Chapter 3

Assessing Responses

Coordinating Lead Authors: Dana R. Fisher, R. David Simpson, Bhaskar Vira
Lead Authors: W. Bradnee Chambers, Debra J. Davidson
Review Editors: Richard Norgaard, Ravi Prabhu

Main Messages **73**

3.1 Introduction **73**

3.2 Political Factors **75**
3.2.1 Stakeholders
3.2.2 Political Structures

3.3 Institutional Factors: Capacity for Governance **77**
3.3.1 International Level
3.3.2 Domestic (National) Level
3.3.3 Local Level

3.4 Economic Factors **80**
3.4.1 Valuing Benefits
3.4.2 Cost-effectiveness
3.4.3 Secure Property Rights

3.5 Social Factors **85**
3.5.1 Equity
3.5.2 Cognitive Differences and Ecological Beliefs

3.6 Methods and Tools for Assessing Responses **86**

3.7 Conclusion **89**

REFERENCES **90**

BOXES

3.1 Political Bargaining over Ecological Responses: "Job Blackmail"

3.2 Institutional Resilience: The Ability to Adapt

3.3 Determinants of Compliance

3.4 Assessing Implementation

3.5 The Equity Implications of Forced Displacement

3.6 The Post-materialism Hypothesis

3.7 Adaptive Management and Adaptive Co-management

3.8 Canada's Model Forest Program: Integrating Multiple Dimensions into Responses

FIGURES

3.1 The Assessment Method

TABLES

3.1 A Framework for Assessing Responses

3.2 Proposed Response Assessment Matrix

3.3 Indicators for the Impact of Responses

3.4 Qualitative Assessment of Trade-offs and Synergies

3.5 Quantitative Assessment of Trade-offs and Synergies

3.6 Deliberative Tools

3.7 Information Gathering Tools

3.8 Planning Tools

Main Messages

Assessment of responses should distinguish between constraints that render a policy option infeasible and the acceptable consequences or side effects of a chosen strategy. We are proposing a multistage assessment process, which focuses first on those factors that may either rule out a particular response or be critical preconditions for its success (*binding constraints*). Responses are then compared across multiple dimensions in order to identify unintended impacts, focusing on identifying compatibility or conflict between different policy objectives. The pursuit of a specific objective may sometimes involve compromising another policy goal. Such considerations, while important, may be seen by decision-makers as acceptable costs associated with the implementation of an option (*acceptable trade-offs*).

Evaluating the relative success of responses requires an assessment of enabling conditions, binding constraints, and acceptable trade-offs across a number of domains. These include the political, which encompasses the legitimacy of and the political context for the response; institutional, which refers to the capacity for governance and implementation; economic, which looks at the availability of resources as well as the aggregate and distributional impacts of policy options; social, which refers to the broad social environment and preconditions for a response; and ecological, which defines systemic preconditions and constraints for a response. As many other chapters of the MA consider the ecological domain in detail, we will, while recognizing the central importance of ecological considerations, focus on the other four domains in this chapter.

The assessment of responses needs to recognize trade-offs between objectives. It is unlikely that all strategies will be able to satisfy diverse and often competing policy objectives. Resolving the trade-offs between these different objectives presents a significant challenge to determining appropriate responses. In some instances, it may be possible to make a "binary" decision: so long as some standard is satisfied, the choice among approaches can be made on other grounds. In other situations, a gain toward achieving one objective may need to be weighed against a negative outcome in some other domain.

Some responses may constitute "win-win" opportunities. While trade-offs between objectives are likely to occur, synergies are certainly possible. Some responses may constitute "win-win" opportunities. Policy-makers ought to remain alert for such opportunities and move aggressively to act upon them, but also remain guarded concerning the prospects of options that may "sound too good to be true."

Aggregating response impacts across different dimensions is a subjective process. Quantitative assessment techniques are not necessarily preferable to qualitative methods. Aggregating impacts across different dimensions (political, institutional, economic, social, and ecological) is difficult. Quantification may provide a "false" objectivity to what is essentially a subjective process. Decision-makers must, in the final analysis, make some assessment of the "weights" to be assigned to each factor and compare impacts along dimensions that are typically incommensurable.

Assessment methods must be sensitive to a plurality of perspectives. The assessment of responses needs to be multidimensional, involve inputs from multiple disciplines, and must attempt to integrate the perspectives of multiple decision-makers. Techniques that adopt a pluralistic disciplinary perspective are particularly pertinent, as they do not privilege any particular viewpoint.

A number of pluralistic decision-making tools and techniques are available. These tools can be employed at a variety of scales—global, sub-global, and local. This chapter presents a simple listing of tools that are available, as well as a preliminary analysis of their most appropriate scale(s) of application. In particular, a distinction can be made between *deliberative* tools, which facilitate the process of dialogue over responses; *information gathering* tools, which are primarily focused on collecting data and opinions; and *planning* tools, which are typically employed for the evaluation of potential policy options.

Assessment is a dynamic and adaptive process, which needs to be constantly updated in light of new information, as well as feedback from the social and ecological systems in which a response is implemented. Techniques such as adaptive management and adaptive co-management have been deployed usefully to create flexible and resilient systems of resource management. The advantage of such approaches is that they are able to deal with new empirical circumstances while ensuring that responses reflect the perspectives and interests of a wide variety of stakeholders.

The assessment process is only as good as the overall decision-making environment within which it is embedded. Trade-offs, choices, and synergies are often hidden or neglected in policy dialogue. Solutions to many intractable problems are likely to be context-specific, and it may not be easy to achieve consensus among stakeholders about the suitability of specific responses. A process in which choices and trade-offs are transparent is desirable, as it is most likely to allow decision-makers to choose locally appropriate responses that are congruent with their desired goals.

Because stakeholders will be affected differently, and may have differences of opinion about the relative desirability of different response strategies, consensus will be difficult. The potential for conflict is particularly high where there is disagreement among stakeholders over the objectives of intervention as well as the means to achieve these ends. For instance, while environmental ministries may prioritize ecosystem integrity, economic ministries may privilege economic growth. Bureaucracies may prefer centralized authority structures, while grassroots organizations may be more comfortable with inclusive and participatory approaches. Some of these differences may be reconcilable, but in other cases it may not be possible to achieve consensus among stakeholders.

3.1 Introduction

In the MA conceptual framework (MA 2003), responses are defined generically as *human actions, including policies, strategies, and interventions, designed to respond to specific issues, needs, opportunities, or problems*. Responses are seen in the context of perceived needs or problems. In the specific context of the MA, these needs or problems relate to the preservation of ecosystems and biodiversity, the accrual of desired ecosystem services, and the improvement of human well-being.

This chapter evaluates the human influences on responses that must be considered by decision-makers in response assessment. In this instance, we employ the term decision-makers broadly, to include all individuals who are in a position to promote an ecological response option, at the local, regional, national, or international level. There are at least two distinct, but interrelated, reasons why decision-makers need to evaluate responses. The first is to improve policy-making by learning from experience. Here, the decision-maker seeks to understand the reasons for perceived success and failure, and considers how such conditions can be replicated for future policy-making that is targeted at enhancing human well-being and ecosystems.

The second reason is to understand the impact of any particular response or set of responses. The need is to identify the linkages between the chosen responses and their effects on a wide range of proximate social, political, economic, and ecological variables, and ultimately on human well-being and ecosystems. We include a discussion of methods that may be employed to assess these variables, in order to maximize the potential success of responses and minimize the potential unintended effects that may arise in relation to response implementation. Evaluating responses can be a complex and costly endeavor, because of the need to understand the multidimensional impact of any chosen strategy, and the multiple actors and interests that may be involved in the process.

In the past, responses have fallen short of their intended goals due to, for example, an inadequate estimation of the skills and resources required for implementation, or a lack of understanding of the sources of cultural resistance to the behavioral changes required. Other responses, whether or not they meet intended ecological goals, can have extremely disruptive social consequences, such as when land tenure allocations are abruptly altered, causing conflict among pre-existing user groups. This chapter stresses the importance of trying to understand how social factors can hinder responses and how responses can lead to unintended social consequences. It proposes the use of evaluation methodologies that stress the employment of multiple criteria and a plurality of inputs into the decision-making process. Such methodologies are relevant to the assessment of a variety of responses, but are intended to be applied in the present context to responses that are targeted at the flows of services from ecosystems, as well as those that are implemented in other social sectors that may have indirect implications for ecosystem services and human well-being.

Understanding the relative success of responses requires an assessment of the enabling conditions and *binding constraints* that determine which specific objectives can be pursued, because they either may rule out a particular response or may represent critical preconditions for its success. Binding constraints are factors that render a policy option infeasible. These are distinguished from what we call *acceptable trade-offs:* unintended impacts associated with the implementation of a response that may be deemed acceptable because they are outweighed by benefits of the response. What is considered a binding constraint, and what is considered an acceptable trade-off, is in all instances context-specific; the proposed assessment method is not intended to elicit generalizations, but intended for use on a case-by-case basis. As a result, we purposefully avoid establishing a list of specific indicators, as these are expected to vary according to the specific contexts under consideration. More importantly, what is considered a binding constraint, and what is considered an acceptable trade-off, may also be seen differently by different stakeholders *within* cases. While transparent processes that utilize a deliberative democratic format have been shown to be tremendously successful in eliciting stakeholder support for a common course of action, deliberation is the key to decision-makers' understanding of different perspectives on any particular response. We recognize that, in some cases, the differences among perspectives may be so great that a resolution is not possible.

This chapter proposes a three-stage assessment process that focuses first on identifying the multiple human impacts associated with responses, along five domains described further below. In the first stage of assessment, those impacts that pose binding constraints are identified. These factors may explain the failure of a previous response, or may rule out its adoption in proposed planning processes, and will require either the selection of an alternative response or significant investments in creating more favorable conditions. If the impacts identified do not impose binding constraints on a particular response, they may be considered acceptable trade-offs, which may include both positive synergies and negative consequences within these domains. In the second stage of the assessment process, these potential trade-offs, and their acceptability in relation to the response, are identified. In this step, responses are compared across multiple dimensions, focusing on identifying compatibility or conflict between different policy objectives. Once these two steps are completed, decision-makers are ready for the third and final stage in the evaluation process, which entails the selection of preferred responses.

The assessment procedure is designed for use by diverse decision-makers at multiple spatial and temporal scales: from the local to the global, for the analysis of previous or current responses, as well as for the evaluation of the feasibility of proposed policies or responses to be implemented in the future. Further, the assessment procedure is intended to be a dynamic process, whereby new information and systemic feedback creates policy learning and the evolution of responses in an adaptive manner.

The assessment method is outlined in Figure 3.1. It is important to note that although this is a new process for assessment, the tools and methods that it draws on are based on the existing literature. Because it is an assessment of what exists, no new techniques/tools are being developed. In both stages of assessment, binding constraints and acceptable trade-offs should be evaluated in relation to five domains:

- the *political,* encompassing the legitimacy of the response and the political context in which the response would be implemented;
- the *institutional,* referring to the capacity for governance and implementation;
- the *economic,* referring to the aggregate and distributional consequences of the response for income and wealth, and economic conditions, including stability of property rights and the efficient use of available resources;
- the *social,* including the broad equity issues associated with a response; and
- the *ecological,* including the ecosystemic preconditions and context within which a response is being considered.

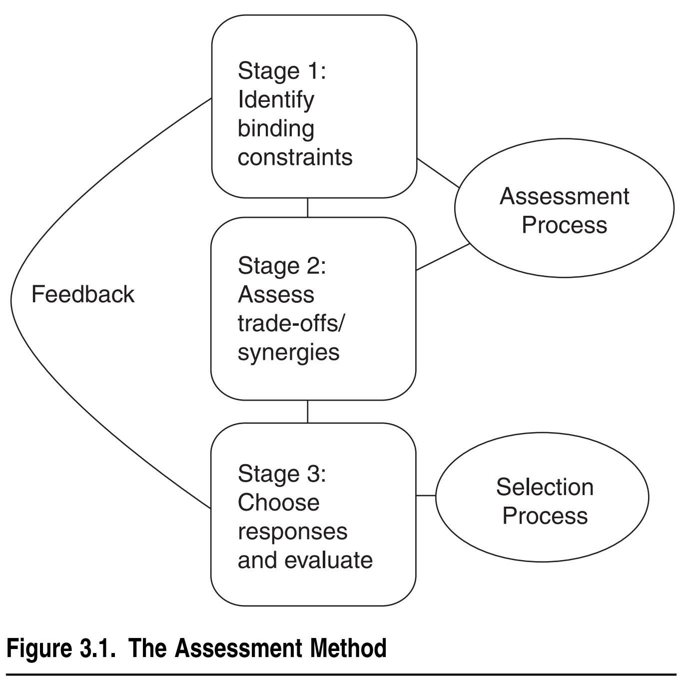

Figure 3.1. The Assessment Method

First and foremost, political context matters. Responses, whether they are limited to a local region, or are national or international in scope, have the potential to generate heated political debate and, in some cases, sufficient opposition to prevent further progress. On the other hand, the institutional sponsor of a given response may depend upon relations with other political stakeholders for resources and support. The following section of this chapter provides a social-scientific understanding of the political environment, paying particular attention to the political feasibility of responses and the potential sources of political opposition. Even those responses that are politically viable may not be effective because they may be beyond the capacity of the organizations that are assigned responsibility for their implementation. Subsequent sections discuss: the need to assess the institutional capacity for implementation at all potential levels of governance, including the local, provincial, national, and international; the need for economic analysis and how different options may perform relative to a range of economic criteria; and the social implications of responses, especially the "unintended consequences," or social externalities, that emerge as a result adopting particular policy choices.

