisbane, Queensland 4072, Australia

Abstract: *Uncertainty in the implementation and outcomes of conservation actions that is not accounted for leaves conservation plans vulnerable to potential changes in future conditions. We used a decision-theoretic approach to investigate the effects of two types of investment uncertainty on the optimal allocation of global conservation resources for land acquisition in the Mediterranean Basin. We considered uncertainty about (1) whether investment will continue and (2) whether the acquired biodiversity assets are secure, which we termed transaction uncertainty and performance uncertainty, respectively. We also developed and tested the robustness of different rules of thumb for guiding the allocation of conservation resources when these sources of uncertainty exist. In the presence of uncertainty in future investment ability (transaction uncertainty), the optimal strategy was opportunistic, meaning the investment priority should be to act where uncertainty is highest while investment remains possible. When there was a probability that investments would fail (performance uncertainty), the optimal solution became a complex trade-off between the immediate biodiversity benefits of acting in a region and the perceived longevity of the investment. In general, regions were prioritized for investment when they had the greatest performance certainty, even if an alternative region was highly threatened or had higher biodiversity value. The improved performance of rules of thumb when accounting for uncertainty highlights the importance of explicitly incorporating sources of investment uncertainty and evaluating potential conservation investments in the context of their likely long-term success.*

Keywords: biodiversity hotspots, conservation planning, conservation resource allocation, decision theory, investment risk, Mediterranean Basin, optimization, socioeconomic uncertainty, systematic conservation assessment

Incorporación de los Efectos de la Incertidumbre Socioeconómica en la Definición de Prioridades para la Inversión en Conservación

Resumen: *La incertidumbre en la planificación y los resultados de acciones de conservación que no es considerada hace que los planes de conservación sean vulnerables a cambios potenciales en condiciones futuras. Utilizamos un método basado en la teoría de decisiones para investigar los efectos de dos tipos de incertidumbre de inversión sobre la asignación óptima de recursos para la conservación global para la adquisición de tierras en la cuenca del Mediterráneo. Consideramos la incertidumbre sobre (1) sí la inversión continuará y (2) sí los activos de biodiversidad adquiridos son seguros, que denominamos incertidumbre de transacción e incertidumbre de ejecución respectivamente. También desarrollamos y probamos la robustez de diferentes reglas generales para guiar la asignación de recursos cuando existen estas fuentes de incertidumbre. En presencia de incertidumbre en la habilidad de inversión futura (incertidumbre de transacción) la estrategia óptima fue oportunista, lo que significa que la inversión prioritaria sería donde la incertidumbre sea mayor mientras que la inversión permanezca posible. Cuando había probabilidad de que las inversiones fallarían (incertidumbre*

Paper submitted May 10, 2007; revised manuscript accepted July 13, 2007.

DOI: 10.1111/j.1523-1739.2007.00832.x

de ejecución) la solución óptima se volvió una disyuntiva compleja entre los beneficios inmediatos de la biodiversidad por acciones en una región y la longevidad percibida de la inversión. En general, las regiones fueron priorizadas para inversión cunado tenían la mayor incertidumbre de ejecución, aun si una región alternativa estaba muy amenazada o su biodiversidad tenía mayor valor. El mejor funcionamiento de las reglas generales cuando se considera la incertidumbre resalta la importancia de la incorporación explícita de fuentes de incertidumbre de inversión y de la evaluación de las inversiones de conservación potenciales en el contexto de su éxito a largo plazo.

Palabras Clave: asignación de recursos para la conservación, cuenca del Mediterráneo, evaluación sistemática de la conservación, incertidumbre socioeconómica, optimización, planificación de la conservación, riesgo de inversión, sitios de importancia para la biodiversidad, teoría de decisiones

Introduction

Preserving biodiversity in the face of escalating habitat loss has become a race against time (Pimm et al. 1995; Kareiva & Marvier 2003). Limited resources mean that conservation investments must be chosen carefully and their performance evaluated critically (Redford & Taber 2000; Mascia et al. 2003). A combination of biological, economic, social, and political factors should influence the choice of investment, and the realized returns (Brandon & Wells 1992; Balmford et al. 2000; O'Connor et al. 2003, Cowling et al. 2004), although techniques for identifying priority conservation areas using nonbiological factors is currently in its infancy.

Existing global priority-setting approaches (e.g., Stattersfield et al. 1998; Myers et al. 2000; Olson & Dinerstein 2002) identify priority regions predominantly on the basis of measures of regional biodiversity value and in some instances on the basis of vulnerability (i.e., threat) (Brooks et al. 2006). These approaches highlight areas in need of conservation action, but provide limited information as to how resources might best be allocated among regions over time. An alternative approach is to use information on biodiversity values, threats, and costs to determine a conservation investment schedule that maximizes our expected biodiversity returns for a fixed budget (Costello & Polasky 2004; Wilson et al. 2006). This approach, however, does not address sources of uncertainty that have the potential to compromise conservation outcomes. Such sources include political instability and corruption; the impact of natural catastrophes; lack of budget continuity; weak governance; project failure; absence of stakeholder willingness to be involved; failure to effectively empower stakeholders or mainstream conservation priorities; and implementer "burnout" (Barrett et al. 2001; Byron et al. 2001; Smith et al. 2003; Laurance 2004; Winter et al. 2005; Knight et al. 2006).

There are compelling examples of the impact of such uncertainties on conservation management efforts. In countries with high levels of corruption and political instability, a shift in political power may result in substantial policy changes (Fredriksson & Svensson 2003). A key example is the large-scale illegal logging associated with the recent political upheaval in Indonesia (Jepson et al. 2001; McCarthy 2002). This and the catastrophic 1997–1998 El Niño forest fires (Siegert et al. 2001) caused widespread forest loss. Other examples include the exploitation of weak governance and corruption by logging companies across Southeast Asia (Sizer & Plouvier 2000; Sodhi et al. 2004) and the 1998 decision by the Brazilian government to cut 90% of its existing conservation programs in response to a looming economic crisis (Laurance 1999).

Broadly, the impact of these socioeconomic uncertainties on conservation investments can be distilled into two types: (1) the ability to invest and the level of investment in a region are not constant over time (i.e., transaction uncertainty) and (2) investment in a region does not guarantee the long-term persistence of biodiversity (i.e., performance uncertainty). Destabilizing government transitions or domestic conflicts may result in the breakdown of centralized control, which may fuel collusive corruption and result in the overexploitation of resources (Laurance 2004). These in turn may result in reduced government spending on public services and the establishment of projects that allow misappropriation of funds (Smith & Walpole 2005). Consequently, funding levels from national and international sources may be reduced. Such negative impacts of weak or unstable governance on capital investments and natural resource use have been documented and evaluated (e.g., Alston et al. 1996; Bohn & Deacon 2000) and influence the selection of countries for conservation programs, such as "debt-for-nature" swaps, and the success of such programs (Deacon & Murphy 1997). Similar effects can also result from cases of increasingly centralized (and regulated) governments in which corruption is prolific and the potential for misappropriation of conservation funding is high (Gibson 1999; Ferraro 2005). Investment uncertainty may result not only from country-level issues, such as weak governance, domestic conflicts, and corruption, but also from shifting priorities among donors and budget shortfalls.

