14.1 Introduction

Changing land use or land management affects the provision and value of a range of ecosystem services as well as biodiversity. The large number of potentially competing objectives can complicate decisions about landscape management. In rare cases the best choice among management alternatives will be obvious because one alternative delivers higher levels of all ecosystem services and biodiversity compared to other alternatives (“win–win” solutions). In most cases, however, comparing among management alternative requires evaluating trade-offs among various ecosystem services and biodiversity conservation.

The models described in this book generate predictions about the provision of multiple ecosystem services and biodiversity for any given pattern of land use and management across a landscape. In this chapter we illustrate how one might use these predictions collectively to analyze alternative conservation and management strategies. By comparing outcomes across different management alternatives, conservationists and managers can gain insight into which alternatives may be most desirable. Further, the analyses can be used to suggest and investigate new strategies that may improve results for key ecosystem services or biodiversity conservation objectives.

Trade-offs and potential win–win solutions are illustrated in Figure 14.1, which shows a simple stylized example for two ecosystem services, carbon sequestration and water quality, under four management alternatives (A, B, C, and D). For simplicity of the illustration, suppose that the cost of implementing each management is the same and that managers care only about water quality (measured on the vertical axis) and carbon sequestration (measured on the horizontal axis). Alternatives B and C are preferred to alternative A because both carbon sequestration and water quality scores are higher under these alternatives. Choosing among B, C, or D (or between A and D), however, involves a trade-off with each alternative providing more of one service and less of the other. Which of these alternatives (B, C, or D) will be preferred depends on the relative value of the two ecosystem services. If carbon sequestration is highly valued relative to water quality, then alternative D will be preferred to the other alternatives. As water quality increases sufficiently in value relative to carbon sequestration, alternative B or C will be the most preferred. Alternative A, however, will never be the most preferred option regardless of the value judgment about how to weight the value of carbon sequestration relative to water quality because it is dominated by other alternatives (B and C).

Here we describe four different approaches to analyzing conservation and management alternatives that illustrate potential application of models of the type described in this book. We start with an example that builds from conservation planning in which the planner chooses sites to include in a reserve network. We expand upon the traditional
conservation planning approach by including the effect of choosing conservation reserves on the provision and value of ecosystem services. Second, we evaluate the provision of multiple ecosystem services from alternative scenarios of land use and land management, illustrating synergies and trade-offs among ecosystem services and biodiversity conservation. Third, we combine the models with optimization to define an efficiency frontier that shows the maximum possible combinations of ecosystem services provision and biodiversity conservation that are feasible from a landscape. Fourth, we illustrate how one can include estimates of monetary value of ecosystem services to provide a benefit–cost analysis of management alternatives. At the end of the chapter we offer some brief concluding comments on the current state of the art and important next steps.

14.2 Applying ecosystem service and biodiversity models in management and conservation contexts

14.2.1 Site selection for conservation

Conservation managers typically face a situation in which they have a large number of worthwhile conservation projects but only have resources sufficient to fund a small fraction of these projects. The systematic conservation planning field developed to provide advice to conservation managers on how best to conserve biodiversity with limited resources (Margules and Pressey 2000; Sarkar et al. 2006). The conservation planning literature has developed a set of methods for choosing which sites to include in a conservation reserve network in a range of applications (e.g., Kirkpatrick 1983; Margules et al. 1988; Cocks and Baird 1989; Camm et al. 1996; Willis et al. 1996; Possingham et al. 2000) incorporating such factors as varying land cost (e.g., Ando et al. 1998; Naidoo et al. 2006), species persistence in reserves (e.g., Cabeza and Moilanen 2001; Nicholson et al. 2006) and sequential choice and threats of habitat loss (e.g., Costello and Polasky 2004; Meir et al. 2004; Wilson et al. 2006).