Although these domains are treated as independent for the purposes of analysis, it is important to recognize their interrelationship in practice. The social context describes a set of broad parameters within which economic, political, and institutional activities function. We know that economic shifts, for example, inevitably influence the social, institutional, and political domains, just as each of these domains exerts influence on all others. Moreover, all human activities take place within an ecological context. Activities in any of the domains discussed here have direct implications for ecosystems, which in turn set ultimate boundaries on the range of human activities that can be sustained. A discussion of assessment of the ecological domain clearly warrants extensive treatment unto itself, and is considered in other chapters. Consequently, although we refer to this domain in several tables as a necessary feature of any response assessment, specific details are not considered further in this chapter.

The final section of Chapter 3 outlines in greater detail our evaluative method, which emphasizes that, in any *particular* policy environment, an effective assessment of responses should be multidimensional, involve inputs from multiple disciplines, and attempt to integrate the perspectives of multiple decision-makers. In such a pluralistic environment, it is possible that difficult trade-offs and choices between alternatives will dominate decision-making, although there may be opportunities for synergy. The suggested methods are intended to make these trade-offs, choices, and synergies explicit, since they are often hidden or neglected in policy dialogue. Difficult choices are often involved in decision-making, and it is usually not possible for strategies to achieve all desirable policy objectives. Solutions to these often-intractable problems are likely to be context-specific, and it may not be easy to achieve consensus among stakeholders about the suitability of specific responses. *However, it is desirable to follow a process in which these choices and trade-offs are made transparent, to enable decision-makers to choose responses that are appropriate to the context and congruent with their desired goals.*

3.2 Political Factors

Since responses are understood here as conscious efforts to change existing social structures or behavior, it is important to consider the political environment in which such changes are to be implemented. Responses may be difficult to introduce if the political conditions are unfavorable. Decision-makers need to assess the feasibility of responses based on the political environment in which these options are to be implemented. The political environment is defined by the actors and interests who have a stake in the response and by the political structures within which their strategies are pursued, including the process by which issues command attention and become part of a policy agenda. If the external political environment is assessed to be favorable, it may be possible to introduce a desired response; alternatively, it may be necessary to invest resources in political activities that would create a more conducive climate for implementation, or to alter the response so that it is more appropriate for a given political climate. Assessment of the political domain thus involves the identification of stakeholders, an evaluation of the relative power of each to influence ecological responses, and a characterization of the political structures involved.

3.2.1 Stakeholders

The first step in assessing the political feasibility of a given response involves the identification of those individuals and groups who are likely to be actively involved in, or may be affected by, either the formulation or implementation of a given response. These stakeholders include, but are not limited to, political actors, interest groups, social movements, implementing groups, political activists, power-brokers, and consumers. It is necessary to look at the specific roles that different stakeholders play in the political processes surrounding ecological response strategies. It is also important to recognize that the implementation of some responses involves multiple scales of decision-making. Building a dam to protect a flood plain in a rural area in a developing country, for example, might involve stakeholders at the local level of the village that is located in the flood plain, as well as the international financial organization that will fund the project.

Understanding the stakeholder community requires going beyond simply identifying potential stakeholders and their interests, to include an assessment of the relative power of each. When considering responses, it is necessary to identify the key stakeholders who are relevant to the strategies under consideration, and their potential for political mobilization. In general, stakeholders include those who already have the political influence to affect the ecological response and are motivated to employ it; those who wish to affect the ecological response and are actively seeking to acquire the political influence to do so; and those who are affected by the ecological response but are not actively seeking, or cannot reasonably be expected to acquire, policy influence.

Measuring power can be a complex task, due to the elusive nature of its exercise. In the past, many political scientists simply evaluated the relative ability of organizations to have their interests addressed in the political arena; clearly those whose agenda received the greatest level of support from policy-makers and other elected officials have the most power. Reliance on observations of such visible expressions of power, however, leaves many groups at the lower end of the power spectrum unidentified. Many individuals, despite the existence of grievances, simply do not participate, because they do not believe their efforts will pay off. Other groups have become so disempowered that they internalize the existing power structure and come to accept their lack of power as justified. This "third dimension of power" (Lukes 1974) often expresses the condition of oppressed groups. In many instances, the lack of power of these groups can itself pose a hindrance to ecological responses, as their participation may be necessary for implementation.

The most effective means to assess the relative power of stakeholders is through identification of their concerns and interests, and an evaluation of the respective success of each stakeholder in their efforts to pursue their concerns in the political arena, while paying particular attention to groups outside the political arena whose interests are clearly not being addressed. Further, it needs to be kept in mind that ''stakes'' are continually being renegotiated and redefined, so that this process is inherently dynamic.

In addition to stakeholders in civil society, the state is central to an understanding of the political environment within which responses are considered. Not only is the state itself a stakeholder, the structure of the nation-state (discussed further) defines relations between state and society by, for example, defining the level of tolerance and legal parameters of organized protest, and the specific steps involved in policy formulation. As such, the specific structure of a given nation-state has tremendous influence over which stakeholders are accorded influence, as well as the extent of, and the nature of, their influence. Nation-states are far from uniform. They not only vary in structure and function across the globe, but a given nation-state should also not be treated as a singular social actor. A state is a combination of actors and institutions, encompassing manifold activities that include everything from political fundraisers, legislative committee hearings, and consultative meetings, to policy implementation on the ground (Laumann and Knoke 1987, p. 381; Chubb 1983). Ecological responses include state involvement with civil society at multiple scales—local, sub-global, and global—which makes the relationship among organized interests and the state in its multiple forms all the more complex.

In other words, responses—which in many cases take the form of political decisions—are the product of the interrelations of multiple stakeholders and state institutions (Fisher 2004). These stakeholders, however, are not only working to affect the state, they are also influenced by the state themselves (Chubb 1983; see also Austen-Smith and Wright 1994).

3.2.2 Political Structures

Just as the list of stakeholders may vary in different geographical contexts, at different scales, and according to the specific political issues in question, the political structures that define stakeholder relations with the state, and establish the process within which policy-making occurs, will also vary. In some countries, it may be acceptable practice for citizens to hold rallies and demonstrations, or to litigate against the government, while in other countries it is not. Other forms of political activity used by stakeholders may include support of legislative candidates, distribution (and receipt) of informational material designed to sway public opinion, lobbying, testifying at public hearings, signing petitions, writing letters to legislators, or serving on citizen advisory panels. In short, these opportunity structures determine the distribution of power in a social system, and are defined by: cultural/traditional institutions; behavioral norms; legal/constitutional mandates; formal political structures that determine the ''rules of political engagement''; and the influence of international regimes.

Political scientists and sociologists often distinguish among several stages in the policy-making process, but emphasize the importance of the first two (which can overlap in practice)—agenda setting and policy formulation. Of all the possible issues of concern among members of a social system, only a small number ever make it to the political agenda and become the focus of policy-making—a process heavily influenced by organizations capable of dominating the discourse and the selection and portrayals of political issues in the media. One of the more notable trends in environmental and ecological governance in recent years has been the tremendous growth in complexity, both of environmental concerns and of the political environment within which agenda-setting takes place. Many ecological concerns fail to receive adequate attention, simply because the immediate costs and latent benefits of many response options render them unattractive to elected officials in representative democratic systems, whose primary concern may be re-election within a short time horizon.

In past decades, governmental policy-makers have worked within a closed network of legislators, regulators, and, in some cases, relevant industry representatives, described as the ''iron triangle'' of regulation (Wilson 1980). Today, this iron triangle remains in place in regard to certain policy issues and in some regions. While industry organizations still tend to dominate in many contexts, agenda-setting in environmental and natural resource domains is coming under increasing scrutiny as groups in civil society, including environmental organizations, community-level justice organizations, the media, and scientific institutions, vie for influence. Not only have more social actors entered into political discussions, even the scale of environmental policy-making has expanded. Many environmental policies play out on the international stage at the same time that they are being negotiated internally within countries (Putnam 1988; Evans et al. 1993; De Sombre 2000).

Although each country is unique in its response, when policies are going through an international process, each country's response involves interaction with decision-makers within the international arena (for a full discussion, see Fisher 2004). At least in those democratic countries in which civil society is sufficiently strong, and the nation-state is sufficiently concerned about its own legitimacy, growing environmental awareness and activism can sometimes impinge upon this closed network of regulators and regulated, often eliciting defensive responses from both. (See Box 3.1.) In fact, there is increasing evidence that international environmental pressures can lead nation-states to build environmental capacity, regardless of the level of development in those countries (for example, Frank et al. 2000).

Interest groups in civil society that advocate for ecological responses include a variety of local, national, and international groups. The nature of many ecological concerns, however, often renders political support elusive. Such cases are frequently characterized by a small set of organized, concentrated economic interests opposed to particular ecological protective measures pitted against a very large, disorganized group of supporters (Olson 1965). This situation is especially true of ecological concerns that

BOX 3.1

Political Bargaining over Ecological Responses: ''Job Blackmail''

One defensive response that has been employed by many resource-based companies has been termed ''job blackmail'' (Kazis and Grossman 1991): as resource-based industries face criticism from environmental groups, companies often emphasize the extent to which environmental protection measures have resulted in the loss of jobs. Although such tactics serve to forge an alliance between industry and the local communities from which their labor pool is drawn, others have claimed that job losses are more likely due to the rapid capital intensification of many resource-based industrial processes, and these tactics have only served to place blame on environmentalists, thereby shielding companies from criticism and labor unrest.

are cognitively ambiguous, are not perceived by politically salient actors as directly associated with livelihood, and are not captured readily by the more dominant conduits of such information.

Biodiversity and related "ecosystem services" are often very broad, and in some instances global, public goods. Seppanen and Valiverronen (2000) found that the destruction of biodiversity is an issue that has been difficult to popularize in industrial countries because it lacks a distinctive visual symbol that could encompass the concept. However, where destruction of biodiversity is linked with more immediate livelihood concerns (especially in developing countries), these issues enter the public agenda very rapidly, depending upon the political power of vulnerable communities and their supporters. A case in point is dam building in the Narmada valley in India, where the issues of displacement and ecological destruction came together to create a powerful movement against the dam (Roy 1999).

Perhaps the most important issue that determines the potential for ecological concerns to be placed on the political agenda and the subsequent formulation of policy is the power of the advocacy groups relative to other groups within government, industry, and civil society. As the highest national authority, the state ultimately must take action in the majority of ecological responses. Civil society organizations attempting to promote a response must in most circumstances convince the relevant state actors of the need for such a response. Ecological responses may also be introduced by the state itself. In both instances, given that ecological responses inevitably represent costs to other sectors of industry and/or civil society, the tendency for a state institution to promote this set of interests can be indexed by its *autonomy.* Autonomy is defined as the ability to determine a policy agenda despite external influence. The state may be completely autonomous if opponents of a proposed policy do not have sufficient strength (either in terms of numbers or political clout) to influence the regime. On the other hand, when groups opposed to a proposed response strategy have the power to threaten the regime, the state has no autonomy at all. In most instances, however, the autonomy of the state lies somewhere between these two extremes (Nordlinger 1981; Domhoff 1996). Where a nation-state is placed between such extremes is contingent upon its historical and cultural conditions, as well as the circumstances of a particular response. In the face of very strong opposition, a state institution may need to consider suitable compensation to "buy off" the opposition, or it may need to compromise on the proposed policy.

3.3 Institutional Factors: Capacity for Governance

While a large portion of research on policy effectiveness is focused on the relative power of stakeholders, *capacity* defines another essential element of governance that should not be taken for granted. History is replete with instances of powerful organizations falling short of their objectives due to a lack of capacity, most notably in the international arena. Governance is the sum of the many ways in which individuals and institutions, public and private, manage issues (Commission on Global Governance 1995). Implicit to this definition is the recognition that effective governance depends not on how any one institution performs or how any one set of actors interacts, but on how they perform and interact as a whole. With regard to ecological concerns, governance comprises a whole network of actors, involves a whole range of functions, and is underpinned by certain implicit or explicit principles, norms, rules, and decision-making procedures (Krasner 1983). It is important to note that local communities are important "institutions" to be included in any assessment, since they are implicated in the local implementation of ecological responses in most cases.

Capacity to govern can be defined as the ability of these institutions to execute responses effectively. If there is a high capacity for governance, a response has a better chance of being effective. The degree of effectiveness also depends on factors external to the institution. However, for an institution that lacks the skills, information, and resources necessary for the implementation of a response, outcomes are likely to be disappointing, regardless of the degree of support and enthusiasm expressed. Capacity for governance cannot be viewed as an artifact frozen in time and space, but as a process that changes over time. Institutional actors can learn, make compromises and change, and forge new relationships that can open the doors to additional skills and resources. For responses to be effective, they must be robust enough to adapt to these shifts. (See Box 3.2.)