Although performance uncertainty is also affected by country-level characteristics, it is predominately determined by site-specific factors such as monitoring and enforcement arrangements, resource-use rights, and social structures. Lack of enforcement of protected area status is recognized as a global issue (Peres & Terborgh 1995; Brandon et al. 1998; Bruner et al. 2001). There have been a few empirical and quantitative evaluations of the performance of protected areas and other conservation investments (Ferraro & Pattanayak 2006). Naughton-Treves et al. (2005), for example, found that deforestation rates were 0.1–14% higher outside protected areas than within. In tropical regions, Bruner et al. (2001) used questionnaire data to assess the impacts of anthropogenic threats (clearing, logging, hunting, fire, and grazing) on 93 protected areas in 22 countries and found park effectiveness was correlated with basic management activities such as enforcement, boundary demarcation, and compensation to land holders. Overall, 17% of protected areas had lost native vegetation due to land clearing. Although these empirical assessments focused on tropical regions, the effectiveness of protected areas in other species-rich regions, such as the Mediterranean Basin, has also been questioned (Halkka & Lappalainen 2001).

The Mediterranean Basin contains 20% of the world's floristic diversity in less than 2% of the world's land area and has been identified as a global biodiversity hotspot (Medail & Quezel 1999). Nonetheless, there are significant threats to the biodiversity values of the Mediterranean Basin, including urbanization, desertification, forest loss, overexploitation of resources, and tourism development. Thus, there is a call for the capacity of protected-area institutions in the region to be strengthened and for the development of monitoring and evaluation systems to assess the effectiveness of conservation investments (World Wildlife Fund 2004). As in other biomes, the potential consequences of transaction and performance uncertainty in the region have not been explored previously because of the paucity of quantitative data and the lack of a rigorous framework within which to examine these uncertainties even if such data were available. Given the variation in socioeconomic and political circumstances between the countries making up ecoregions in the Mediterranean Basin, factoring in uncertainty in the implementation and outcomes of investment is likely to be important for improving long-term effectiveness of conservation investments in this region.

We used data that reflect existing political and socioeconomic conditions within the Mediterranean Basin to investigate the impact of transaction and performance uncertainty on the optimal allocation of global conservation resources. We focused on the allocation of global resources, such as from an international nongovernmental organization, that can be allocated to any of the countries within the Mediterranean Basin. A general framework for allocation of conservation resources under investment and performance uncertainty was developed, but for simplicity we applied it here only to land acquisition. We evaluated the impact of transaction uncertainty by incorporating a probability that the ability to invest either ceases or substantially increases. We evaluated the impact of performance uncertainty by incorporating a probability that in each year existing protected areas fail (i.e., their biodiversity assets are left vulnerable to land-degradation processes). For both sources of uncertainty, we assessed the performance of existing resource-allocation approaches and identified robust rules of thumb that may guide the allocation of conservation resources when these sources of uncertainty exist.

Methods

Basic Resource-Allocation Framework

Our conservation resource allocation framework is underpinned by region-specific characteristics (biodiversity value, habitat loss rates, and the cost of conservation actions), a dynamic landscape model, and an annual budget. Each region is composed of a number of homogenous land parcels, each of which can be in one of three states: reserved, available for reservation or development, or lost to development (with associated biodiversity values assumed destroyed). Although we limited our conservation action to the acquisition of land for reserves, we recognize that investment can take many forms, such as the establishment of forestry concessions, covenants on private land, and species recovery programs. Our framework for the allocation of conservation resources accounts for such investments, and although we conducted our analysis at the resolution of entire regions, the approach is amenable to supporting investment decisions at finer scales.

We treated the allocation of conservation resources as an ongoing, dynamic process (Wilson et al. 2006). Funds to acquire parcels were made available annually, and we assumed that surplus funds at the end of the year would be used for other purposes. Parcels that remained available for acquisition at the end of each year had a probability of being destroyed that depended on region-specific rates of habitat loss. This process was stochastic (i.e., the number of parcels lost to development in a region each year were drawn from a binomial distribution). We assumed that for each successive parcel reserved, the increase in number of species reserved diminished in a nonlinear fashion following the species–area relationship (MacArthur & Wilson 1967), so the number of species reserved in region i was represented by

$$\mathrm{SR}_i = \alpha_i \mathrm{AR}_i^{z_i}, \tag{1}$$

where AR_i is the area of land reserved in region i, α_i and z_i are region-specific constants. Here, α_i is calculated for each region:

$$\alpha_i = \frac{\mathrm{ST}_i}{\mathrm{AT}_i^{z_i}}, \tag{2}$$

where ST_i is the total number of species in region i, AT_i is the total area of region i, and z_i is assumed to be 0.2 across all regions (Rosenzweig 1995). We used stochastic dynamic programming (SDP) to determine an optimal resource allocation schedule, which is a state-based backwards-iteration algorithm. For each possible system state, the algorithm determines the optimal solution based on the current state and the expected return, given the likely transition probabilities (Bellman & Kalaba 1965; Clark & Mangel 2000). Stochastic dynamic programming has been used to solve a range of conservation problems (e.g., Milner-Gulland 1997; Richards et al. 1999; Tenhumberg et al. 2004) and for dynamic reserve selection (Costello & Polasky 2004; Meir et al. 2004; Strange et al. 2006). The SDP algorithm ran until a time-independent equilibrium solution was reached. We used the resulting investment strategy to simulate the land-acquisition process until the terminal time T was reached. The terminal time occurred when no parcels remained available for reservation or development.

The need to consider all possible outcomes and intermediary system states limits the use of SDP to areas with only very small numbers of conservation regions. In light of this limitation, we devised heuristic rules that approximated the optimal strategy (Costello & Polasky 2004; Turner & Wilcove 2006; Wilson et al. 2006). One such heuristic allocated funding each year to the regions that would yield the greatest increase in the number of species reserved per unit dollar—that is, where the greatest biodiversity bang per conservation dollar could be achieved. We referred to this as the maximize-gain heuristic, in which the value function $V(t+1, X)$ to be maximized at time t at state X was

$$V(t, X) = \sum_{i=1}^{I} \alpha_i \mathrm{AR}_i(t)^{z_i}. \tag{3}$$

An alternative heuristic, which we referred to as the minimize-loss heuristic, allocated funding to the region that would result in the greatest number of species remaining in undeveloped land (both reserved and available) in the subsequent year—that is, land with the greatest biodiversity value that would otherwise have been lost would be protected. For this heuristic, the value function to be maximized was

$$V(t, X) = \sum_{i=1}^{I} \alpha_i \left(\mathrm{AA}_i(t) + \mathrm{AR}_i(t)\right)^{z_i}, \tag{4}$$

where AA_i is the area of land available in region i.

Incorporating Investment Uncertainty

Although incomplete knowledge about the future may appear to make efficient decision making impossible, we used a method with which one can make optimal decisions under uncertainty. There is however a dearth of publicly available, quantitative, and empirical data that describes the relative successes of past conservation investments (Ervin 2003; Stem et al. 2005). We overcame this by using two alternative approaches. First, we assessed the impact of transaction and performance uncertainty with a range of feasible values for both sources of uncertainty. Second, we parameterized our analyses with empirical data on transaction uncertainty obtained from a cross-national archive of time-series data on domestic conflicts (held by Databanks International, http://www.databanks.sitehosting.net/) and data on performance uncertainty based on estimates derived from the peer-reviewed literature. These data sources are explained in detail later.