Even in the well-studied context of conservation site selection, spatially explicit models of ecosystem services and biodiversity can expand the type of information available to conservation managers and improve conservation decision-making. Such models can identify areas of high and low value for a variety of ecosystem services that can be compared spatially to areas of high and low value for biodiversity (Chan et al. 2006). In areas of high overlap, conservation organizations can partner with other groups interested in water quality, carbon sequestration or other services to affect outcomes, effectively increasing the resources available for conservation (Goldman et al. 2008). Conservation organizations can then concentrate their own resources on areas of high biodiversity value but that do not have high values for services.

An example of this type of analysis is shown in Naidoo and Ricketts (2006). They map the monetary values of five ecosystem services (bushmeat harvest, timber harvest, bioprospecting, existence value, and carbon storage) in the Mbaracyau Forest Biosphere Reserve in Paraguay (see Figure 14.2; Plate 8). (Some conservationists express concern about putting monetary values on nature; we discuss these issues in Section 14.2.4. Also, see Chapter 2). Naidoo and Ricketts (2006) develop maps that show areas where conservation benefits are high and would more than cover the costs of conservation and other areas where the converse is true. Naidoo and Ricketts (2006) also use these maps to evaluate three alternative locations for a proposed corridor linking two protected areas, and find that one corridor would provide much higher benefits relative to costs than the other two. This is an example of how such maps can help direct conservation efforts to high benefit areas.

![Figure 14.1](http://example.com/figure14_1.png)

**Figure 14.1** Stylized example of output of two ecosystem services, carbon sequestration and water quality, evaluated under four hypothetical management alternatives (A, B, C, and D). Moving from A to either B or C increases both carbon sequestration and water quality. Comparison among all other management alternatives involves trade-offs of an increase in one service and a decrease in the other.
Figure 14.2  Net present values in US$/ha$ for selected ecosystem services in the Mbaracayu Forest Biosphere Reserve, Paraguay. (a) Sum of all five services; (b) sustainable bushmeat harvest; (c) sustainable timber harvest; (d) bioprospecting; (e) existence value; and (f) carbon storage. (See Plate 8.)

In some cases, particularly in cases involving the protection of municipal drinking water supply (e.g., Bogota, New York City, Quito), the value of ecosystem services is high enough to choose management decisions that also support conservation (Chichilnisky and Heal 1998; NRC 2000; Echevarría 2002). In such cases, payments for ecosystem services can be more than sufficient and there is little or no need for a conservation organization to spend their scarce resources to accomplish conservation objectives. In other cases, promoting the provision of ecosystem services may align with conservation, but the services themselves may not be valuable enough to tip the balance toward biodiversity-friendly management. In this case, conservation organizations can usefully partner with other groups interested in the provision of ecosystem services. Finally, there will be other cases where management for ecosystem services does not align with conservation objectives. In these cases, conservation organizations will be on their own, just as they would be with no consideration of ecosystem services.

Spatially explicit information on ecosystem services can also be integrated with conservation planning exercises in other ways. For example, if ecosystem services are valued in monetary terms, the cost of including a particular site could be reduced by the increase in value of ecosystem services provided if the site is chosen as a reserve. Doing so would shift conservation priorities toward sites that generate valuable ecosystem services in a fashion similar to priority given to inexpensive sites. It is also possible to require that targets could be specified for certain ecosystem services and only reserve networks that met these targets would be considered as potential solutions in the conservation planning exercise.

Using spatially explicit models that incorporate both ecosystem services and biodiversity is powerful because conservation decisions are often inherently spatial: Where to protect? How much area is needed? Where to allow development? In this way, adding maps of ecosystem services broadens an existing approach to conservation planning that is used and understood by many conservation practitioners (Groves 2003). Using spatially explicit models of ecosystem services expands the set of outcomes considered in planning beyond biodiversity conservation targets. Doing so can show areas on the landscape that are of high priority for conservation targets and various ecosystem services.