In short, an assessment of the institutional domain entails an evaluation of the skills and resources possessed by the institutions that will bear responsibility for the implementation of a proposed response, relative to the skills and resources that would be required to implement that response. A gap between what is available and what is required may become a binding constraint and necessitate adjusting the proposed response in light of capacity limitations. Alternatively, the constraint may be overcome through a sustained effort in institutional capacity-building. As responses vary across manifold scales, the capacity to execute responses will depend on the institutions that operate at these scales: international (including regional and sub-regional); national (including provincial or state levels); and local (encompassing both urban and rural contexts) levels.

3.3.1 International Level

The spatial scale of several ecological concerns demands a response at the international level. Although efforts at international governance have multiplied exponentially in the past sixty years, international responses are enormously difficult to achieve, largely because the current international system of governance lacks the degree of stability and order that characterizes systems of governance at national and sub-national levels. The international system is characterized by the struggle for power between states, with a small number of states dominating this struggle (Strange 1983). The influence of certain non-state actors, such as financial

BOX 3.2

Institutional Resilience: The Ability to Adapt

Early international treaties, as well as many domestic policies, were not designed to take on new commitments, nor were they easily amended. As a result, many became stagnant and irrelevant to governments, and/or lost their effectiveness. Modern treaty-making, however, has incorporated a more adaptive approach, recognizing, for instance, that commitments by governments may strengthen when issues become better understood or when shifts in public opinion encourage governments to take action. Modern treaties contain various mechanisms that allow their parties to adapt or learn, or shift with societal norms and values. These include mechanisms such as framework and protocol approaches, learning systems such as education clauses, science and technology mechanisms that review progress in knowledge and advancement on the issue area (Chambers 2003a). Several recent domestic policy efforts have attempted to incorporate such an adaptive approach.

institutions and some nongovernmental organizations, has been increasing dramatically in recent years as well. Various institutions, both formal and informal, are designed to mitigate the influence of these dominant actors, including the formally recognized principle of sovereignty, numerous customary rules, and international treaties, as well as international governmental organizations such as the United Nations. These systems of international governance are often referred to as regimes, which serve as the frameworks through which international actors mediate their behaviors and play out their roles. These actors include states, sub-state actors, epistemic communities, business/industry, and civil society (for a full discussion of the limitations many actors face when trying to participate in international regimes, see Fisher and Green 2004).

While regimes can mediate actors' behaviors, the nature of the inter-state system renders these regimes only partially effective in determining outcomes and regulating the behavior of states and other actors. Several other factors come into play when determining the capacity of governance systems, including compliance; institutional legitimacy; implementation mechanisms; horizontal and vertical interlinkages between institutions; access to financial resources; and institutional adaptability.

Compliance describes the degree to which states follow formal rules and obligations dictated by international law. Though some international law is self-executing, requiring no ratifying legislation, most rules require implementation at the domestic level and thus national policy measures to ensure compliance. Such measures may include financial incentives, legislation, directives, procedures, or sanctions (Brown-Weiss and Jacobson 1998). In straightforward legal obligations, such as submitting progress reports, assessing compliance can be relatively easy. In other cases, however, evidence of compliance can be elusive, and the ability to apply sanctions at the international level can be problematic. *Assessing the potential for and/or evidence of compliance is an essential first step in evaluating effectiveness of responses at the international level.* (See Box 3.3.)

Another important aspect of governance is *legitimacy.* A number of attributes contribute to the perceived legitimacy of international governance regimes. These include the clarity of the rules, their ''symbolic validation'' (the states or entities responsible for creating the rules), their coherence (the interpretation of a rule according to some form of consistency) (Brown-Weiss and Jacobson 1998, p. 136), and their adherence to the existing hierarchy of rules. At the top of the hierarchy is the rule of recognition, which grants each country its sovereignty (Brown-Weiss and Jacobson 1998), and beneath these are ''secondary rules'' that guide making of constitutions, bills of rights, etc. Accordingly, if an international law is in adherence with these secondary rules then there is additional incentive for state compliance (Brown-Weiss and Jacobson 1998, p. 187). Actors are more likely to comply with international laws when they perceive those laws, and/or the institutions sponsoring them, to be legitimate (Franck 1990); hence legitimacy is an important component of any response assessment.

Although compliance is necessary, it is not sufficient to ensure response effectiveness. A particular international response strategy may fail regardless of the extent to which states are in compliance with international obligations. For example, experts agree that the 5.2% reduction of greenhouse gases called for in the Kyoto Protocol will not be enough to stave off climate change, and should only be viewed as a first step. It is also possible that a treaty may unintentionally create incentives to switch to other technologies that also have the potential to damage to the environment, such as increasing the use of nuclear power to reduce air pollutants associated with the burning of hydrocarbon fuels. The extent to which the proposed *implementation mechanism* is appropriate to the stated goals of the response is a key variable that must be assessed to determine whether adequate changes will occur in the behavior of the target group (Raustiala 2000). (See Box 3.4.)

Effective international governance also depends upon the nature of the *interlinkages between international institutions,* as several institutions must inevitably become involved in response formulation and implementation to ensure effectiveness. Unfortunately, international governance regimes are not conducive to the development of coordinated or synergistic approaches to collective environmental problem solving. The complexities of the issues involved, as well as the political nature of policy-making, mean that international responses are often negotiated in relative isolation. Negotiations are often carried out by specialized ministries or functional organizations in forums that are completely detached from the negotiating arena of other international agreements (Chambers 2003b). Even in this isolated context, the consensus building process that is necessary for effective multilateralism is difficult, but with the added burden of accounting for the multiple interrelations across policy domains—such as biodiversity protection and agriculture, for example—effective in-

BOX 3.3
Determinants of Compliance

Compliance may depend on an array of factors that vary from case to case, such as the intrusiveness of the activity; the characteristics of the accord; the negotiating environment; the actors involved; and the depth of the accord, which includes its obligations (binding or hortatory) as well as its precision (Brown-Weiss and Jacobson 1998). Consideration must also be given to the mechanisms for implementation, treatment of non-parties, the existence of free-riders, other countries' approaches to compliance, and the role of international organizations and the media. The ''social, cultural, political and economic'' conditions and how they influence compliance with the accord are also important considerations (Brown-Weiss and Jacobson 1998, p.7).

BOX 3.4
Assessing Implementation

To measure the effectiveness of implementation, most studies have looked at the implementation process both from the international and domestic levels and from the perspective of the state and civil society. These studies generally examine the use of international institutions to review implementation and the ways in which problems are resolved. The focus here is the ''systems of implementation review'' (Victor et al. 1998). This approach looks not only at the legal requirements set out in the agreements, but also at the participation of actors and the system-wide operating environment of the commitment—even in cases where formal procedures do not exist. Some scholars have, however, found that a focus on the establishment or diffusion of institutional forms of environmental protection may actually have little to say about the extent to which such measures or forms ''have, or are likely to have, any definite connections with actual environmental protection outcomes'' (Buttel 2000; see also Fisher and Freudenburg 2005). Several factors influence the effectiveness of implementation of international commitments at the national level; these may include the nature of the problem, configurations of power, institutions, nature of the commitment, linkages with other issues and objectives, exogenous factors, and public concern.

ternational governance is more often than not an elusive goal. Regardless of the difficulties associated with accounting for these interrelations, ignoring them has created global environmental institutions that are ineffective because they attempt to deal with extremely complex, interrelated systems—ecosystems—in piecemeal ways (Chambers 2003b). Therefore, instilling stronger mechanisms that facilitate interlinkages as well as intra-linkages between and across regimes, at the vertical level (ranging from global to local) and at the horizontal level (between regimes at the same level), is one means of improving response effectiveness (for example, Young 1999).

Adequate financing is, without question, one of the key factors for improving the capacity for governance. Financial resources are important for supporting an adequate implementation infrastructure, and for addressing the environmental problem itself. Financing is also important to ensure ratification and compliance on the part of developing countries, as in many instances these countries do not have the resources to meet the obligations. Not only the level of financing, but the institutional skills that are necessary to secure financing, as well as the efficiency of distribution of finances, are all important factors in determining the effectiveness of international responses.

3.3.2 Domestic (National) Level

At the national level, good governance is defined as the manner in which power is exercised in the management of a country's economic and social resources for development (World Bank 1992). Many determinants of the capacity for governance discussed above are equally relevant at the national level, including institutional legitimacy, implementation mechanisms, inter- and intra-linkages throughout the federal and state-provincial governance apparatus, and financial resources. Several additional issues have been highlighted as determinants of the domestic capacity for governance, including the largely administrative elements necessary to implement structural change, as well as political commitment (Leftwich 1994). In particular, effective ecological responses demand the following administrative features: a pluralist polity, in which multiple interests and ideologies can be represented (usually, but not always, through multi-party democratic systems); a clear separation between executive, legislative, and judicial functions that ensures the accountability and transparency of the decision process; and adherence to the rule of law. Furthermore, a committed and efficient public sector is needed that has the capacity to manage reform processes (World Bank 1992; see also Cardoso 2003). Countries lacking one or more of these features generally either are more resistant to the adoption of certain ecological policies and/or exhibit difficulties in their implementation.

In addition to administrative structure, a state's ability to deliver good governance is a function either of its political commitment or of the extent to which it has rationalized environmental concerns into its set of primary goals (Frickel and Davidson 2004). Although administrative structure can be assessed by focusing on the institutions of the state themselves, most decisions regarding ecological response options involve the interrelations among multiple social actors. Therefore, understanding political commitment is more complex, entailing an assessment of the state's relationship with society (for example, Habermas 1975). Particularly when addressing contemporary environmental concerns, a state institution may be dependent upon the expertise and resources available among groups in civil society for effective policy development, a condition known as "Embedded Autonomy" (Evans 1995). When addressing the reauthorization of the Clean Air Act in the United States, for example, the national government recognized that a cap-and-trade system that allowed companies to determine their own emissions policies would be most effective.

Organized nonstate actors that function at the peripheries of bureaucratic state systems can be particularly important sources of capacity (or hindrance) to the pursuit of ecological policies (for example, Adger et al. 2003). These may include, for example, the relationships among local economic interests and local tenure holders and regional regulatory offices (Davidson 2001). Assessment of nation-state capacity to implement ecological responses must include an evaluation of the ability of state institutions to forge facilitative relations with organizations in society that can bring additional resources to bear, while avoiding those relations that pose a hindrance (such as with interests opposed to ecological reform).

3.3.3 Local Level

While there has been extensive work that illustrates the success of community-based efforts in managing and implementing ecological responses, many scholars acknowledge that communities are not simply homogenous groups that work harmoniously to promote group objectives. Communities are more accurately seen as complex and dynamic institutions that are often characterized by internal differences and processes (Leach et al. 1999; Agrawal and Gibson 1999). In short, the debate over the effectiveness of community management has come full circle: from early pessimism about community action as exemplified in the work of Hardin (1968), to a relatively uncritical, and arguably idealistic, view of community-based conservation initiatives through the 1990s (Western and Wright 1994), to a contemporary recognition that community-based management regimes may be appropriate in some circumstances, but not in others (Agrawal 2001).

Scholars have highlighted several conditions upon which the ability of decentralized, locally embedded organizations to manage ecological responses is contingent, based on both field experiences and theoretical development (for example, Wade 1988; Ostrom 1990; Baland and Platteau 1996; Agrawal 2001). The conditions most frequently identified include: (1) perceived local benefits from cooperating; (2) clearly defined rights and boundaries for any natural resources implicated in the response; (3) knowledge about the state of those resources, including for example their extent, accessibility, and potential for regeneration; (4) small size of user groups; (5) low degree of heterogeneity of interests and values within user groups; (6) long-term, multilayered interaction across the communities and other governing institutions involved; (7) simple, unambiguous rules and adaptable management regimes; (8) graduated sanctions as punishment; (9) ease of monitoring and accountability; (10) conflict resolution mechanisms; (11) strong, effective local leadership; and (12) congruence with the wider political economy within which those communities function. These factors refer to characteristics of the resource itself (2 and 3), of the user group (4, 5, and 6) , and of the institutional arrangements for resource management (1, 7, 8, 9, 10, 11, 12). Interestingly, the literature has tended to neglect the role of technological factors while identifying conditions for successful local resource management; in part, this is a reflection of the institutional focus adopted by most authors in this tradition.

Assessments of community-level social capacity can be fairly intensive endeavors, and there are numerous variations within the social sciences regarding which community features should be used as indicators. For the purposes of evaluating the potential for local-level support for, and participation in, ecological responses, the conditions listed above can serve as a general guide, recognizing that some conditions will be more relevant than others in any given context. The characteristics of the natural resources themselves, and to a certain extent those of the user groups, are resistant to change, so it is particularly important to assess the appropriateness of a given response relative to these conditions as they might pose binding constraints. On the other hand, since institutional arrangements at the local level are conducive to change through the use of specific policy measures, intervention to enhance local capacity for governance is usually targeted at these institutional features.

3.4 Economic Factors

The economic effectiveness of responses needs to be considered for both pragmatic and philosophical reasons. From a pragmatic perspective, those who are promoting and financing ecological responses need to be concerned about the costs of the programs and the economic impact of those programs on affected groups. Money is the unit of account in which most governments and donors deal, and a limiting factor in what planners can achieve.