TRANSACTION UNCERTAINTY

We explored two alternative expressions of transaction uncertainty to assess the effects of the possibility that resource allocation will either cease or increase. First, we introduced an annual probability that the ability to acquire more parcels might be lost and that all currently available parcels might become unavailable. Likewise, to examine the effects of potential increases in funding or capacity to mitigate parcel loss, we incorporated a probability that each year enough funding might become available to acquire all remaining available parcels. In the model we represented this increase in ability to acquire land as an increase in conservation funding, but from a practical perspective, such increases in the ability to protect land could also be interpreted as a legislative ban on all future habitat destruction, on the basis of the assumption that this provides permanent and ongoing protection (i.e., investment and performance uncertainty are independent). For each of these types of transaction uncertainty, we simulated the land-acquisition process with both the optimal solution and the heuristics. We ran the simulations 10,000 times with an annual budget of US\$1 million and probabilities of funding ceasing (f_c) and increasing (f_i) of 0, 0.001, and 0.01.

Second, we considered the effects of incorporating a less extreme expression of transaction uncertainty by relaxing the assumption that once funding ceased, it could never be reinstated. Instead we allowed for the possibility of variable lengths of funding unavailability, following which conservation investment in a region could recommence. To model this we incorporated an additional binomial probability that represented the probability of funding resuming in a given region for each year that it was unavailable. We used empirical data on the number of years of legislature ineffectiveness since 1980 and assumed that the frequency of an ineffective legislature represented the likelihood of cessation of funding. For countries in the Mediterranean Basin with data available, 3 out of 22 had experienced an almost complete loss of legislature effectiveness since 1980 giving an upper probability that funding would cease within countries of 0.14 (Databanks International, http://www.databanks.sitehosting.net/). For countries where the ability to invest had been lost, the probability that funding resumed following each year that it remained unavailable was approximated with the inverse of the number of years a country remained under ineffective legislature and ranged from 0% for countries such as Jordan, Libya, and Syria up to 33% for Turkey.

PERFORMANCE UNCERTAINTY

To assess the effects of performance uncertainty, we introduced a region-specific probability that each year any currently reserved parcel within a region had a fixed probability of failing. As in the previous scenarios, conversion to an unsuitable state was considered irreversible. The inclusion of a probability that reserves may fail meant that all land (available and reserved) would eventually be made unsuitable for conservation, leaving no species remaining at the terminal time. We therefore altered our objective function to maximize the number of "biodiversity years" (i.e., number of years a given species is present in a reserved parcel) acquired by the terminal time. The eventual loss of all biodiversity resulted from the assumption that no single investment process would be able to secure the complete, ongoing protection of a given species or habitat. Under this framework the "biodiversity years" concept allowed for the assessment of investment returns in terms of expected number of years of protection provided. This is equivalent to optimizing for the number of species reserved when there is no performance uncertainty.

To incorporate performance uncertainty, we modified the value functions so they would be maximized by accounting for region-specific probabilities of reserve failure:

$$V(T_{\mathrm{END}}, X) = \sum_{i=1}^{I} \frac{\alpha_i}{p_i d_i} \mathrm{AR}_i(t)^{z_i}, \tag{5}$$

modified maximize-gain heuristic, and

$$V(T_{\mathrm{END}}, X) = \sum_{i=1}^{I} \frac{\alpha_i}{p_i d_i} (\mathrm{AR}_i(t) + \mathrm{AA}_i(t))^{z_i}, \tag{6}$$

modified minimize-loss heuristic, where p_i is the probability of reserve failure in region i, d_i is the rate of habitat loss, and T_{END} is the terminal time at which no species remain. A low probability of reserve failure therefore resulted in greater overall expected returns than when there was a high probability of reserve failure. For example, if a parcel contained just one species, then reserving that parcel when there was a reserve failure probability of 0.1 would yield an average return of 10 biodiversity years, whereas if the probability of reserve failure was 0.01, then the expected return would be 100 biodiversity years. In addition to the two modified heuristics, we also evaluated an alternative, myopic (looks only one step ahead) version of the SDP algorithm. This heuristic extended the original minimize-loss heuristic to account for reserve failure probabilities, but directed funds to the region that would yield the greatest expected increase in biodiversity years in the following year rather than at the terminal time.

We evaluated the unmodified heuristics and the heuristics that incorporated performance uncertainty relative to the optimal SDP solution for two regions. In doing so, we evaluated the performance of the heuristics over a range of combinations of biodiversity value and rates of habitat loss, which were informed by the relative range of these measures from ecoregions within the Mediterranean Basin. In each case, region 1 was assigned higher or equal levels of performance uncertainty than region 2 and lower or equal levels of biodiversity and habitat loss. We ran 10,000 simulations with 30 parcels of land in each region and an available budget that allowed one parcel to be acquired each year. Contour plots were generated to present the variation from the optimal SDP solution in terms of average percentage of biodiversity years acquired across the range of data values explored.

We considered a one to sixfold increase for variation in biodiversity values between regions on the basis of the combined richness of plants and vertebrates within the Mediterranean Basin (assumed to be a surrogate for biodiversity value), which ranges from 668 in the southeastern Iberian shrubs and woodlands ecoregions to 4387 in the southern Anatolian montane conifer and deciduous forests ecoregion (Kier et al. 2005; World Wildlife Fund 2006). Relative differences of between 1 and 10 were used for the predicted rates of habitat loss, which were obtained by predicting changes in human-footprint values on the basis of predicted rates of human population growth. The lowest predicted rate of population growth was 0.002 in the forest ecoregions of northeastern Spain and southern France, and the highest predicted rate of population growth was 0.02 in the eastern Mediterranean conifer-sclerophyllous-broadleaf forests (E. Underwood et al., unpublished data). We did not consider relative differences in costs because the effects scale with the relative effects of biodiversity value and threat (Wilson et al. 2006).

We based our measure of performance uncertainty (i.e., the average annual rate of reserve loss) on the average annual percentage of protected areas experiencing forest loss over 20 years (DeFries et al. 2005). The performance uncertainty levels of DeFries et al. ranged from 0% to approximately 2%, with an average of 1.24%. This study did not extend to the Mediterranean Basin; nevertheless, an assessment of forested protected areas within the Basin suggests that levels of management ineffectiveness in European countries reflect the overall average across non-European regions (World Wildlife Fund 2004).