The main disadvantage of using spatially explicit models in this manner is that results do not necessarily indicate how the landscape should be managed. Management to promote a particular ecosystem service might differ from management to promote another service or biodiversity conservation. For example, carbon sequestration may be maximized by planting trees but this may decrease surface water runoff and reduce availability for downstream users (Jackson et al. 2005). In classic conservation site selection the management choice is simple—either protect a site or don’t!—and a protected site is assumed to benefit all species. With the inclusion of ecosystem services, however, the choice of management options is of greater interest and complexity. When different management options at the same spatial location are best for different objectives just highlighting high priority areas on the landscape is not enough. What is needed in this case is an analysis that shows outcomes for ecosystem services and biodiversity under different types of management.

14.2.2 Analysis of management alternatives

The spatially explicit models defined in earlier chapters are designed to evaluate multiple ecosystem services and biodiversity objectives under alternative conservation or management plans. In this section we highlight the use of these models to analyze the effect of alternative land-use plans on the provision of ecosystem services and biodiversity conservation. The first case study involves evaluating alternative future scenarios for land use in the Willamette Basin in Oregon. The second case study involves evaluating alternative land uses for a watershed owned by Kamehameha Schools on O‘ahu, Hawai‘i.

14.2.2.1 Alternative future scenarios in the Willamette Basin, Oregon

Nelson et al. (2009) applied several of the spatially explicit models described in previous chapters, or their precursors. These models were used to predict changes in ecosystem services and conservation of
Figure 14.3  Maps of the Willamette Basin with the land-use pattern for 1990 and three land-use change scenarios for 2050.

Source: Nelson et al. (2009).
terrestrial vertebrate species for the Willamette Basin in Oregon, USA (Figure 14.3). Using stakeholder-defined land-use change scenarios for the period 1990 to 2050, they compared outcomes for the basin in terms of carbon storage, water quality (reduction of phosphorus discharge), soil conservation (reduction of erosion), storm peak mitigation, terrestrial vertebrate conservation and value of marketed commodities (agriculture, forestry and rural residential housing development). Basin-wide maps for the three land-use change scenarios and the 1990 land-use pattern for the Willamette Basin (Figure 14.3) were developed by the Pacific Northwest Ecosystem Research Consortium, an alliance of government agencies, non-government organizations, and universities (Hulse et al. 2002; USEPA 2002; Baker et al. 2004).

The three land-use change scenarios were: (i) “plan trend” that extended current policies and trends into the future, (ii) “development” that relaxed current land-use policies and allowed greater freedom for market forces, and (iii) “conservation” that gave greater emphasis to ecosystem protection and restoration (USEPA 2002, pp. 2–3).

Of the three scenarios, the conservation scenario produces the best results for all ecosystem services and biodiversity conservation (Figure 14.4). The results for the conservation scenario were significantly better than for either the plan trend or development scenarios for carbon sequestration, water quality, and soil conservation. Only the market value of commodity production was higher in the plan trend and development scenarios than in the conservation scenario. Under the plan trend and development scenarios, more land was devoted to housing development and timber production increasing the value of market returns (but lowering the scores for biodiversity conservation and many ecosystem services).

The trade-off between the value of marketed commodities and ecosystem services changes if we expand the market of carbon credits. Nelson et al. (2009) calculated the aggregate market value of carbon sequestration under the three scenarios using a price of $43 per metric ton of carbon, which is the mean of estimates of the social value of carbon sequestration from peer-reviewed studies (Tol 2005). Because there was more carbon sequestered under the conservation scenario, adding the carbon sequestration value to the market value of commodities meant that the conservation scenario generated the highest monetary returns of the three scenarios (Figure 14.5). A carbon market that rewarded carbon sequestration could turn a trade-off curve with a negative slope (Figure 14.5, circles) into one with a positive slope (triangles), converting a trade-off into a win–win. Making payments for other ecosystem services would further increase the value of the conservation scenario relative to the other two scenarios.