It is telling that one of the seminal treatises in formal economics is titled *The Theory of Value* (Debreu 1954). Unlike some other social scientists, economists are often not hesitant to offer value judgments concerning whether one outcome is ''better'' or ''worse'' than another

For these reasons, economic arguments and principles are often used to motivate responses to ecological degradation and to assess the effectiveness of responses. Individuals conducting assessments should, however, be aware of the strengths and weaknesses of economic tools. Among the strengths is a large body of formal theory demonstrating that social welfare—the ''values'' arising from production, consumption, and preservation—may, under certain conditions, be related to readily observable measures of economic performance: asset prices, incomes, and consumption. Even when these conditions are not met, economic tools may provide a useful, if perhaps not necessarily the only, metric for evaluating responses. Moreover, even if the economic paradigm does not always offer an overall objective with which all observers agree, it may still provide a useful means to an end by suggesting principles to reach any given goal in the most cost-effective manner.

Individuals engaged in assessments should also be aware of the weaknesses of received economic theory. The economist's standard measure of performance is rather narrow. One outcome is better than another, by this standard, if at least one person is made better off without making anyone else worse off. An outcome is *Pareto efficient* or *Pareto optimal* (after the Italian economist and sociologist Wilfredo Pareto [1848–1923]) if there is no better alternative under the no-one-is-made-worse-off criterion. Pareto optimal outcomes are, in general, not unique. Alternative outcomes with very different implications for different individuals may each be Pareto optimal. In short, Pareto optimality is about efficiency, not equity.

Attempts have been made to generalize from Pareto efficiency to a ''Potential Pareto Improvement'' criterion in which everyone could *in theory* be made better off, even if they may not necessarily be compensated in fact. Such approaches underlie benefit-cost analysis. Under a PPI criterion, the policy that maximizes the sum of monetary benefits net of costs across all individuals would be the ''best,'' in as much as a redistribution of gains among individuals could assure that everyone would prefer it. While this prescription is implemented in many analyses, it remains controversial in some quarters, not least because the possibility of hypothetical improvements for everyone in society offers no assurance that everyone in society will actually realize an increase in his/her welfare.

Moreover, technical problems arise in applying the PPI criterion. Kaldor (1939) and Hicks (1939) proposed criteria by which policy changes would be socially preferred if ''winners'' could afford to compensate ''losers'' without the winners sacrificing all their gains (Kaldor) or if ''losers'' could not afford to bribe ''winners'' to forego their gains and still be better off without the policy change (Hicks). Following Scitovsky's (1941) observation that the criteria do not always favor the same policy, the suggestions have often been combined as the ''Kaldor-Hicks criterion'' that winners can compensate losers and losers cannot compensate winners. The PPI criterion has remained problematic, however, as other commentators uncovered other paradoxes and limitations (for example, Gorman 1955; Boadway 1974; Mishan 1981). Problems arise in considering the amount of *money* that would have to change hands in order for all parties to be indemnified fully against a policy that also changes *prices*. This calculation depends on whether it is made under the status quo or after a policy change, and may make PPI criteria difficult to apply in the consideration of major structural changes in economic organization. Of course, many environmental advocates believe that major structural changes will be required to achieve a sustainable future.

While the concept is problematic, justifications may still be offered to continue to base policy advice on PPI criteria. First, at least compared to alternatives that necessarily rely on subjective assessments of the moral worth of different individuals, BCA is relatively easy to apply and, because it respects the actual distribution of income rather than some subjective system of social weights, it is ''objective.'' Second, it might be argued that since most societies have chosen relatively limited reallocations of wealth among their constituents as part of their general taxation and transfer policies, there is limited evidence that such societies really care enough about equity to make it an ancillary goal of their environmental policies. Third, some policy-makers may argue that that the ''winners'' and ''losers'' of policy action are sufficiently similar in either their economic circumstances or moral worth as to justify the working assumption that monetary gains and losses are equally socially valuable for each. Finally, with respect to the more technical problems of making logically consistent comparisons in a shifting landscape of relative prices, many analysts proceed under the often-implicit assumption that they are considering changes of small enough magnitude to obviate such concerns.

None of these arguments is wholly convincing and, indeed, some authors call for the abandonment of the benefit-cost paradigm (for example, Gowdy 2004). Perhaps a more balanced view is to regard BCA as one useful approach among others and to augment its prescriptions with additional criteria to address equity (Little 1950) and, perhaps, sustainability in the face of potentially irreversible ecological losses. Pezzey (1997), for example, argues that society should opt for the ''optimal sustainable'' path should a conflict arise between the objectives of maximizing net benefits and providing for future generations.

While the conceptual foundations underlying BCA remain somewhat unsettled, there are also practical impediments to conducting thorough and reliable BCAs. Economists have developed an elegant conceptual apparatus with which to interpret the prices of goods and services traded in existing markets. They have been less successful in the more difficult task of assigning prices to goods that are not yet transacted in markets.

We might digress to note that conducting a thorough BCA is not always required. In order to preserve an imperiled ecological system or resource, it ought to be sufficient to demonstrate that the benefits of its preservation exceed the costs. In some instances, this might be accomplished by establishing only one or a few of its many ecological values. More generally, however, rational policy requires that ecological assets be preserved when the sum of their values exceeds the costs of their preservation. Restricting analysis to only a subset of values is unlikely to motivate the preservation of all ecological assets that merit it. It should be noted that, to an economist, the "value" of something is not a measure of what it is worth to society to save it in total, but rather a measure of what society might be willing to give up to save a little more of it—the "marginal unit." Economists care about *marginal,* rather than *total,* values. Nevertheless, many environmental and resource economists have taken to the useful abbreviation of discussing the "total economic value" of ecological assets (Pearce and Turner 1990). In TEV, the totaling is done across attributes, rather than across quantities. These attributes may include the simultaneous and mutually compatible uses of, say a forest, for carbon sequestration, for recreation, and in appreciation of the biological diversity it harbors. The value of each attribute is calculated on the margin, but then these marginal values are summed across categories.

A result so fundamental as to be labeled the "First Welfare Theorem" in economics holds that a perfectly competitive economy is Pareto optimal (for example, Varian 1992). A perfectly competitive economy is one in which no actor can influence the prices at which he/she completes transactions, all agents are well-informed, and all commodities are privately owned. Under such circumstances, all goods and services trade at prices that reflect their worth to all members of society. Such circumstances essentially rule out inefficiency by construction: if one person valued something more than another, the two could enter into a mutually beneficial transaction to exchange it (although the formal demonstration of this result in a general setting involves the performance of a great deal of sophisticated mathematics; see, for example, Arrow and Hahn 1971).

The limitation inherent in this result is that public policy is enacted precisely because not all firms are competitive, not everyone is well informed, and not all commodities of interest are privately owned. In the context of ecological systems and services, economists are typically concerned with the second and third elements, particularly the third. The services provided by ecosystems are often "public goods," that is, they support people who bear little if any cost for their preservation yet share in their benefits. Public goods can lead to free riders—individuals who benefit without paying. The problem, in formal terms, is that those who provide ecosystem services by foregoing the destruction of natural ecosystems cannot assert ownership over the services such systems provide. A market economy allocates goods and services efficiently because it facilitates transactions by which one person can pay another to provide a desired good or service. Public goods give rise to a "market failure": there is typically no mechanism by which someone in North America can pay someone in Africa, for example, for the ecosystem services the latter provides the former.

Economists offer a prescription for such market failures by creating or simulating the "missing market" (for example, Kolstad 2000). If there *were* a market in which such transactions might occur, the efficiency properties of an ideal economy would be restored. Another problem arises, however. Just as the hypothetical North American might free-ride on the ecological contributions of the African, one North American might free-ride on the contributions made by another to compensate the African. Public goods are not adequately provided by private markets and must instead be allocated via political decisions and financed from public revenues. The classical argument is presented by Samuelson (1954), although others have argued that "the private provision of public goods" may not always be so inefficient due to altruism or other considerations (for example, Andreoni 1989; Bagnoli and Lipman 1992; and the numerous empirical accounts reported by Ostrom 1990).

Public goods lead to a problem of efficient allocation that does not arise in private transactions. The vaunted "invisible hand" of the market is said to result in the efficient allocation of private goods because each purchaser chooses to buy precisely the amount he wants at the market price, and each provider chooses to sell precisely the amount she wants at that price. There is no comparable mechanism for public goods. There is, however, some dissent on this point. The philosopher Sagoff (1994) suggests that a "missing market" is an economic oxymoron: transactions occur when the benefits of their consummation exceed the costs of their arrangement. But few economists are willing to take so extreme a view of the efficiency of economic arrangements, although some suggest a careful calculation of the administrative costs of allocating public goods and a corresponding constitutional reluctance to appropriate such functions (for example, Stroup 2003). Their point might best be summarized by noting that effort devoted to overseeing the allocation of public goods is itself a public good.

3.4.1 Valuing Benefits

If one accepts the benefit-cost approach to public decision-making—perhaps with some amendments or additions to ensure equity or sustainability—the next order of business is to determine the monetary consequences to be assigned to particular outcomes. Those falling under the rubric of "costs" are often relatively easy to infer. The costs of providing the services afforded by natural ecosystems are represented by the opportunity foregone to preserve them. These can often be inferred from, for example, the price of similar land that has been cleared and used for agriculture or other purposes. The benefits are much more difficult to measure. Some conservation organizations have adopted economic valuation as a response in itself, the idea being that demonstrating high economic values will motivate conservation (for example, Barbier et al. 1997). Yet the methods available for such studies remain imprecise, and, in many instances, spark fierce controversy.

While many economists would acknowledge the possibility that members of society can and do place large values on the preservation of biological diversity and natural habitats, they also despair of being able to measure such values accurately, at least in the near future. To quote a major study conducted by the U.S. National Academy of Sciences, "True public goods, such as biodiversity, species preservation, and national parks, present major conceptual difficulties" (Kokkelenberg and Nordhaus 1999, p. 8). Even when and if techniques can be established for conducting such valuations accurately, the same report states, "The overriding problem with all [valuation approaches] is that they require voluminous data and statistical analysis and can hardly be used routinely" (Kokkelenberg and Nordhaus 1999, p. 125). Perhaps the best hope, if such analyses cannot be "used routinely" is that representative studies can be identified and general results proposed.

Some approaches to ecological valuation have been widely reported. The most frequently cited, a study of global ecosystem service values by Costanza and numerous coauthors (1997) has been widely derided by economists (for example, Smith 1997; Pearce 1998; Toman 1998; Bockstael et al. 2000). Criticisms of the report by Costanza et al. have largely focused on its interpretations of economic concepts. Critics allege that the work confuses average and marginal values and reports an economic impossibility: a willingness to pay in excess of the ability to pay.

Other authors have pursued analyses on more modest geographical scales. Kremen et al. (2000) suggest that the public goods values arising from ecosystem preservation are larger at a global than a local scale, while Balmford et al. (2002) find that some local ecosystems are more valuable for the natural services they provide than they would be if converted to agricultural or residential use. While neither finding is implausible, it remains unclear how broadly representative each may prove to be.

Alternative valuation paradigms have also been suggested. Roughly a generation before the Costanza et al. work, ecologist Odum (1981) proposed that ecosystem services be valued at the cost of the energy embodied in their functions. More recently other researchers have suggested that the ecological costs of intensive urban development be proxied by the size of their ''footprint'' on the landscape, as determined by the area of land required to grow their food, provide their energy, dispose of their wastes, etc. (Rees and Wackernagel 1994). Whatever their other merits might be, such single-metric approaches are typically not accepted by mainstream economists. A celebrated result known as the ''non-substitution theorem'' (Samuelson 1951) establishes that economic values cannot in general be reduced to the contribution of a single primary commodity or input (see Dasgupta 2002 for an argument specifically in the context of environmental valuation). The nonsubstitution theorem is often presented as, and arose as, a refutation of Marx's labor theory of value.

There is a wide range of methods for BCA. Surveys of nonmarket valuation approaches can be found in a number of articles and books; among the most complete and authoritative are Freeman (2002) and the essays assembled by Mäler and Vincent (2004). Economists generally prefer to work with data arising from ''revealed preferences''; that is, evidence arising from the decisions people have actually made when they have to pay for, and live with, their choices. A number of studies have been conducted in which the benefits of environmental improvement have been inferred from, inter alia, the increased harvests of timber, food, or fish afforded by environmental improvement (for example, Bell 1998; Barbier and Strand 1998); provision of goods, services, and amenities such as water (for example, Acharya 2000; Pattanayak and Kramer 2001); demand for recreational opportunities and services (for example, Hausman et al. 1995); the costs averted by the provision of ecosystem services (for example, Shogren and Crocker 1999); and the price premia commanded by land parcels situated in favorable positions vis a vis natural ecosystem services (for example, Irwin 2002; Thorsnes 2002). Sophisticated and difficult analyses are often required to tease out the value of ecosystem services from ''revealed preference'' data. Multiple imputations, each introducing its own statistical uncertainty, may be required to infer the values of goods and services that are not traded in markets from the prices of commodities that are. For an illustration of some of the data requirements and conceptual issues involved in such studies, see Irwin (2002).