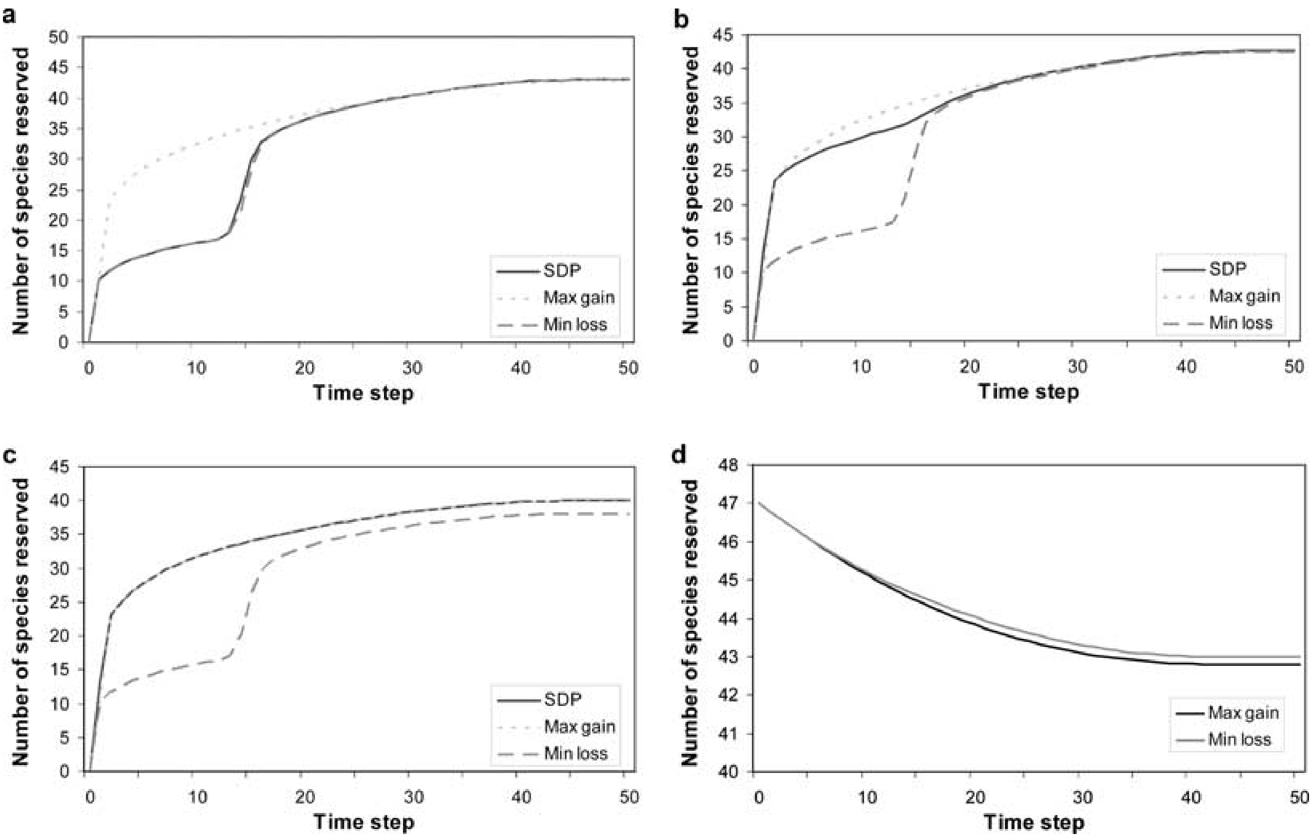

Figure 1. The average total number of species reserved over time across two regions for the optimal resource allocation strategy (SDP) and maximize-gain and minimize-loss heuristics under funding uncertainty when the probability of cessation of funding is (a) 0, (b) 0.001, (c) 0.01. (d) Comparison of the difference in final outcomes for the heuristics at each time point in terms of average number of species reserved if a funding increase occurs and all remaining available land can be reserved. In each case two hypothetical regions with biodiversity levels and rates of habitat loss drawn from typical values of ecoregions in the Mediterranean Basin are evaluated, with 3000 vertebrate and plant species in region 1 and 1500 in region 2, rates of habitat loss each of 1%/year, and equal costs of land acquisition. We assumed that no reserves already existed in the region.

Results

Impact of Transaction Uncertainty on Investment Strategies

When there is no transaction uncertainty, the minimize-loss heuristic outperformed the maximize-gain heuristic. Nevertheless, with an increasing probability that funding would cease, the optimal solution provided an allocation schedule consistent with the maximize-gain approach (Figs. 1a-c). Alternatively, when there was a probability that all remaining available land could be acquired immediately, performance of the heuristics was almost identical. The minimize-loss heuristic was only marginally more effective than the maximize-gain heuristic at any time step during the acquisition process (Fig. 1d). Increases in the number of ecoregions being considered for land acquisition did not alter these overall results.

When the level of transaction uncertainty differed between regions, the optimal strategy was precautionary and opportunistic. Investment was prioritized in the more uncertain region (region 1) when investment was still possible, and the optimal strategy differed from both the maximize-gain and minimize-loss approach (Figs. 2a & 2b). Nevertheless, when differences in transaction uncertainty were less extreme, the relative vulnerability or the biodiversity value of each region remained more influential on the optimal approach. The minimize-loss heuristic again underperformed relative to the maximize-gain heuristic, delivering final outcomes that differed from the optimal solution by up to 20% (Fig. 2c). The performance of the maximize-gain heuristic under uncertainty was in most cases relatively robust. The impacts of transaction uncertainty on the performance of the heuristics were reduced significantly when the loss of funding was not final, and there was a chance that funding would resume at some time in the future (Fig. 2d). The heuristics' performance under such circumstances remained within 5% of the optimal solution at the terminal time.