14.2.2.2 Kamehameha Schools, O‘ahu, Hawai‘i

A subset of the spatially explicit models described in earlier chapters were also used to evaluate impacts on ecosystem services for local land-use planning in Hawai‘i in collaboration with Kamehameha Schools, an educational trust and the largest private landowner in the state (see Box 14.1 for additional information on Kamehameha Schools). The analysis focused on Kamehameha Schools’ land holdings on the north shore of the island of O‘ahu (Figure 14.6; Plate 9). This region contains approximately 26,000 acres stretching from ocean to mountain tops, including ~2000 acres of coastal rural community lands, ~9000 acres of agricultural lands in the middle section (once a sugarcane plantation, now largely abandoned and invaded by exotic species), and ~15,000 forested acres in the upper part.

With extensive input from Kamehameha Schools, three spatially explicit scenarios were created to explore contrasting directions that could be taken with the agricultural lands:

1. Sugarcane ethanol—returning the plantation lands to sugarcane cultivation to produce ethanol biofuel;
2. Diversified agriculture and forestry—using the lower irrigated fields for diversified agriculture, establishing vegetation buffers to reduce field runoff, and undertaking native forestry plantings on the remaining higher elevation fields;
3. Residential subdivision—selling coastal and plantation lands for a residential housing development.

These scenarios were compared in terms of effects
on water quality (for nitrogen discharge), carbon storage, and income generation.

All three scenarios are projected to generate positive income streams that exceed the current negative returns (Figure 14.7). The residential subdivision scenario, not surprisingly, has the greatest net present value of income. This income boost, however, is linked with reductions in carbon stock (6.8%) and water quality (21.1%) relative to the current landscape. Impacts on carbon stock and water quality are even more pronounced for the sugarcane ethanol scenario with reductions of 12.6 and 44.2%, respectively. In both cases, losses in carbon stock are driven by clearing invasive woody vegetation on abandoned fields. While both scenarios lead to reductions in carbon stock, the sugarcane ethanol scenario has the potential to “pay off” the lost carbon stock through use of ethanol to offset more carbon-intensive energy sources. Following the biofuel carbon debt methodology of Fargione et al. (2008), the estimated payback period is approximately 10 years to return to baseline conditions.

The remaining scenario, diversified agriculture and forestry, is projected to improve carbon stock (9.8%) and water quality (7.0%) relative to the current landscape, while also generating positive income. These improvements are driven by plantings to restore native forest cover and establishing...
Box 14.1 Plight of a people

Neil Hannahs

Disease and change exacted a horrific toll on the native people of Hawaii throughout the nineteenth century. The thriving population of more than half a million Hawaiians at the beginning of the century had dwindled to a mere 40,000 by the 1880s. To address these desperate conditions and to assure the perpetuation of Hawaiian culture and welfare of her people, Princess Bernice Pauahi Bishop and her husband left over 400,000 acres of Hawaii land, as well as personal resources, in a perpetual charitable trust dedicated to improving the wellbeing of Hawaiian people through educational services offered by Kamehameha Schools (KS).

Since its inception in 1884, the endowment of KS’ founders has been managed to produce financial resources to build and maintain campuses and educational programs. For much of the School’s history, the trust was considered land rich, but cash poor. To fund construction, operation and growth, an asset management strategy was adopted to maximize economic productivity. This provided the means for KS to become one of the largest private educational institutions in the world and afforded the Schools the opportunity to greatly expand its educational reach.

However, the commercial, residential, and agricultural land developments that brightened KS’ economic prospects were often conducted with insufficient regard for cultural resources, environmental impacts and community values. This tendency, coupled with rapid population growth fueled by in-migration and the introduction of invasive exotic species of flora and fauna, resulted in displacement of Hawaiian communities and degradation of indigenous resources and the cultural practices that thrived upon them. These circumstances produced a tragic conundrum: Hawaiians being helped by KS suffered the most from land-use changes implemented to provide resources for their educational programs.