While the data requirements and statistical sophistication required to conduct valuation through revealed preference are daunting, the alternative of attempting valuation more directly raises even more contentious issues. In recent decades, many environmental economists have turned to ''stated preference'' methods. This involves asking people what value they place on public goods such as biodiversity and ecosystem services. Current techniques involve variants such as asking how people would vote in a referendum to secure more biodiversity as well as pay higher taxes, or asking people to rank different tax-and-public-goods outcomes. Such an approach may be unavoidable in the calculation of *existence values* (values wholly independent of any present or future use or bequest). Yet stated preference approaches are anathema to many economists, as they do not require respondents to ''put their money where their mouth is'' by *actually* paying for what they profess to value. Regarding the general reliability of statements uttered without economic consequences, see Glaeser (2003). The evidence concerning whether people actually contribute what they claim they will is mixed; compare the contrasting findings of Strand and Seip (1992) with those of Vossler and Kerkvliet (2003). A fierce debate has also raged between those who allege that stated preferences reflect the ''purchase of moral satisfaction'' more than they do any carefully considered estimate of willingness to pay for the specific good in question (Kahneman and Knetsch 1992) and their opponents (for example, Smith 1993). For a perspective from outside economics on the elicitation of preferences, see Fischoff (2004).

Commenting on the use of stated preference methods in economics, V. Kerry Smith (1998) wrote:

> Indeed, there is a curious dichotomy in the research using [stated preference methods] for non-market valuation. Environmental economists actively engaged in non-market valuation continue to pursue very technical implementation or estimation issues, while the economics profession as a whole seems to regard the method as seriously flawed when compared with indirect methods. They would no doubt regard this further technical research as foolish in light of what they judge to be serious problems with the method.

Smith, who has contributed importantly to the literature on stated preference methods, probably overstates the case in characterizing the attitude of the ''economics profession as a whole.'' A panel on which two Nobel Prize winners served provided a cautious but positive endorsement of stated preference methods (Arrow et al. 1993), and one of the journals published by the North American branch of the ''profession as a whole'' has devoted considerable space to a symposium on the subject (*Journal of Economic Perspectives* 1994). Still, stated preference methods, and, indeed, albeit to a lesser degree, nonmarket valuation methods in general, continue to spark controversy. An example of this tension is seen in a comment by Jerry Hausman, the 1985 winner of the American Economic Association's John Bates Clark Medal—awarded every two years to the individual regarded as the most accomplished American economist under 40 and often seen as a predictor of a future Nobel Prize; Hausman remarked that ''Environmental economics is to economics what military music is to music'' (*Business Week,* June 30, 1997). A benefit-cost assessment that relies on nonmarket valuation to establish a case for conservation is unlikely to be regarded as clearly dispositive by all commentators. Nor, it should also be noted, would a finding that conservation is *not* warranted, be based on the same methods.

Another very contentious issue in the calculation of benefits concerns the weighting of benefits received at different times. How should we weigh the interests of unborn generations in making current economic decisions? General practice has been simply to assume that future benefits should be discounted back to the present at a constant exponential rate. This gives rise to compound discounting and, with it, many often-cited anomalies; for example, discounted for 500 years at a rate of 5% per annum, the world's current total economic product would be worth considerably less than most used automobiles (the discounted present value computes to a little less than 500 dollars).

It is interesting to note the comments of some leading early economists on discounting. Ramsey, whose model of economic growth has dominated the analysis of that subject for almost a century, wrote that discounting ''is ethically indefensible and arises merely from the weakness of the imagination'' (1928). Koopmans (1960) provides a simple counter-argument. *Not* discounting, he wrote, might lead to the ''tyranny of the future'': current generations would be obliged to save for the unborn, accumulating capital to be enjoyed by presumably wealthier later generations. In addition to Koopmans' argument, exponential discounting has the desirable property of *time consistency*. One would not be tempted this year to revise savings or investment plans made last year simply because time has passed.

Whether or not later generations will, in fact, be wealthier is both pivotal and unknowable. One view of discounting is simply that a discount rate is a price like any other. Price ratios reflect ratios of marginal satisfactions achieved from consumption. If we choose to discount future relative to present consumption, that choice ought to reflect, at least in part, an expectation that we will not find ourselves in desperate straits in the future. Of course, if we did expect our future prospects to be worse, we would discount less. Moreover, recent research (Weitzman 1998; Gollier 2002; Newell and Pizer 2003) suggests that uncertainty about future production and consumption prospects ought also to motivate lower discounting.

While there are good reasons for discounting, there are also some problems. First, if we are interpreting discount rates as prices, the analysis of the previous paragraph applies only when a single decision-maker is deciding his/her own consumption plan over time. In as much as human lifetimes are limited, long-term decisions based on discounting necessarily involve one generation making a choice that affects others. While the literature is replete with references to the role of bequests from benevolent ancestors to their heirs (for example, Blanchard and Fischer 1989), it is an open question how best to protect the interests of one's progeny. Some researchers favor treating the present generation as agents empowered to save and invest for the future. Others feel the present generation should be required instead to preserve essentially the same configuration of natural assets as we now enjoy for future generations' use and enjoyment. These two views of the current generation's savings obligations represent in broad terms the schools of ''weak'' and ''strong'' sustainability, respectively (for example, Pezzey and Toman 2002). The issue of how best to preserve natural assets for future needs is also closely related to matters of option value, the precautionary principle, and the safe minimum standard (see below).

Some scholars would replace the discounted-present-value of utility formulation that has motivated most models of economic growth with a different criterion for social decision-making. Alternatives include a ''Green Golden Rule'' approach in which the objective would be to maximize sustainable long-term well-being and, implicitly, reduce the importance attached to the welfare of current generations (Beltratti et al. 1995). Graciela Chichilnisky (1996) proposed an objective that would combine traditional discounting with concern for the long term.

While such alternative approaches have merit, it seems reasonable to advise researchers and practitioners assessing response options to adopt a discounting approach, albeit one with conservatively low and, in the limit of the far-distant future, perhaps vanishingly small rates. This will be analytically more straightforward in making calculations. In as much as there are now compelling arguments in the literature for discounting distant-future gains at extremely low rates (Weitzman 1998; Gollier 2002; Newell and Pizer 2003), it seems reasonable to suppose that the apparatus of discounting can now be accepted in very broad terms without necessarily trivializing well-being in the distant future. The suggestion is sometimes made that analysts should—and people do—apply ''hyperbolic'' discount rates (see Heal 2000). With hyperbolic discounting, the net present value of a sum to be received in the future declines as a function of the *relative* time lag involved: the ratio of values assigned to events one and ten years in the future would be the same as that applied to events 10 and 100 years in the future. While survey and psychological evidence may support the approach, it may not matter much in practice if we adopt hyperbolic discounting or simply recognize the effects of manifest uncertainty on calculations concerning the far-distant future.

Finally, with respect to the estimation of benefits, it should be noted that uncertainty regarding the magnitude of benefits should not translate into a disregard for them. In fact, the opposite might be true for two reasons. First, just as the prudent household may purchase insurance against unavoidable risks, prudent resource managers may choose to indemnify themselves against the unknown ecological consequences of radical change by foregoing the associated short-term benefits. Second, even absent aversion to risk, it is generally wise to defer decisions with irreversible consequences until their expected benefits more than exceed their expected costs. The theory of option value, first developed in finance (for example, Dixit and Pindyck 1994), argues that there is a premium, or ''option value,'' associated with retaining the option to wait for better information. This idea has been applied to environmental economics in general (Arrow and Fisher 1974) and the analysis of ecological assets in particular (Perrings and Walker 1997, Albers and Goldbach 2000). Some commentators put forward a stronger argument. The future is not simply unknown; it is unknowable. Hence, rather than assign option values to reflect known uncertainties, society should adopt a safe minimum standard or invoke a precautionary principle (for example, Ready and Bishop 1991). While the safe minimum standard and option value approaches are conceptually distinct, they may yield operationally equivalent policy prescriptions. (See Chapter 4 for a more detailed discussion of decision-making under uncertainty.)

3.4.2 Cost-effectiveness

If it is not possible to infer reasonable estimates of all values, it is not possible to perform a full BCA. In some instances, a partial analysis may suffice. There are circumstances in which measuring some of the benefits of ecosystem services and human well-being establishes that a response should be undertaken. The existence of other, as yet unquantified, values simply makes the case even stronger. Or society may simply have made the decision to allocate a certain sum to conservation or environmental improvement. The challenge then is to determine how to spend this sum most wisely.

In performing such cost-effectiveness analyses, the question is simply ''how much do the relevant alternatives cost?'' Economically relevant costs are *opportunity* costs, rather than financial, historical, or replacement costs. That is, the true measure of the cost of, for example, preserving a parcel of land as habitat for endangered species is the earnings (or more, generally, satisfaction) foregone in not exploiting it for other purposes. It does not matter if these are earnings that a landowner would ''pay to herself'' rather than to someone else if she were to convert the habitat to another use, nor would opportunity costs be obviated if a government were to condemn the land for public use rather than compensate the landowner for her lost earnings.

The economist's prescription for implementing environmental or conservation policy at least cost to society is typically to use ''market based incentives'' such as taxes on environmentally harmful products, subsidies to environmentally friendly ones, or a system of tradable permits or obligations in such products. The approach was first proposed by the British economist Pigou (1932)—hence the designation ''Pigovian taxes'' or ''Pigovian subsidies''—and has been a staple of textbooks since, although the first significant policy applications did not materialize for many years. Subsequent research has noted the need for refinements to, for example, fine-tune taxes or permit prices to reflect spatial variation (Tietenberg 1978) or compensate for interaction with existing taxes on other commodities (Parry et al. 1997). However, the principal practical impediments to implementing cost-effective market-based incentives would appear to be largely political (Pearce 2004) and technological (Henriquez 2004). The latter barriers may be removed as capacity is enhanced to track environmental damage in real time and trace specific harms to their originators.

The question of cost-effectiveness has proved particularly contentious in international biodiversity conservation policy. Several authors have suggested that ''direct payments'' will prove more effective than ''indirect approaches'' such as those embodied in Integrated Conservation and Development Programs (Simpson and Sedjo 1996; Ferraro 2001; Ferraro and Kiss 2002). Direct payments are, in essence, Pigovian subsidies (Ferraro and Simpson 2002). Such approaches are being increasingly adopted by organizations such as Conservation International (CI 2000), and various nations and organizations are experimenting with payments for ecosystem services (Pagiola et al. 2002). These steps are being taken in large part in response to perceptions that ''indirect approaches'' have not proved cost-effective (Wells and Brandon 1992; World Bank 1997; Brandon 1998; Reid and Rice 1997; Southgate 1998; Oates 1999; Terborgh 1999; Terborgh et al. 2002). Others have pilloried such criticisms as attempts to ''reinvent a square wheel'' (Brechin et al. 2002) and might well regard economic injunctions to adopt direct incentives as similarly insensitive to cultural context.

In assessing the cost-effectiveness of conservation policies it seems reasonable to suggest that researchers be able to trace a clear and unambiguous trail from the response proposed to the conservation (and, as appropriate, development) outcome desired. Some arguments for direct incentives are stripped to their bare conceptual basics and, consequently, may ignore social and cultural factors. Yet conservation practitioners and project evaluators should be skeptical of broad interventions intended to solve multiple problems with single programs. Brandon (1998) comments trenchantly on the limitations of conservation interventions in addressing broader social ills.

3.4.3 Secure Property Rights

An important assumption underlying the claim that a competitive market economy is Pareto optimal is that property rights are well-defined. That is, someone owns each of the goods and services that people might trade with one another. The owner cannot be deprived of what he/she owns without agreeing to be compensated with something else. When it is impossible or infeasible to exclude other users, there is little incentive for one person to conserve resources for her future use, as it is likely that another will capture them in the interim. Hence, many commentators note that strengthening tenure—in essence, designating ''owners'' and empowering them to exclude others—will induce more conservation (for example, Scher et al. 2002; Terborgh et al. 2002).

There is, perhaps, more agreement among conservation practitioners and advisors on the importance of property rights than there is on the matter of how best to compensate owners for conservation. This could be troubling, as property rights alone may not suffice to motivate conservation. There are both costs and benefits to property rights. The benefits arise from security in investment: maintenance or improvement of property is repaid with future earnings. The costs arise from enforcement: property must be defended against those who would infringe upon it. The economic theory of property rights (Barzel 1997) suggests that property rights come to be defined when the benefits of their definition justify the costs of their enforcement.

Some commentators suggest that biodiversity might be protected better if property rights were more clearly established in its products. Establishing and strengthening property rights is, then, a potential response. This observation raises a question of cause and effect. If property rights come into being when the benefits of their establishment justify them, are the potential owners of biodiversity deprived of benefits because they cannot claim property rights, or do they not claim property rights because the potential benefits of doing so have not yet proved sufficient to compensate them? To give one example, some have argued that local people did not have sufficient incentives to conserve biodiversity because, at least absent participation in and protection from the Convention on Biological Diversity, they could not claim ownership of pharmaceutical preparations derived from natural products (for example, Vogel 1994). Others argue that property rights in such preparations have generally remained unspecified because the economic value of untested natural products is not high enough to justify their establishment (Simpson et al. 1996).

In general, practitioners and evaluators assessing the role of property rights in crafting effective responses should consider carefully the function to be served by stronger property rights. If it is to capitalize on as-yet unrealized values, it is reasonable to ask what impediments have prevented their realization to date. If it is to facilitate the receipt of payments for conservation, the source of payments must, of course, also be secured.