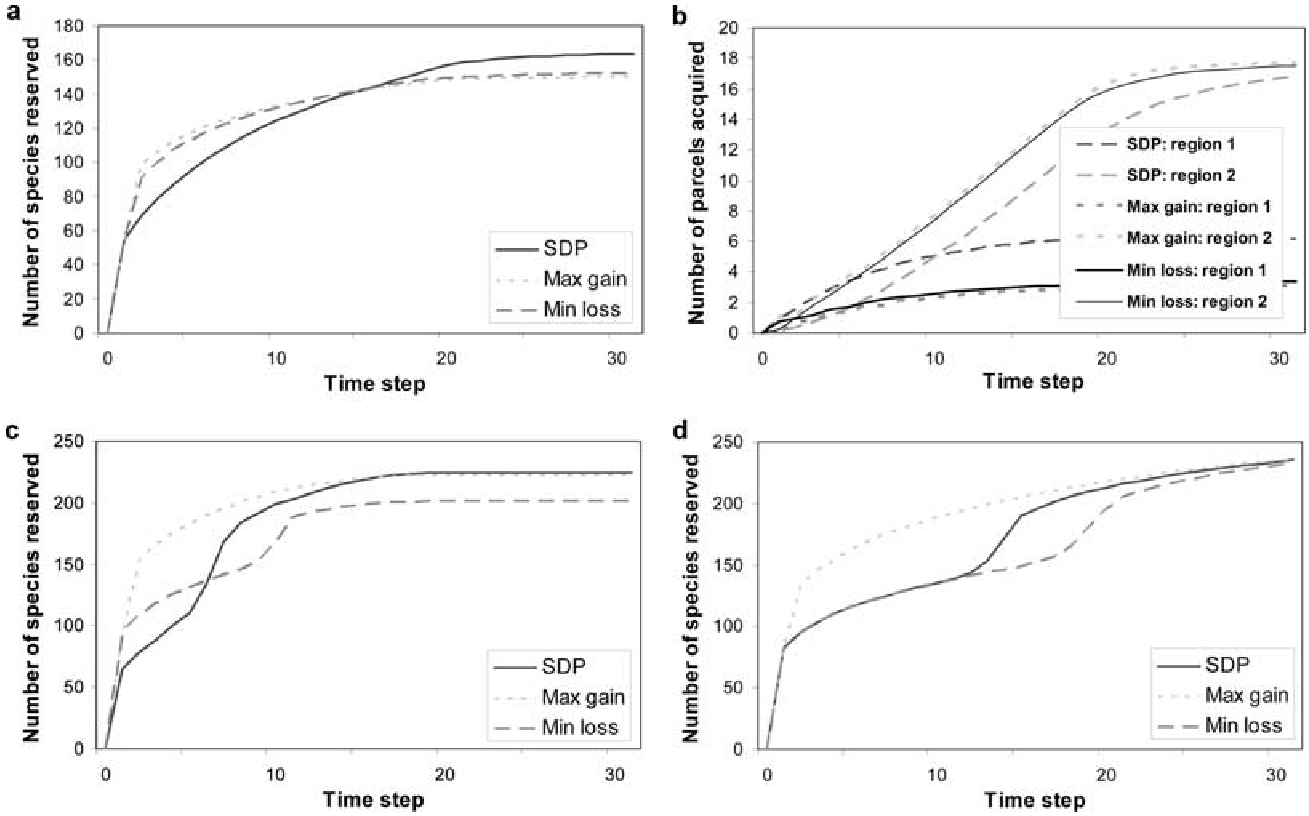

Figure 2. The average total number of species reserved across two regions (panels a, c, d) and number of parcels reserved in each region (panel b) over time for the optimal resource-allocation strategy (SDP) and maximize-gain and minimize-loss heuristics under different scenarios of funding uncertainty. In (a) and (b), the probability that funding will cease is 0.14 in region 1 and 0.0 in region 2, and the probability of funding resuming is 0 for both regions. Biodiversity levels are equal in each region. (c) The probability that funding will cease is 0.05 in region 1 and 0.01 in region 2; the probability of funding resuming is 0 for both regions, and regions 1 and 2 have biodiversity levels of 2000 and 3000 species of vertebrates and plants, respectively. In (d), the probability that funding will cease is 0.05 in region 1 and 0.01 in region 2; the probability of funding resuming is 0.2 in each region; and regions 1 and 2 have biodiversity levels of 2000 and 3000 species of vertebrates and plants, respectively. In each case, the rate of habitat loss is 1%/year in each region and costs of land acquisition are equal. We assumed that no land is initially reserved.

Impact of Performance Uncertainty on Investment Strategies

When performance uncertainty was incorporated, the optimal solution became a complex trade-off between the immediate biodiversity benefits of acting in a region and the perceived longevity of the investment. In general, variation in the probability of reserve failure was more influential in determining the optimal solution than either the regional rate of habitat loss or biodiversity value, and the optimal SDP solution almost always favored the region with the greatest performance certainty, even if the alternative region was highly threatened or had higher biodiversity value (Fig. 3). The unmodified heuristics produced outcomes ranging from 5% to >20% from the optimal solution when the range of biodiversity value and rates of habitat loss exhibited in ecoregions across the Mediterranean Basin were used (Fig. 4). Both heuristics were least effective when large differences in performance uncertainty were coupled with small differences in biodiversity value (Figs. 4a & 4b). The performance of the myopic SDP solution was on average comparable to the two unmodified heuristics and differed from the optimal solution by up to 15% (Fig. 4c).

The performance of the heuristics significantly improved when they were modified to account for region-specific probabilities of reserve failure (Fig. 5). In this case, the heuristic that minimized loss differed from the optimal solution by <4% (Figs. 5a & 5b). The minimize-loss heuristic underperformed when there were high relative differences in performance uncertainty coupled with high relative differences in biodiversity value or rates of habitat loss, which resulted in it directing all investment toward a region that might have high instability. The maximize-gain heuristic differed from the optimal

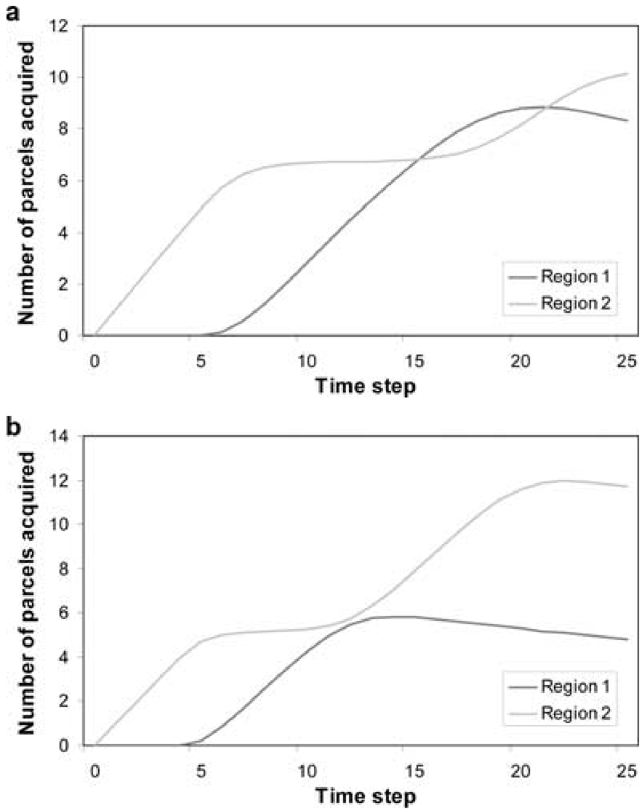

Figure 3. Average number of parcels reserved through time in two regions for two examples of the optimal investment strategy under performance uncertainty. (a) Region 1 has a biodiversity value of 400 and performance uncertainty level of 0.03. Region 2 has a biodiversity value of 100 and performance uncertainty level of 0.01. Rates of habitat loss in both regions are 1%/year. (b) Region 1 has a habitat loss rate of 2%/year and performance uncertainty level of 0.02. Region 2 has a habitat loss rate of 1%/year and performance uncertainty level of 0.01. For biodiversity levels, the number of vertebrate and plant species in each region is 3000. Variation in the probability of reserve failure generally has greater impact on the optimal solution than either regional rates of habitat loss or biodiversity values, with the region with greatest performance certainty favored for investment.

solution by <3% over much of the parameter space, but underperformed when the relative difference in the rate of habitat loss was high compared with the level of performance uncertainty (Figs. 5c & 5d). This was because the maximize-gain heuristic did not account for rates of habitat loss.

Discussion

In societies that too often value short-term profits over long-term gains, it is important to look beyond the quick fix and focus conservation efforts on achieving lasting benefits for biodiversity. Current priority-setting approaches typically neglect the range of economic, political, and social factors that affect the likelihood of investment success. We incorporated aspects of investment uncertainty into an explicitly defined resource allocation framework and formulated resource allocation schedules that will optimize biodiversity outcomes despite this uncertainty. We have assessed two sources of investment uncertainty that are likely to affect the conservation-resource allocation process: transaction uncertainty and performance uncertainty.

Losing the ability to acquire new land in a region (i.e., cessation of funding) had considerable impact on the optimal resource allocation strategy. The optimal strategy changed if there was even a small chance that funding would cease. Under such uncertainty, strategies that maximize short-term gains are the most robust in the long term. For countries within the Mediterranean Basin, where the probability that funding would cease was approximately 14%, an allocation schedule that focuses primarily on maximizing gain is likely to be the optimal approach. In comparison there was little variation in the performance of the heuristics when there was a possibility that future funding levels would increase. In this instance one might expect the minimize-loss heuristic to outperform the maximize-gain heuristic because by abating loss, this heuristic effectively maximizes the amount of land remaining available to be acquired. Nevertheless, diminishing returns and comparatively small rates of loss mean this advantage remains quite small. These results, although intuitive, highlight the importance of considering the sensitivity of funding strategies over time to system dynamics and uncertainties.

When the level of transaction uncertainty differed greatly between regions, the value of an opportunistic approach to conservation investment became evident. The regions with relatively high transaction uncertainty had a higher relative priority for conservation investment because in the future the ability to protect biodiversity in these regions could be lost. Interestingly, the effects of this future uncertainty depended largely on how permanent or irreversible the loss of investment ability was likely to be: the potential for transactions to resume considerably reduced their impact on the optimal investment strategy. Nevertheless, these results must be interpreted within the bounds of the associated assumptions because regions with high levels of transaction uncertainty are often subject to performance uncertainty, and we did not explicitly consider such interactions here. The presence of multiple sources of funding or management within a region may also reduce the impacts of transaction uncertainty. Regardless, our results highlight the pragmatic importance of a precautionary and opportunistic approach (i.e., Noss et al. 2002; Knight & Cowling 2007), which would likely be further emphasized under both sources of uncertainty.