Concern for Hawaii’s ecosystems and traditional lifestyles mounted over the past four decades as Hawaiian culture experienced a renaissance and natural resources became increasingly stressed. Resource supply has declined in the face of rising demand and ravaging impacts of invasive plants and ungulates. Conflicts manifested as resistance to new development and Western concepts of property rights, as well as advocacy for constitutional and regulatory protections of the environment, at-risk species and Hawaiian cultural practices. Consequently, KS’ efforts to apply economic maximization strategies to undeveloped

Figure 14.A.1 Kamehameha Schools’ representation of the indigenous Hawai‘i worldview.
lands faced increasing resistance in the latter twentieth century.

**Paradigm shift**

The "Kamehameha Schools' Strategic Plan 2000–2015" promised an organization that would align itself to the values of the founders, incorporate the views of stakeholders and set new directions. The Plan established the following goals for the management of the endowment. Kamehameha Schools will optimize the value and use of the current financial and non-financial resources and actively seek and develop new resources; and to practice ethical, prudent and culturally appropriate stewardship of lands and resources.

These goals provide an opportunity to re-think the value of land and each asset’s role in fulfilling the mission as part of a dynamic portfolio. The emergent Integrated Management Strategy has attracted the interest of cultural stakeholders and other First Nations peoples, as well as the conservation and business communities, including the Natural Capital Project (NatCap).

NatCap’s InVEST tool, the software framework for several of the models described in earlier chapters, has helped to inform courses of action and land management decisions that propel a shift from one dimensional returns to a balance of desirable outcomes. KS, owner of the Kawailoa lands to which InVEST is now being applied, has depicted its efforts to achieve an optimal balance of multi-value returns as an image of over-lapping spheres.

A risk inherent in this view, as well as in using a tool like InVEST, is that the challenge might be met by assembling indiscriminate and disconnected considerations in each value domain. An alternative approach is to maintain focus on holistic, living systems. This is depicted in the taro (kalo) image (Figure 14.A.1).

Kamehameha Schools is now monitoring several key performance metrics of sustainability to determine whether this high standard is being achieved. These include carbon footprint; assessments of ecosystem services; financial values and returns; and various measures of well-being impact. InVEST is playing an integral role in helping KS and others in projecting the outcome of land-use decisions on many of these indicators of vitality.

**14.2.3 Generating an efficiency frontier**

In the previous section, we showed how to use multiple spatially explicit models to analyze specific scenarios (management alternatives) of interest to users. Such analyses can show which of the considered management alternatives generates better performance in terms of provision of ecosystem services or meeting biodiversity targets. Another use of these models is to show what is possible to achieve on the landscape by considering all potential land-use scenarios. In reality, of course, not all land-use scenarios will be politically or socially acceptable. But considering all possible alternatives can often identify solutions that are far superior to the narrow range of options currently being considered. Providing this evidence can broaden the perspective of users and begin a dialog about what options should be on the table.
Figures 14.6 and 14.7. Land use/land cover maps on the north shore of O‘ahu. The area shown here includes all of Kamehameha Schools’ north shore land holdings, as well as small adjacent parcels that make for a continuous region. The baseline map is from the Hawai‘i Gap Analysis Program’s land cover layer for O‘ahu (Hawai‘i Gap Analysis Program 2006) (See Plate 9.)

Figure 14.7 Projections of carbon stock and income from the plantation lands for the north shore region of O‘ahu for the baseline land use/land cover map and the three planning scenarios (sugarcane ethanol, diversified agriculture and forestry, residential subdivision).
By combining ecosystem service models with optimization methods, one can determine the maximum feasible combinations of ecosystem services and biodiversity that can be achieved on a landscape. The results of this analysis can be presented with an efficiency frontier, which is defined as the outcomes for which it is not possible to improve on any particular objective (ecosystem service or biodiversity conservation) without decreasing performance on some other objective.