None of this discussion is intended to suggest that an adequate source of income in combination with secure ownership would *not* motivate effective conservation incentives. Geoffrey Heal (2000, 2002) has suggested that private and public goods might be, and in fact in many instances are being, ''bundled'' so as to make payments for the former and as an incentive for maintaining the latter. In southern Africa, for example, private farmers have found it more profitable to devote their lands to game animals than to raise cattle and other non-indigenous species (Bond 1993; Heal 2000; Muir-Leresche and Nelson 2000). By selling the right to enter, photograph, or hunt (in a regulated and sustainable fashion), such game ranchers are earning private returns while protecting a public good. Similar considerations may motivate private conservation of forest habitat in Costa Rica (Langholz et al. 2000) or the incorporation of onsite nature reserves at new housing developments (Heal 2000, 2002). It is certainly appropriate to consider roles for private markets in the economic assessment of responses.

Unfortunately, there are no simple policy prescriptions regarding the optimal or ideal system of property rights to protect and maintain ecological systems. A detailed review of the empirical literature on different property rights regimes concluded, ''Success in the regulation of uses and users is not associated with any particular type of property-rights regime'' (Feeny et al. 1990, p.12). An important implication of such work is to demonstrate that there are a variety of alternative institutional contexts under which ecosystems and their services may be appropriately managed. The case for secure property rights, therefore, should not be seen as a case that supports any *specific* system of property rights, such as privatization or nationalization.

As a final comment on property rights, we note again some themes already touched upon. Theoretical models of general competitive equilibrium often assume ownership of *everything* at every *time* and in every conceivable *state of nature* (more formally and compactly, ''complete futures and contingency markets''). This is obviously impossible: future consumers cannot conduct transactions before they are born. Those alive at present might represent the interests of their descendents, and markets may reflect concern for future resource scarcity. However, different commentators differ as to whether such considerations constitute adequate safeguards for intergenerational equity.

Some similar issues may also arise in the consideration of intragenerational consequences of environmental and conservation policy. Instituting a widespread program of payments for the preservation of biodiversity might be expected to occasion a large transfer of wealth from richer to poorer nations as the relatively undisturbed land holdings of the latter increase in value. By the same token, a policy that obliged poor countries to conserve natural capital without corresponding compensation would exacerbate existing inequities. Researchers assessing the social consequences of responses should consider these factors, and note that changes in the distribution of wealth could have more profound impacts.

3.5 Social Factors

As discussed, stakeholders are those individuals or organizations who wish to affect a response, those who are able to affect it, and those who are affected by it. Groups characterized chiefly by the last criterion may not necessarily be actively engaged in the political process, but may nonetheless face direct or indirect impacts as a result of the implementation of a particular response. Any attempt to develop a strategic response to ecosystem maintenance will have implications for social systems beyond what can be considered strictly economic and political parameters. This section discusses a set of issues that are conventionally classified as relating to the equity of responses; it also introduces the idea of pluralistic belief systems as a potential source of inequity. A distinction is drawn between material inequity and cultural inequity. The latter relates to the ways in which different cultural groups, who perceive the environment and their relationship to nature in culturally distinctive ways, may also be impinged upon, with significant social impacts.

The distribution of gains and losses across stakeholders rarely occurs in a manner that is perceived as equitable by all. Devising and implementing ecological responses always entails gains and losses and, in many instances, these impacts have been distributed in markedly inequitable fashion, regardless of whether this outcome was intended by policy proponents. Equity issues emerge because of distributional impacts that affect stakeholders who can be distinguished along a number of dimensions (some of which may potentially be overlapping). These include gender, ethnicity, race, class, and generation. Some equity concerns may also arise from the pragmatic concern that power be given to, or retained by, those with the material wherewithal to ''get the job done.'' In many contexts this is not solely an economic consideration, but may also involve ethnicity, gender, class, and other attributes. Equity concerns also emerge because groups may have beliefs and ideological opinions that are inadequately represented when choosing among responses and strategies. Although effectiveness is sometimes seen as a benefit to be traded off in terms of a reduction in equity, in some instances the inequitable social impacts of a response have the potential to impinge upon the effectiveness of the response itself.

3.5.1 Equity

Many stakeholders have divergent *material* interests, and these are differentially affected by policies that are targeted at meeting defined ecological objectives. While the economic paradigm treats such issues as distributional (and thus not relevant to the question of efficient resource allocation and management), there are usually profound implications for groups that suffer costs as a result of adoption of specific responses. Furthermore, if such groups are socially or politically marginalized, these costs are often not taken into account in determining the desirability of a response, and the groups are not compensated in any way.

Well-developed response strategies should ideally contribute to the reduction of material inequity between social groups. In many cases, however, a complete assessment of potential equity concerns is not carried out before implementing a policy. In some cases in the past, such policies have exacerbated existing social inequities, or have inadvertently caused new ones, by imposing changes in rights/access to resources or by providing (unintentionally or intentionally) compensation to certain groups. (See Box 3.5.) In many cases, there is a tension between the pursuit of ecological goals and social equity for one or more of three reasons. First, ecological goods are not uniformly distributed. Second, the pursuit of these goals more often than not imposes a greater degree of behavioral changes onto specific groups. Third, the ability to guarantee all individuals access to the necessary resources to support even a minimally sufficient standard of living, much less raise those standards to the quality of life enjoyed by those in industrial countries, simply not be physically possible at current global population levels, particularly in regions facing high population growth within ecosystems that are already strained beyond capacity.

3.5.2 Cognitive Differences and Ecological Beliefs

Social differences may emerge at the cognitive level, because of divergent perspectives, ideologies, or worldviews held by various

BOX 3.5

The Equity Implications of Forced Displacement

A particular issue in the discussion of equity relates to the forced movement of people and the displacement of existing livelihood strategies as a consequence of adopting a particular response. Any strategy that imposes constraints on the use of land and resources may impinge upon current lifestyles and livelihood strategies of a local population. This problem is particularly contentious in the conservation context, since the creation of areas that are reserved for the maintenance of habitats for biodiversity often impinges on existing land-uses. Such strategies may directly or indirectly encourage resettlement that, while relieving pressure on a particular conservation area, may only shift those pressures to other regions that may be equally stressed, or lead to decreased living standards when resettlement is not an option. In addition, population pressures (the need for living space, conversion of land to agriculture, and fuelwood collection) also present an imposing obstacle to the implementation of response strategies. This issue applies not only to developing countries, but has also become relevant in the industrial world, where absolute population pressures may not be an issue but the rapid growth of concentrated population centers in ecologically sensitive zones can place limits on the ability to exert control over ecosystem services.

stakeholders. Responses may be contested, not because of the perceived loss of material well-being, but because they imply a rejection of deeply held beliefs. In order to assess the social impacts of a particular response, policy-makers must recognize the existence of a plurality of worldviews and acknowledge that the adoption of a specific policy response may (inadvertently or deliberately) privilege specific worldviews. Thus, just as the material interests of stakeholders may be differentially affected by policy, it is also possible that the implementation of policy responses that privilege specific worldviews may elicit enormous resentment among cultural groups that are affected by the policy, which can hinder policy effectiveness, encourage social conflict, or reduce the potential for political viability of future responses. One of the most significant impacts of such oversight may be a failure to incorporate systems of knowledge and worldviews that may well be the most appropriate for the ecological goals sought.

Typically the worldviews that tend to be accorded privilege in ecological policy encompass an emphasis on certain environmental issues identified by predominantly North American and European environmental interests, and a reliance on Cartesian scientific methods and technological inputs. The issues identified by these interests may not be the priorities of peoples in other countries. (See Box 3.6.) The worldviews that become marginalized in the policy-making process include those of alternative environmental priorities, local knowledge, and ecological perspectives of peoples living in regions that are targeted, or at least implicated, by response strategies. This situation may not only lead to the undermining of important information and monitoring abilities, but can also lead to resentment among the affected groups. One such example can be seen in the case of whale hunting among the Makah people in Neah Bay, Washington, which is seen as a ''cultural necessity'' (Erikson 1999). This practice is bound up with traditions, rituals, and taboos integral to the tribe's cultural survival. In another example, some scholars have suggested that granting trade-related intellectual property rights to new plant and animal products is not only in conflict with the goals of the Convention of Biological Diversity, but may also undermine the lifestyles of indigenous peoples (Anuradha 2001).

BOX 3.6

The Post-materialism Hypothesis

Recent scholarship has debated the differences between ecological beliefs among the populations of advanced industrialized countries and those that are relatively poor. Many researchers use the notion of ''post-materialism'' to study the attitudes and role of civil society in industrialized countries (for example, Abramson 1997; Abramson and Inglehart 1995; Kidd and Lee 1997a, 1997b; Pierce 1997).

Abramson and Inglehart summarize the post-materialism thesis: it ''assumes that the economic security created by advanced industrial societies gradually changes the goal orientations of mass publics'' (p. 9). This is similar to the conventional wisdom among economists that environmental protection is a ''luxury good'' demanded in greater quantity by the wealthy (for example, Kolstad 2000; also Dasgupta 2003, who argues that the poor find themselves more dependant upon the services provided by natural environments). In general, scholars find that people with post-materialist values are more apt to prioritize environmental protection. Rather than focusing on social movement organizations and civic associations, the research on post-materialism looks at lifestyle issues and consumer behavior and how they are related to environmental protection.

However, the post-materialism thesis has come under criticism by scholars looking at global environmentalism (for example, Brechin 1999; Brechin and Kempton 1994; Dunlap and Mertig 1992; Guha and Martinez-Alier 1997; Adeola 1998). These scholars suggest that there are material and non-material attitudes that inform environmental beliefs in both rich and poor countries, and argue that it is inappropriate to assume that ecological issues will necessarily be a higher priority in advanced industrial nations.

3.6 Methods and Tools for Assessing Responses

For effective decision-making, it is necessary to examine the implications of chosen responses across the five domains: political, institutional, economic, social, and ecological. Failure to assess options across all these dimensions may cause several weaknesses, and strengths, to be overlooked. The material presented here has highlighted issues that emerge in assessing responses *within* any one of these dimensions, but an overall assessment must find ways of comparing outcomes *across* these different domains as well. The task of an integrated assessment is particularly difficult, and demands extensive resources, since it needs to recognize the multidimensional nature of impacts, but also requires methods that are sensitive to a plurality of perspectives from diverse intellectual disciplines. Further, decision-makers may differ on the relative desirability of response strategies, particularly when there are legitimate differences of opinion about both the objectives of intervention and the means to achieve these ends.

Along any one dimension, using any particular criterion for assessment, the evaluation process can distinguish between constraints that render a policy option infeasible, that is, the *binding* constraints, and those considerations that, although important, may be treated as costs associated with the implementation of an option that stakeholders might be willing to bear, that is, the *acceptable trade-offs*. The distinction is important, since classifying an impact as a binding constraint effectively rules out a particular response, while identifying acceptable trade-offs serves to alert a decision-maker to unintended consequences and potentially harmful side effects of an otherwise feasible strategy. (As noted earlier, the obvious caveat is that what is seen by a particular decision-maker as a binding constraint in a particular context may not be seen in the same light by other decision-makers.)

Recognizing that decisions are typically made in a pluralistic environment, this section presents some general principles that can guide decision-makers in assessing the effectiveness of responses. The principles that are outlined here simply follow a systematic thought process that makes explicit some of the trade-offs and choices that are inevitably involved in such decision-making. What this discussion emphasizes is that no single discipline, and indeed no particular perspective, can claim the greatest legitimacy in a debate. What is appropriate and desirable in one context may be completely unacceptable in another. To this extent, the effectiveness of responses is relevant to a specific scale of analysis, and to a particular spatial and temporal context.

Although we offer a comprehensive guide for response assessment, we recognize that in some instances decision-makers will not have sufficient time or resources to collect all the information that is necessary, particularly in instances where indicators are not readily available, and primary data must be gathered. Fieldwork in the social sciences can be costly as well as time-consuming, as it may involve a combination of historical research, participant observation, and personal interviews and/or surveys. In cases where resources fall short of what may be required, decision-makers have a number of options. They may explore cost-sharing options with other potential supporters, or garner local community participation in data gathering, for example. If decision-makers cannot raise enough resources to conduct a comprehensive assessment, we advise primary focus on potentially binding constraints, and the formulation of response strategies that are sufficiently flexible that they can be responsive to dynamic and uncertain conditions.

The framework involves three stages: identifying binding constraints, classifying acceptable trade-offs, and scoring potential responses against one another to choose the best option. The questions involved in the first stage of the assessment process—identifying binding constraints—are shown in Table 3.1. It provides examples of the types of issues that would allow decision-makers to identify whether processes in any one of the political, institutional, economic, or social domains could potentially be regarded as binding constraints in the context of specific responses. Clearly, if this is the case, the response itself is unlikely to be accepted, and the decision-making process must attempt to identify suitable and feasible alternatives. This procedure can be seen as the first filter that eliminates potential strategies that either fail to achieve important objectives, or are technically, morally, or politically unacceptable.

In the second stage of the assessment method, trade-offs can often be identified quickly. For example, autocratic responses may economize on public consultation that might dilute the impact of proposed responses targeted at specific, one-dimensional change.