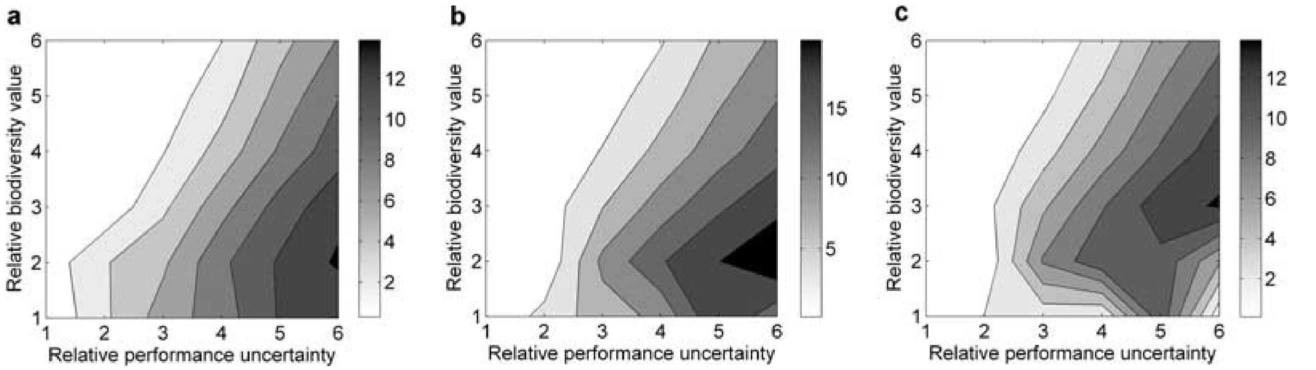

Figure 4. A comparison of the performance of the maximize-gain and minimize-loss heuristics relative to the optimal resource allocation solution (SDP) over a range of biodiversity values and performance uncertainty values: (a) maximize-gain heuristic (for a full definition of the maximize-gain and minimize-loss heuristics refer to the Basic Resource-Allocation Framework section in Methods), (b) minimize-loss heuristic, and (c) the myopic (looks only one step ahead) version of the SDP solution. The relative differences depicted are between regions 1 and 2 and range from equal (at the origin) to sixfold differences in levels of biodiversity and performance uncertainty. Shading represents the relative performance of each heuristic in terms of the percent difference in average number of biodiversity years (number of years a given species is present in a reserved parcel) acquired relative to the optimal (SDP) solution. Performance levels with differing threat and performance uncertainty levels are similar to the differing biodiversity levels and performance uncertainty shown in panels a, b, and c for each of the maximize-gain, minimize-loss, and myopic SDP solutions.

When we included a measure of performance uncertainty—a possibility that investments fail to protect biodiversity assets—our resource-allocation strategies changed drastically. Resources were directed away from regions with high levels of performance uncertainty, particularly during the early years of an investment term. Because it affected the length of time reserves were able to contribute to biodiversity conservation, the level of performance uncertainty was the regional characteristic that had the greatest influence in determining where and when resources were allocated. As a result, both of the unmodified heuristics performed poorly. This is in contrast to previous results in systems without performance uncertainty, where these heuristics differed by less than 3% from the optimal solution (Wilson et al. 2006). The unmodified heuristics should therefore not be applied in situations where the performance of investments is uncertain.

Modifying the heuristics to account for region-specific probabilities of investment failure considerably improved their performance. The modified heuristics approximated within 3% the optimal solution over a wide range of investment scenarios. These findings demonstrated the benefits of explicitly incorporating performance uncertainty into the conservation-resource allocation process. The underperformance of the myopic SDP algorithm further highlighted the importance of making decisions based on long-term expected outcomes, as opposed to decisions that optimize only in the short term.

Our results demonstrated the importance of accounting for the likely success and longevity of conservation investments when prioritizing the allocation of conservation resources at the interregional scale. Our results confirmed commonly held beliefs that, in systems such as the Mediterranean Basin, the exclusion of social and political factors may preclude the efficient allocation of conservation funds. These factors (and their potential influences on conservation outcomes) remain largely neglected within the field of systematic conservation assessment.

Lack of research into the effects of sociopolitical uncertainties on conservation priorities is likely to be driven by the paucity of empirical data on the relationships between the characteristics of conservation investments and their performance outcomes. An important component in applying our conservation-resource allocation framework that is robust to uncertainty is the development of empirical measures of performance uncertainty. This is important for regions outside as well as within the tropics, where the limited data currently available on protected-area performance have been primarily based. The index of performance uncertainty should incorporate information on a range of economic, social, and political factors derived from the perceived impacts of these factors on conservation investments (Robertson & van Schaik 2001; Smith et al. 2003; Cowling et al. 2004; Laurance 2004). Before such an index can be developed, research is required into the complex interdependencies between political and economic institutions and the performance of conservation investments (Barrett et al. 2006). The development of empirical measures of performance and transaction uncertainty across a range of biomes would allow