Polasky et al. (2008) estimated such an efficiency frontier for conservation of terrestrial vertebrates and the value of marketed commodities (timber, agricultural output and housing) for the Willamette Basin in Oregon. They developed models that used a land-use plan for the basin as input and reported output in terms of the expected number of terrestrial vertebrate species that would persist in the basin (biological score) and the value of marketed commodities (economic score). Using optimization methods from operations research, they searched for land-use plans that maximized the biological score for a given economic score. Then by repeating this analysis across the full range of economic scores ($0–27.6 billion) they traced out an efficiency frontier (Figure 14.8; Plate 10). The results show that it is possible to achieve both high biological and economic scores by thinking carefully about the spatial pattern of land use in the basin. For example, the land-use plan that for point D in Figure 14.8 (Plate 10) generates a biological score of 248.5 species and an economic score of $25.8 billion. This outcome is far better than the outcome generated by the current land use (point I in Figure 14.8; Plate 10), a biological score of 238.6 and an economic score of $17.1 billion (Polasky et al. 2008).

Analyses such as these can demonstrate what is possible for a given region and how much improvement can be made by careful planning. Because of political, social and economic complications, it may not be possible to reach efficiency frontiers. Still, knowing what it is possible can provide a spark to ignite efforts to improve upon current performance.

![Figure 14.8](Plate 10)
14.2.4 Benefit–cost analysis

Results from ecosystem service models can be reported in biophysical units or in monetary values. Much of the analysis on ecosystem services to date has been reported in biophysical units, including most of the case studies discussed above. In some settings, such as dealing with government or private sector managers used to thinking in monetary terms, it may be advantageous to report results of the analysis in terms of monetary values. Doing so may also make it easier to compare management options. Because results are reported in a single metric (i.e., dollars), managers can compare apples with apples rather than with oranges.

Economists have developed a variety of market and non-market valuation methods that can be applied to estimate the monetary value of ecosystem services (Freeman 2003). The estimates of monetary value can be incorporated into benefit–cost analysis to analyze the net benefits of alternative management alternatives. Naidoo and Ricketts (2006) in Section 14.2.1 and Nelson et al. (2009) in Section 14.2.3 are examples of how monetizing ecosystem service values can result in benefit–cost analyses that can inform managers and potentially improve management decisions.

Translating from biophysical units to monetary value units, however, is problematic for biodiversity targets and some types of ecosystem services. In some cases trying to convert oranges into apples will result in pulp rather than a recognizable fruit. For example, trying to estimate the monetary value of cultural and spiritual values is controversial (see Chapters 2 and 12; Norton 1991; Sagoff 1988). Valuing the existence of species is viewed as morally objectionable and inherently misguided by some (e.g., Ehrenfeld 1988; McCauley 2006), and even some economists who have tried to value biodiversity admit to the practical difficulties of doing so (e.g., Stevens et al. 1991). Other economists think that all values, including the value of biodiversity, can be measured using economic methods as long as the analysis is done properly (e.g., Loomis and White 1996). For some ecosystem services, estimating monetary values using market prices or applying non-market valuation techniques may be relatively uncomplicated and uncontroversial.

Examples include the value of provisioning services such as timber or fish (e.g., Naidoo and Ricketts 2006; Barbier 2007; Polasky et al. 2008; Nelson et al. 2008, 2009; Chapter 8), or crop pollination, which is an input to a priced commodity (e.g., Ricketts et al. 2004; Chapters 9, 10). Depending on the ecosystem services and the decision context at hand, the user of these models can decide whether it is better to use biophysical units or monetary values.

Chapters in this book typically aim to monetize the value of ecosystem services, but we do not attempt to translate biodiversity targets to a monetary measure of value (Chapter 13). That is because biodiversity is a fundamental attribute of natural systems, which may contribute to the provision of various ecosystem services but which also has intrinsic value (i.e., value in and of itself). Even without attempting to put monetary value on biodiversity, one can still show feasible combinations of biodiversity and services, along with potential trade-offs between them (as shown in Section 14.2.3). Then managers can decide for themselves what trade-offs are acceptable.