Table 3.1. A Framework for Assessing Responses

Domain	Issue	Evaluation
Political	Can all potential stakeholders be identified?	Not likely to be a binding constraint, unless neglected stakeholders mobilize political opposition.
	Is the political context supportive?	If not, could be a binding constraint.
	Can the political context be changed?	If not, is a binding constraint.
Institutional	Is there adequate capacity for governance at an appropriate scale?	If not, could be a binding constraint.
	If not, can it be created?	If not, is a binding constraint.
Economic	Is the outcome cost-effective?	Could be a binding constraint if funds are limited.
	Are there secure and well-defined property rights?	Could be a binding constraint, if there are numerous competing demands on the resource.
Social	Is the outcome equitable, in a distributional/material sense?	Could be a binding constraint, if this is a high priority, or those disadvantaged by the response can effectively oppose it.
	Does the outcome violate the cultural norms of particular groups?	Not likely to be a binding constraint, unless consensus is an explicit objective.

These same trade-offs, however, are also likely to undercut the political feasibility and, in consequence, the equity of such approaches. Conversely, responses developed with the participation of numerous special interest groups may assemble the coalition of interests required for their implementation, but have limited impact due to the need to serve the disparate interests of the coalition.

This is not to say, of course, that synergies never arise. Some responses may constitute ''win-win'' opportunities in which, perhaps, native ecosystems might be maintained while enhancing local incomes, redressing inequities, or achieving other ends. Many examples have been proposed of such ''win-win'' opportunities (for example, Heal 2000; Daily and Ellison 2002). Changes in technology, consumer tastes, political liberty, and other factors may be expected to generate more such opportunities over time. While policy-makers ought to remain alert for such opportunities and move aggressively to act upon them, they must also retain a realistic outlook. One must always ask why an activity that affords multiple benefits has not yet been initiated. Closer inspection may reveal that there are, in fact, other stakeholders whose interests are imperiled by what may first appear to be a ''win-win'' opportunity. In other words, if it sounds too good to be true, it probably is (O'Brien and Leichenko 2003).

It seems unlikely, then, that instances could often be identified in which one response dominates all others in all categories (and, we might add, in the estimation of all stakeholders). Thus, even once unacceptable responses have been eliminated, the decision-maker will generally need to make further choices between a set of potentially feasible responses, each of which may have different specific implications for any one of these multiple domains. In other words, the third stage in the assessment process is to score potential responses against one another. Table 3.2 presents the Response Assessment Matrix, which provides a method for decision-makers to assess responses across the five domains identified in this chapter.

In order to make this matrix operational, sub-criteria must be developed within each of the five domains. These would then be associated with specific indicators that would help to assess the impact of the response. No general listing of such indicators is possible, however, since these are likely to reflect the specific resource and the particular local context, as well as the preferences of the decision-makers engaged in the evaluation process. In some contexts, such as the development of criteria and indicators for sustainable forest management, there is considerable progress toward developing a set of widely acceptable and measurable criteria and indicators (see Prabhu et al. 1999). In other areas, however, progress is much slower. Moreover, there are a number of issues for which readily accessible indicators either do not exist at present or their validity is disputed. Table 3.3 summarizes the main issues that relate to the assessment process and comments on the availability and acceptability of relevant indicators for each of these issues.

Table 3.2. Proposed Response Assessment Matrix

	Political	Institutional	Economic	Social	Ecological
Response 1					
Response 2					
Response 3					
. . . and so on					

Table 3.3. Indicators for the Impact of Responses

Domain	Issue	Availability of Indicators
Political	Identification of relevant stakeholders	Stakeholder analysis
	Balance of power between stakeholders	No indicators—subjective judgment
	Degree of policy support by stakeholders	No indicators—subjective judgment
	Relations between the state and key stakeholders—relative autonomy	No indicators—subjective judgment
Institutional	Legitimacy of international institutions	No indicators—subjective judgment
	Linkages between institutions	No indicators—subjective judgment
	Compliance with established rules and norms	Progress reports, voluntary submissions
	Implementation	Progress reports, voluntary submissions
	Institutional adaptability	No indicators—subjective judgment based on longevity and resilience of institutions
	Access to financial resources	Budget analysis
	Existence of pluralist democracy	Electoral participation rates, number of political parties
	Separation of executive, judicial, and legislative functions	Constitutional provisions
	Bureaucratic competence	No indicators—subjective judgment
	Capacity for local management	Presence of organized user groups
Economic	Cost-effectiveness	Benefit cost analysis, valuation studies
	Security of property rights	Legal framework
Social	Equity (material)	Distributional analysis of costs and benefits
	Conflicting worldviews and ideologies	Anthropological and cultural studies

Assessment of trade-offs can be qualitative, quantitative, or both, as shown in the examples in Tables 3.4 and 3.5. Resolving the trade-offs presents the greatest challenge to determining appropriate responses. In some instances, it may be possible to view indicators as binary: so long as some standard is satisfied, the decision between approaches can be made on other grounds. For example, a decision-maker might determine that, so long as local material inequities are not further exacerbated, the most cost-effective approach might be chosen. More generally, however, aggregating across these different dimensions is likely to be difficult. But it is not impossible, as is demonstrated by Brown et al. (2002), who use a matrix scoring method to reflect economic, social, and ecological considerations in the context of integrated coastal conservation and development. They involved a wide range of stakeholders in developing specific indicators across each

Table 3.4. Qualitative Assessment of Trade-offs and Synergies

	Political	Institutional	Economic	Social	Ecological
Response 1	+ +	+	–	+	+
Response 2	–	+ +	+ +	–	+
Response 3	+	–	+	–	–
. . . and so on					

of the domains; these were then converted into standardized scores that were aggregated using different weights in order to rank the alternatives. It is not clear that such methods are appropriate for all qualitative indicators, however, and their results are always highly sensitive to the weighting procedure that is adopted. Weights will also necessarily change over time, as circumstances change.

This is not a counsel of despair by any means, however. We suggest only that decision-makers must, in the final analysis, make some subjective assessment of the weights to be assigned to each factor. Having done so, the construction of such matrices will help decision-makers focus on the important issues that emerge in choosing among alternatives. This process will also aid decision-makers in constructing alternative future scenarios, thereby making better-informed choices among the alternatives.

Although the assessment process outlined here suggests a rational, linear approach to evaluating options, it is important to emphasize that preferences, perceptions, information, and exogenous factors continually change, and an effective system must be responsive and flexible to such changes. The process of decision-making is an on-going one, not a one-time assessment of trade-offs and synergies between different objectives and the interests of different stakeholders. Thus the assessment of responses needs to accommodate feedback, as well as the possibility of learning and adaptation. There is a growing emphasis on the use of methods such as ''adaptive management'' and ''adaptive co-management'' to incorporate such dynamism into decision-making environments. (See Box 3.7.)

Such assessment processes can be undertaken by a number of diverse stakeholders at a variety of scales. In such circumstances, the use of decision-making methods that adopt a pluralistic perspective is particularly pertinent, since these techniques do not privilege any particular viewpoint. Box 3.8 presents an example of such a pluralistic decision making process, which has achieved significant progress towards sustainable adaptive forest management in Canada.

A number of pluralistic decision-making tools and techniques have been documented in the literature. These tools can be employed at a variety of scales, including global, sub-global, and local. Tables 3.6, 3.7, and 3.8 list some of the tools that are available, as well as a preliminary assessment of their most appropriate scale(s) of application. Although not necessarily complete, as new techniques and tools are constantly being developed, the *typology*

Table 3.5. Quantitative Assessment of Trade-offs and Synergies

	Political	Institutional	Economic	Social	Ecological
Response 1	4	2	–1	1	1
Response 2	–2	3	3	–2	2
Response 3	1	–1	2	–1	–2
. . . and so on					

BOX 3.7

Adaptive Management and Adaptive Co-management

Adaptive management (Holling 1978) draws upon a variety of techniques for the management of ecological resources that emphasize the wider system as an appropriate unit for analysis. These techniques are particularly sensitive to dynamism and feedback; they are holistic and process-oriented. Adaptive management recognizes the role of uncertainty in the decision-making environment, but uses adaptation (in the Darwinian sense) to allow responses to learn from these environmental variables, and thereby to produce more stable policy outcomes. Such systems are modeled across a variety of temporal and spatial scales, and usually involve the coupling of highly complex social and ecological systems (Berkes and Folke 1998; Berkes et al. 2003).

Adaptive co-management builds on these principles by emphasizing the importance of local ecological and social contexts within which decision-making operates. Ecological processes at the local scale display a particularity and variation that have an impact on the ways in which responses work in practice; incorporating such place-specific knowledge is an important element of an adaptive management strategy. Sensitivity to the social context demands the recognition of local communities as critical elements of any response strategy. Adaptive co-management systems have been described as "flexible, community-based systems of resource management tailored to specific places and situations and supported by, and working with, various organizations at different levels" (Olsson et al. 2004, p. 75). Responses developed through such processes emphasize the central role of learning, adaptation, and collaboration in the evolution of strategies, in an attempt to build more resilient social-ecological systems.

Assessments of empirical examples of such systems are increasingly seeking to understand why they are especially appropriate to the choice, implementation, and evaluation of responses in a dynamic world characterized by uncertainty and change (Olsson et al. 2004, for instance, evaluate two cases from Lake Racken in western Sweden and James Bay in Canada in order to identify the essential features of the adaptive co-management process).

is reasonably comprehensive. Distinctions are made between *deliberative* tools (Table 3.6), which facilitate transparency and stakeholder dialogue over responses; *information gathering* tools (Table 3.7), which are primarily focused on collecting data and opinions; and *planning* tools (Table 3.8), which are typically employed for the evaluation of potential policy options. Chapter 4 presents a more comprehensive overview of the specific issues that arise due to the presence of uncertainty in the decision-making environment, and reviews specific decision analytical frameworks that have been adopted in order to deal with such uncertainty.

3.7 Conclusion

Chapter 3 has attempted to outline a general guide to be used for assessing ecological responses through the identification of the impacts of responses across four domains: the political, institutional, economic, and social. Tools for assessing a fifth critical domain, the ecological, can be found in other chapters. These impacts may pose binding constraints to response efforts, or they may be considered acceptable trade-offs. We have consequently developed a three-stage assessment process, involving the identification of binding constraints, comparison of trade-offs, and the selection of a response that avoids binding constraints and minimizes negative trade-offs.

BOX 3.8

Canada's Model Forest Program: Integrating Multiple Dimensions into Responses

In an effort to develop innovative ideas and methods to promote more sustainable, adaptive forest management in Canada in a manner that ensures consideration of sociocultural and economic factors in any forest management response option, the federal government established Canada's Model Forest Program in 1992.

The Program now encompasses eleven model forests across Canada, selected to represent the diversity of ecosystems and social systems that characterize the Canadian forest milieu. Each model forest is designed to function as a living laboratory in which new, integrated forest management techniques are researched, developed, applied, and monitored in a transparent forum characterized by partnerships with stakeholders, including representatives from environmental organizations, native groups, industry, educational and research institutions, all levels of government, community-based associations, recreationists, and landowners, all of whom assist in the development of a research agenda, and participate in the research process.

Some accomplishments include: development of a voluntary wetland conservation program for private lands; establishment of protocols for reporting on socioeconomic indicators based on Statistics Canada census data; an ecosystem-based Integrated Resource Management Plan now being used by the province of Saskatchewan; production of a code of forestry practice booklet to help landowners understand and apply the principles of sustainable forest management; establishment of the Grand Lake Reserve to protect three eco-regions and habitat for the endangered Newfoundland pine marten. For more information visit the Model Forest Program Web site: http://www.nrcan.gc.ca/cfs-scf/national/what-quoi/modelforest_e.html.

Table 3.6. Deliberative Tools

	Scale of Application			
Tool	Global	Sub-global	Local	References
Area/neighborhood forums			√	Lowndes et al. 1998
Citizens' juries		√	√	Lowndes et al. 1998
Citizens' interactive panels		√	√	Richardson 1998; IPPR 1999
Community issues groups			√	Clarke 1998
Consensus conferences	√	√	√	IPPR 1999
Electronic democracy	√	√	√	IPPR 1999
Focus groups			√	Lowndes et al. 1998
Issue forums			√	Lowndes et al. 1998
Service user forums			√	Lowndes et al. 1998

Response assessments can be complex and costly endeavors, but their utility can be measured in the increased probability of response effectiveness. While in the past, assessments have tended to be undertaken from a disciplinary perspective, we emphasize the importance of maintaining an interdisciplinary approach to assessment to ensure that important impacts or assessment meth-

Table 3.7. Information Gathering Tools

	Scale of Application			
Tool	Global	Sub-global	Local	References
Citizens' research panels		√	√	IPPR 1999
Deliberative opinion poll		√	√	IPPR 1999
Environmental impact assessment		√	√	Taylor 1984
Participatory rural appraisal			√	Chambers 1983, 1997
Rapid rural appraisal			√	Chambers 1997

ods are not overlooked. Any evaluation of the human dimensions of ecological responses, however, will inevitably be characterized by subjectivity and difficulties associated with attempting to reconcile heterogeneous assessment methods.

No matter how comprehensive an assessment, decision-makers must be prepared for the likelihood that consensus is not always reached among all stakeholders involved in a particular response. Among the most important steps that should be taken to limit potential conflict are emphasizing an inclusive evaluation processes, so that assessment is not undertaken by elite decision-makers; maintaining transparency and accountability throughout the assessment process; and, ultimately, developing responses that are flexible enough to maintain effectiveness despite dynamic human conditions. In the end, however, consensus building may be difficult, but with work, it is likely to be possible.