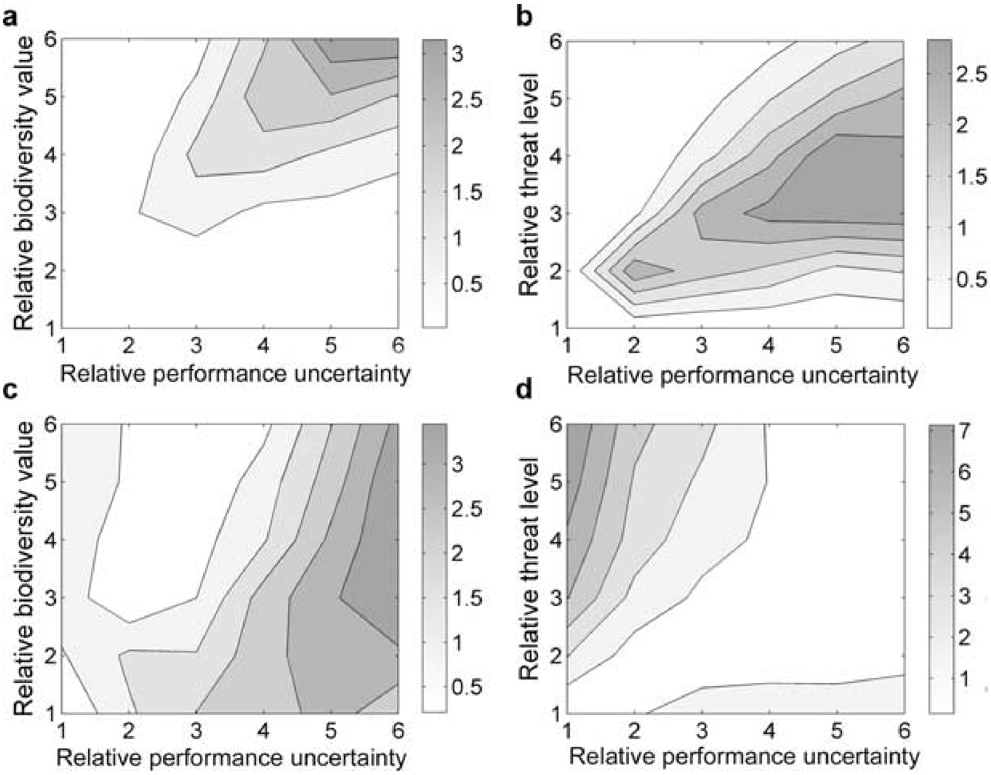

Figure 5. A comparison of the performance of the modified maximize-gain and minimize-loss heuristics relative to the optimal resource allocation solution (SDP) over a range of relative biodiversity value, threat, and performance uncertainty levels (for a full definition of the modified maximize-gain and minimize-loss heuristics refer to the Performance Uncertainty section in Methods). Performance of the modified minimize-loss heuristic over (a) a range of biodiversity value and performance uncertainty levels and (b) a range of threat and performance uncertainty levels. Performance of the modified maximize-gain heuristic over (c) a range of biodiversity value and performance uncertainty levels and (d) a range of threat and performance uncertainty levels. The relative differences depicted are between regions 1 and 2 and range from equal (at the origin) to sixfold differences in levels of either biodiversity or threat and performance uncertainty. Shading represents the relative performance of each heuristic in terms of the percent difference in average number of biodiversity years (number of years a given species is present in a reserved parcel) acquired relative to the optimal (SDP) solution.

the relationship between both forms of uncertainty to be assessed. We anticipate a positive correlation across regions, but exploration of these complex connections will require input from both the social and natural sciences.

Given the influence of performance uncertainty on conservation strategies, it is clear that more proactive approaches are required to secure conservation investments. One course of action is to avoid countries with low management effectiveness, but such a course of action should only be considered after the opportunity costs, in terms of risks to biodiversity, are determined. With such information, the conservation community and the public at large will have the option to carefully evaluate what may be lost if more stable regions are targeted. Another course of action is to facilitate reserve management (Bruner et al. 2001) or the investment of resources in alternative projects, such as policy reforms, education, or community-development projects (Barrett et al. 2001). Without the development of complementary programs for enhancing and empowering conservation practitioners and local land managers, achieving effective conservation outcomes in important areas for biodiversity conservation seems unlikely. Indeed, if the relationship between investment per unit area in either acquisition or management and conservation outcomes can be determined, then a foreseeable extension of this research would be to determine the optimal trade-off between the allocation of resources in acquiring land and improving existing land-management capacity.

We applied our framework for conservation-resource allocation, which is robust to uncertainty at a regional scale. Although we restricted our analysis to global conservation funds, the presence of additional country-based funding could be incorporated by constraining the allocation of a certain proportion of funds to particular ecoregions. With information on factors such as market prices for land, community support, future demand for land use, and landowner's willingness to sell, our framework could also be applied to a systematic conservation assessment at the property scale. This would assist local planners to more directly weigh the risks, opportunity costs, and conservation outcomes associated with investments in particular properties.

Our comprehensive framework for conservation-resource allocation effectively integrates the effects of multiple, region-specific factors: biodiversity value, existing protected area, costs, threats, and investment uncertainty. The framework has four main simplifications (1) spatial heterogeneity within each priority region was ignored; (2) reservation is assumed to be the only possible investment option and once developed, land cannot be restored; (3) regional rates of habitat loss, costs, and levels of investment uncertainty are assumed to be constant through time; and (4) performance and transaction uncertainty are assumed independent. Each of these simplifications can be relaxed within this framework and should be explored. We have recently extended the conservation-resource allocation problem to deal with multiple threats and conservation actions (such as land acquisition, predator control, invasive plant control) (Wilson et al. 2007), and the approaches to dealing with uncertainty are equally applicable in this more realistic problem formulation. To evaluate our rules of thumb against an optimal solution, we considered only three land states (available, reserved, and destroyed), but with confidence in our heuristics for this simple case, one can extend our problem formulation to applications that involve multiple land-use types that provide different contributions to biodiversity conservation. Furthermore, additional dynamics such as changes in land cost with changes in the availability of land can be incorporated (Armsworth et al. 2006).

Our results show that the ability to invest and the performance of investments should significantly alter investment strategies and long-term conservation outcomes. It is clear from these results that the impact of investment uncertainty must be accounted for explicitly when prioritizing the allocation of conservation resources. By shifting the focus to achieving longer-term outcomes and by accounting for investment uncertainty when prioritizing the allocation of conservation funds, the ability to maximize conservation returns from every dollar invested improve substantially.

Acknowledgments

We acknowledge funding from the Australian Research Council and the Commonwealth Environmental Research Facility (AEDA Hub). We thank B. Murdoch for guidance and feedback on this research and A. Knight, C. Rondinini, and B. Pressey for helpful comments on an earlier version of the manuscript.

Literature Cited

Alston, L. J., G. D. Libecap, and R. Schneider. 1996. The determinants and impact of property rights: land titles on the Brazilian frontier. Journal of Law, Economics and Organization **12:**25–61.

Armsworth, P. R., G. C. Daily, P. Kareiva, and J. N. Sanchirico. 2006. Land market feedbacks can undermine biodiversity conservation. Proceedings of the National Academy of Sciences of the United States of America **103:**5403–5408.

Balmford, A., K. J. Gaston, A. S. L. Rodrigues, and A. James. 2000. Integrating costs of conservation into international priority setting. Conservation Biology **14:**1–9.

Barrett, C. B., K. Brandon, C. Gibson, and H. Gjertsen. 2001. Conserving tropical biodiversity amid weak institutions. BioScience **51:**497–502.

Barrett, C. B., C. C. Gibson, B. Hoffman, and M. D. McCubbins? 2006. The complex links between governance and biodiversity. Conservation Biology **20:**1358–1366.

Byron, I., A. Curtis, and M. Lockwood. 2001. Exploring burnout in Australia's Landcare program: a case study in the Shepparton region. Society and Natural Resources **14:**901–910.

Bellman, R., and R. Kalaba. 1965. Dynamic programming and modern control theory. Academic Press, New York.

Bohn, H., and R. T. Deacon. 2000. Ownership risk, investment, and the use of natural resources. American Economic Review **90:**526–549.

Brandon, K., K. H. Redford, and S. E. Sanderson. 1998. Parks in peril: people, politics, and protected areas. Island Press, Washington, D.C.

Brandon, K. E., and M. Wells. 1992. Planning for people and parks: design dilemmas. World Development **20:**557–570.

Brooks, T. M., R. A. Mittermeier, G. A. B. da Fonseca, J. Gerlach, M. Hoffmann, J. F. Lamoreux, C. G. Mittermeier, J. D. Pilgrim, and A. S. L. Rodrigues. 2006. Global biodiversity conservation priorities. Science **313:**58–61.

Bruner, A. G., R. E. Gullison, R. E. Rice, and G. A. B. da Fonseca. 2001. Effectiveness of parks in protecting tropical biodiversity. Science **291:**125–128.

Clark, C. W., and M. Mangel. 2000. Dynamic state variable models in ecology. Oxford University Press, New York.

Costello, C., and S. Polasky. 2004. Dynamic reserve site selection. Resource and Energy Economics **26:**157–174.

Cowling, R. M., A. T. Knight, D. P. Faith, S. Ferrier, A. T. Lombard, A. Driver, M. Rouget, K. Maze, and P. G. Desmet. 2004. Nature conservation requires more than a passion for species. Conservation Biology **18:**1674–1676.

Deacon, R. T., and P. Murphy. 1997. The structure of an environmental transaction: the debt-for-nature swap. Land Economics **73:**1–24.