14.3 Extending the frontier: challenges facing ecosystem management

Integrated landscape-level analysis that tracks changes across a number of dimensions of ecosystem services and biodiversity conservation is still a relatively young discipline. Models of ecosystem services and geographically explicit data sets are developing rapidly, offering the prospect of further improvements in the near future. To date, applications in the USA, South Africa, Paraguay, and elsewhere have demonstrated the power and utility of an integrated spatially explicit landscape-level approach. Application of such models can generate information for decision-makers showing the consequences of choices for a range of important ecosystem services and biodiversity conservation objectives. In principle, putting this information in the hands of decision-makers should lead to improved landscape planning and management.

To fully realize the promise of spatially explicit integrated modeling approaches, further improvements will be necessary. As discussed in Chapter 15,
more work on improving and validating the component models of particular services is needed. Our understanding of the links between management actions and provision of ecosystem services is limited for many services. Additional empirical research on provision of services in a wide variety of circumstances will improve understanding and accuracy of models. Additional understanding of ecosystem functions and conditions that link together provision of multiple services, such as connections between land cover, water availability, nutrient cycling and local climate, will also improve the overall modeling effort. Perhaps the greatest need on the biophysical modeling side, however, is improved understanding and inclusion of system dynamics and feedback effects. Coupled human and natural systems may exhibit threshold effects and non-linear responses in which provision of ecosystem services might change suddenly as conditions in the system evolve.

Even with knowledge of biophysical systems, understanding the provision of ecosystem services also requires detailed understanding of what is of value to people. For example, the provision of clean drinking water in areas without people will not provide an ecosystem service of value while the same provision in a watershed providing water to a major city will have great value. Understanding the value of ecosystem services requires integration of natural and social science. Such integrated work, partly in response to the focus on ecosystem services, has begun to expand rapidly in recent years but is still limited relative to what is needed to seamlessly integrate the supply of services (primarily the province of natural science) with the demand for services (primarily the province of social science). Integrated understanding of ecosystem services has progressed to the point where we can highlight important areas on a landscape for ecosystem services. In many cases, however, we cannot yet provide the level of certainty, either in terms of biophysical or economic modeling, to underpin payments for ecosystem services or other policy approaches that require numerical estimates of value (see Chapters 15 and 19 for further discussion).

An important aspect of integrated spatially explicit models is the ability to show not only the total value of ecosystem services to society but also the distribution of benefits to various groups in society. Such distributional analysis is important for understanding the effects of conservation and management decisions on the poor (see Chapter 16). Distributional analysis of the people who benefit and bear the costs of alternative conservation and management is also important for the design of policy approaches to ensure that those who make decisions affecting ecosystems have incentives to provide ecosystem services of value to society (see Chapter 19).

It is important to recognize that estimates of value, spatial priorities, trade-off analyses, and most other results reviewed in this chapter depend strongly on the choice of ecosystem services to include. Ecosystem management affects a large range of ecosystem services, not all of which may be feasible to model given limited time, resources, data or scientific understanding. In many cases, water quantity and quality, carbon sequestration, and the market value of commodities will be of great importance. Biodiversity conservation will be of primary importance in many conservation applications. However, other services may also be important in particular applications (e.g., non-timber forest products, pollination services, effects on poverty, number of jobs). Early and continuous engagement with people potentially impacted by ecosystem management is the best approach to ensuring that the most important ecosystem services and other policy dimensions (e.g., number of jobs) are included in the analysis.

Finally, the analysis of integrated spatially explicit models is but one step in a much larger and longer process needed to implement real change on the ground. As Knight et al. (2006) and Cowling et al. (2008) emphasize, there are plenty of analyses and reams of plans but far less action, and that “our understanding of these techniques currently far exceeds our ability to apply them effectively to pragmatic conservation problems” (Knight et al. 2006, p. 408). Spatially explicit integrated models can provide useful information but unless they are embedded in a larger policy process that involves those who use land and resources the information will not be utilized to improve ecosystem management or conservation outcomes.
References