DeFries, R., A. Hansen, A. C. Newton, and M. C. Hansen. 2005. Increasing isolation of protected areas in tropical forests over the past twenty years. Ecological Applications **15:**19–26.

Ervin, J. 2003. Rapid assessment of protected area management effectiveness in four countries. BioScience **53:**833–841.

Ferraro, P. J. 2005. Corruption and conservation: the need for empirical analyses. A response to Smith & Walpole. Oryx **39:**257–259.

Ferraro, P. J., and S. K. Pattanayak. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. Public Library of Science Biology **4:**482–488.

Fredriksson, P. G., and J. Svensson. 2003. Political instability, corruption and policy formation: the case of environmental policy. Journal of Public Economics **87:**1383–1405.

Gibson, C. 1999. Politicians and poachers: the political economy of wildlife policy in Africa. Cambridge University Press, Cambridge, United Kingdom.

Halkka, A., and I. Lappalainen. 2001. Insight into Europe's forest protection. World Wide Fund for Nature, Gland, Switzerland.

Jepson, P., J. K. Jarvie, K. MacKinnon, and K. A. Monk. 2001. The end of Indonesia's lowland forests? Science **292:**859–861.

Kareiva, P., and M. Marvier. 2003. Conserving biodiversity coldspots—recent calls to direct conservation funding to the world's biodiversity hotspots may be bad investment advise. American Scientist **91:**344–351.

Kier, G. J., E. Mutke, T. H. Dinerstein, W. Ricketts, H. Küper, and B. W. Kreft. 2005. Global patterns of plant diversity and floristic knowledge. Journal of Biogeography **32:**1107–1116.

Knight, A. T., R. M. Cowling, and B. M. Campbell 2006. An operational model for implementing conservation action. Conservation Biology **20:**408–419.

Knight, A. T., and R. M. Cowling. 2007. Embracing opportunism in the selection of priority conservation areas. Conservation Biology **21:**1124–1126.

Laurance, W. F. 1999. Reflections on the tropical deforestation crisis. Biological Conservation **91:**109–117.

Laurance, W. F. 2004. The perils of payoff: corruption as a threat to global biodiversity. Trends in Ecology & Evolution **19:**399–401.

MacArthur, R. H., and E. O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton, New Jersey.

McCarthy, J. F. 2002. Turning in circles: district governance, illegal logging, and environmental decline in Sumatra, Indonesia. Society and Natural Resources **15:**867–886.

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

Medail, F., and P. Quezel. 1999. Biodiversity hotspots in the Mediterranean basin: setting global conservation priorities. Conservation Biology **13:**1510–1513.

Meir, E., S. Andelman, and H. P. Possingham. 2004. Does conservation planning matter in a dynamic and uncertain world? Ecology Letters **7:**615–622.

Milner-Gulland, E. J. 1997. A stochastic dynamic programming model for the management of the Saiga antelope. Ecological Applications **7:**130–142.

Myers, N., R. A. Mittermeier, C. G. Mittermeier, G. A. B. da Fonseca, and J. Kent. 2000. Biodiversity hotspots for conservation priorities. Nature **403:**853–858.

Naughton-Treves, L., M. B. Holland, and K. Brandon. 2005. The role of protected areas in conserving biodiversity and sustaining local livelihoods. Annual Review of Environment and Resources **30:**219–252.

Noss, R. F., C. Carroll, K. Vance-Borland, and G. Wuerthner. 2002. A multicriteria assessment of the irreplaceability and vulnerability of sites in the Greater Yellowstone Ecosystem. Conservation Biology **16:**895–908.

O'Connor, C., M. Marvier, and P. Kareiva. 2003. Biological vs. social, economic and political priority-setting in conservation. Ecology Letters **6:**706–711.

Olson, D. M., and E. Dinerstein. 2002. The Global 200: priority ecoregions for global conservation. Annals of the Missouri Botanical Garden **89:**199–224.

Peres, C. A., and J. W. Terborgh. 1995. Amazonian nature reserves: an analysis of the defensibility status of existing conservation units and design criteria for the future. Conservation Biology **9:**34–46.

Pimm, S. L., G. J. Russell, J. L. Gittleman, and T. M. Brooks. 1995. The future of biodiversity. Science **269:**347–351.

Redford, K. H., and A. Taber. 2000. Writing the wrongs: developing a safe-fail culture in conservation. Conservation Biology **14:**1567–1568.

Richards, S. A., H. P. Possingham, and J. Tizard. 1999. Optimal fire management for maintaining community diversity. Ecological Applications **9:**880–892.

Robertson, J. M. Y., and C. P. van Schaik. 2001. Causal factors underlying the dramatic decline of the Sumatran orang-utan. Oryx **35:**26–38.

Rosenzweig, M. 1995. Species diversity in space and time. Cambridge University Press, Cambridge, United Kingdom.

Siegert, F., G. Ruecker, A. Hinrichs, and A. A. Hoffmann. 2001. Increased damage from fires in logged forests during droughts caused by El Nino. Nature **414:**437–440.

Sizer, N., and D. Plouvier. 2000. Increased investment and trade by transnational logging companies in Africa, the Caribbean and the Pacific. World Wide Fund for Nature–Belgium, World Resources Institute and WWF-International.

Smith, R. J., R. D. J. Muir, M. J. Walpole, A. Balmford, and N. Leader-Williams. 2003. Governance and the loss of biodiversity. Nature **426:**67–70.

Smith, R. J., and M. J. Walpole. 2005. Should conservationists pay more attention to corruption? Oryx **39:**251–256.

Sodhi, N. S., L. P. Koh, B. W. Brook, and P. K. L. Ng. 2004. Southeast Asian biodiversity: an impending disaster. Trends in Ecology & Evolution **19:**654–660.

Stattersfield, A. J., M. J. Crosby, A. J. Long, and D. C. Wege. 1998. Endemic bird areas of the world: priorities for biodiversity conservation. Birdlife International, Cambridge, United Kingdom.

Stem, C., R. Margoluis, N. Salafsky, and M. Brown. 2005. Monitoring and evaluation in conservation: a review of trends and approaches. Conservation Biology **19:**295–309.

Strange, N., B. J. Thorsen, and J. Bladt. 2006. Optimal reserve selection in a dynamic world. Biological Conservation **131:**33–41.

Tenhumberg, B., A. J. Tyre, K. Shea, and H. P. Possingham. 2004. Linking wild and captive populations to maximize species persistence: optimal translocation strategies. Conservation Biology **18:**1304–1314.

Turner, W. R., and D. S. Wilcove. 2006. Adaptive decision rules for the acquisition of nature reserves. Conservation Biology **20:**527–537.

Wilson, K. A., M. F. McBride, M. Bode, and H. P. Possingham. 2006. Prioritising global conservation efforts. Nature **440:**337–340.

Wilson, K. A., et al. Maximising the conservation of the world's biodiversity: what to do, where and when. Public Library of Science Biology **5:**1850–1861.

Winter, C., and M. Lockwood. 2005. A model for measuring natural area values and park preferences. Environmental Conservation **32:**270–278.

World Wildlife Fund (WWF). 2004. Are protected areas working? An analysis of forest protected areas. WWF-International, Washington, D.C.

World Wildlife Fund (WWF). 2006. WildFinder: online database of species distributions. Version January 2006. WWF-International, Washington, D.C. Available from www.worldwildlife.org/wildfinder (accessed March 2006).