Table 3.8. Planning Tools

	Scale of Application			
Tool	Global	Sub-global	Local	References
Consensus participation	√	√	√	Warner 1997
Cost-benefit analysis		√	√	Hanley and Spash 1993
Future search conferences		√	√	IPPR 1999
Innovative development			√	Del Valle 1999
Issue forums		√	√	Lowndes et al. 1998
Multicriteria analysis	√	√	√	Stirling and Maher 1999
Participatory learning and action			√	Guijt 1998; Holland 1998
Planning for real			√	IPPR 1999
Service user forums			√	Lowndes et al. 1998
Stakeholder decision analysis	√	√	√	Grimble et al. 1995; ESRC 1998
Trade-off analysis	√	√	√	Brown et al. 2002
Visioning exercises		√	√	Lowndes et al. 1998

References

Abramson, P.R., 1997: Postmaterialism and environmentalism: A comment on an analysis and a reappraisal, *Social Science Quarterly,* **78(1),** pp. 21–23.

Abramson, P.R. and R. Inglehart, 1995: *Value Change in Global Perspective,* University of Michigan Press, Ann Arbor, MI.

Acharya, G., 2000: Approaches to valuing the hidden ecological services of wetland ecosystems, *Ecological Economics,* **35,** pp. 63–74.

Adger, W.N., K. Brown, J. Fairbrass, A. Jordan, J. Paavola, et al., 2003: Governance for sustainability: towards a ''thick'' analysis of environmental decision-making, *Environment and Planning A*. In press.

Adeola, F., 1998: Cross-national environmentalism differentials: Evidence from core and noncore nations, *Society and Natural Resources,* **11,** pp. 339–364.

Agrawal, A. and C.C. Gibson, 1999: Enchantment and disenchantment: The role of community in natural resource conservation, *World Development,* **27(4),** pp. 629–649.

Agrawal, A., 2001: Common property institutions and sustainable governance of resources, *World Development,* 29, pp. 1649–1672.

Albers, H.J. and M.J. Goldbach, 2000: Irreversible ecosystem change, species competition, and shifting cultivation, *Resource and Energy Economics,* **22,** pp. 261–280.

Andreoni, J., 1989: Giving with impure altruism: Applications to charity and Ricardian equivalence, *Journal of Political Economy,* **97** (December), pp. 1447–1457.

Anuradha, R.V., 2001: IPRs: Implications for biodiversity and local and indigenous communities, *Review of European Community and International Environmental Law,* **10(1),** pp. 27–36.

Arrow, K. and A. Fisher, 1974: Environmental preservation, uncertainty, and irreversibility, *Quarterly Journal of Economics,* **88(1),** pp. 312–319.

Arrow, K., E.E. Leamer, P.R. Portney, R. Radner, R. Solow, et al., 1993: Report of the NOAA (National Oceanic and Atmospheric Administration) panel on contingent valuation, *Federal Register,* 15 January, **58(10),** pp. 4601–4614.

Arrow, K. J. and F.H. Hahn, 1971: *General Competitive Analysis,* Holden-Day, San Francisco, CA.

Austen-Smith, D. and J.R. Wright, 1994: Counteractive lobbying, *American Journal of Political Science,* **38,** pp. 25–44.

Bagnoli, M. and B.L. Lipman, 1992: Private provision of public goods can be efficient, *Public Choice,* **74** (July), pp. 59–77.

Baland, J.M. and J.P. Platteau, 1996: *Halting Degradation of Natural Resources: Is There a Role for Rural Communities?* Clarendon Press, Oxford, UK.

Balmford, A., A. Bruner, P. Cooper, R. Costanza, S. Farber, et al., 2002: Economic reasons for conserving wild nature, *Science,* **297(5583),** pp. 950–953.

Barbier, E.B., M.C. Acreman, and D. Knowler, 1997: *Economic Valuation of Wetlands: A Guide for Policy Makers and Planners,* RAMSAR Convention Bureau, Gland, Switzerland.

Barbier, E.B. and I. Strand, 1998: Valuing mangrove–fishery linkages: A case study of Campeche, Mexico, *Environmental and Resource Economics,* **12(2)** (September), pp. 151–166.

Barzel, Y, 1997: *Economic Analysis of Property Rights,* 2nd ed., Cambridge University Press, Cambridge, UK.

Bell, F.W., 1998: The state of nature as an input into the production function, *Journal of Economic Research,* **3,** pp. 1–20.

Beltratti, A., G. Chichilnisky, and G. Heal, 1995: The green golden rule, *Economics Letters,* **49(2)**: pp. 175–179.

Berkes, F. and C. Folke (eds.), 1998: *Linking Social and Ecological Systems: Management Practices and Social Mechanisms for Building Resilience,* Cambridge University Press, Cambridge, UK.

Berkes, F., J. Colding, and C. Folke (eds.), 2003: *Navigating Social-Ecological Systems: Building Resilience for Complexity and Change,* Cambridge University Press, Cambridge, UK.

Blanchard, O.J. and S. Fischer, 1989: *Lectures on Macroeconomics,* MIT, Cambridge, MA.

Boadway, R., 1974: The welfare foundations of benefit–cost analysis, *Economic Journal,* **84** (December), pp. 926–939.

Bockstael, N., R.J. Kopp, A.M. Freeman III, P.R. Portney, and V.K. Smith, 2000: On valuing nature, *Environmental Science and Technology,* **24(8),** pp. 1384–1389.

Bond, I., 1993: *The Economics of Wildlife and Landuse in Zimbabwe: An Examination of Current Knowledge and Issues,* World Wildlife Fund, Harare, Zimbabwe.

Brandon, K., 1998: Comparing cases: A review of findings. In: *Parks in Peril: People, Politics, and Protected Areas,* Chapter 13, K. Brandon, K.H. Redford, and S.E. Sanderson (eds.), The Nature Conservancy, Washington, DC, pp. 375–414.

Brechin, S.R. and W. Kempton, 1994: Global environmentalism: A challenge to the postmaterialism thesis? *Social Science Quarterly,* **75(2),** pp. 245–269.

Brechin, S.R. 1999: Objective problems, subjective values and global environmentalism: Evaluating the postmaterialist argument and challenging a new explanation, *Social Science Quarterly,* **80(4),** pp. 793–809.

Brechin, S.R., P.R. Wilshusen, C.L. Fortwangler, and P.C. West, 2002: Beyond the square wheel: Toward a more comprehensive understanding of biodiversity conservation as social and political process, *Society and Natural Resources,* **15(1),** pp. 41–64.

Brown, K., E.L. Tompkins, and W.N. Adger, 2002: *Making Waves: Integrating Coastal Conservation and Development,* Earthscan, London, UK.

Brown-Weiss, E. and H. Jacobson (eds.), 1998: *Engaging Countries: Strengthening Compliance with International Environmental Accord*s, MIT Press, Cambridge, MA.

Buttel, F.H., 2000: World society, the nation-state, and environmental protection, *American Sociological Review,* **65(1),** pp. 117–121.

Cardoso, F., 2003: Civil society and global governance: contextual paper prepared by the panel's chairman. Available at www.un.org/reform/pdfs/cardosopaper13june.htm. Accessed 26 June 2003.

Chambers, R., 1983: *Rural Development: Putting the Last First,* Longman, London, UK.

Chambers, R., 1997: *Whose Reality Counts? Putting the Last First,* Intermediate Technology Publications, London, UK.

Chambers, W. Bradnee, 2003a: Towards an improved understanding of legal effectiveness of international environmental treaties, UNU/IAS working paper series, UN University/Institute of Advanced Studies International Environmental Governance, Yokahama, Japan.

Chambers, W.B., October, 2003b: How can interlinkages improve the legal effectiveness of international environmental agreements? UNU/IAS working paper series, UN University/Institute of Advanced Studies International Environmental Governance, Tokyo, Japan.

Chichilnisky, G., 1996: An axiomatic approach to sustainable development, *Social Choice and Welfare,* **13(2),** pp. 219–248.

Chubb, J.E., 1983: *Interest Groups and the Bureaucracy: The Politics of Energy,* Stanford University Press, Stanford, CA.

CI (Conservation International), 2000: Conservation international creates new market mechanism to conserve global forests, press release, 25 September, Conservation International, Washington, DC.

Clark, R., 1998: Community issues groups, UKCEED Bulletin, 55:19.

Commission on Global Governance, 1995: *Our Global Neighbourhood,* The Commission on Global Governance, Oxford University Press, Oxford, UK.

Costanza, R., R. d'Arge, R. de Groot, S. Farber, M. Grasso, et al., 1997: The value of the world's ecosystem services and natural capital, *Nature,* **387(6230),** pp. 253–260.

Daily, G. and K. Ellison, 2002: *The New Economy of Nature*: *The Quest to Make Conservation Profitable,* Island Press, Washington, DC.

Dasgupta, P., 2002: Modern economics and its critics. In: *Fact and Fiction in Economics: Models, Realism, and Social Construction,* U. Maki (ed.), Cambridge University Press, Cambridge, UK.

Dasgupta, P., 2003: *Human Well-Being and the Natural Environment,* Revised ed. with new appendix, Oxford University Press, Oxford, UK.

Davidson, D.J., 2001: Federal policy in local context: The influence of local state-societal relations on endangered species act implementation, *Policy Studies Review,* **18(1),** pp. 212–240.

Debreu, G., 1954: *The Theory of Value: An Axiomatic Analysis of Economic Equilibrium,* Cowles Foundation Monograph 17, Yale University Press, New Haven, CT.

De Sombre, E.R., 2000: *Domestic Sources of International Environmental Policy: Industry, Environmentalists and US Power,* MIT Press, Cambridge, MA.

Del Valle, A., 1999: Managing complexity through methodic participation: The case of air quality in Santiago, Chile, *Systemic Practice and Action Research,* **12(4),** pp. 367–380.

Diamond, P. and J. Hausman, 1994: Contingent valuation: Is some number better than no number? Symposium on Contingent Valuation, *Journal of Economic Persepctives,* **8(4)** (Fall), pp. 45–64.

Dixit, A. and R. Pindyck, 1994: *Investment Under Uncertainty,* Princeton University Press, Princeton, NJ.

Domhoff, G. William, 1996: *State Autonomy or Class Dominance? Case Studies on Policy Making in America,* Aldine de Gruyter, New York, NY.

Dunlap, R.E. and A.G. Mertig, 1992: *American Environmentalism: The U.S. Environmental Movement, 1970–1990,* Taylor and Francis, Philadelphia, PA.

Erikson, P.P., 1999: A whaling we will go: Encounters of knowledge and memory at the Makah Cultural and Research Center, *Cultural Anthropology,* **14(4),** pp. 556–583.

ESRC (Economic and Social Research Council), 1998: Strengthening decision-making for sustainable development, Report of a workshop held at Eynsham Hall, Oxford, 15–16 June, ESRC, Swindon, UK.

Evans, P.B., H.K. Jacobson, and R.D. Putnam, 1993: *Double-Edged Diplomacy: International Bargaining and Domestic Politics,* University of California Press, Berkeley, CA.

Evans, P., 1995: *Embedded Autonomy,* Princeton University Press, Princeton, NJ.

Feeny, D., F. Berkes, B.J. McCay, and J.M. Acheson, 1990: The tragedy of the commons: Twenty-two years later, *Human Ecology,* **18(1),** pp. 1–19.

Ferraro, P.J., 2001: Global habitat protection: Limitations of development interventions and a role for conservation performance payments, *Conservation Biology,* **15(4),** pp. 990–1000.

Ferraro, P.J. and R.D. Simpson, 2002: The cost-effectiveness of conservation payments, *Land Economics,* **78(3),** pp. 339–353.

Ferraro, P.J. and A. Kiss, 2002: Direct payments to conserve biodiversity, *Science,* **298(5599),** 29 November, pp. 1718–19.

Fischoff, B., 2004: Cognitive processes in stated preference methods. In: *Handbook of Environmental Economics, Volume II: Valuing Environmental Changes,* K-G. Mäler and J. Vincent (eds.), North Holland, Amsterdam, The Netherlands.

Fisher, D.R., 2004: *National Governance and the Global Climate Change Regime,* Rowman & Littlefield Publishers, Inc., Lanham, MD.

Fisher, D.R. and J.F. Green, 2004: Understanding disenfranchisement: Civil society and developing countries' influence and participation in global governance for sustainable development, *Global Environmental Politics,* **4(3).**

Fisher, D.R. and W.R. Freudenburg, 2005: Post industrialization and environmental quality: An empirical analysis of the environmental state, *Social Forces,* **83(1).**

Franck, T.M., 1990: *The Power of Legitimacy among Nations,* Oxford University Press, Oxford, UK.

Frank, D.J., A. Hironaka, and E. Schofer, 2000: The nation-state and the natural environment over the twentieth century, *American Sociological Review,* **65** (February), pp. 96–116.

Freeman III, A.M., 2002: *The Measurement of Environmental and Resource Values: Theory and Methods,* 2nd ed., Resources for the Future, Washington, DC.

Frickel, S., and D.J. Davidson, 2004: Building environmental states: Legitimacy and rationalization in sustainability governance, *International Sociology,* **19(1),** pp. 89–110.

Glaeser, E.L., 2003: Psychology and the market, Discussion paper 2023, December, Harvard Institute of Economic Research, Cambridge, MA.

Gollier, C., 2002: Discounting an uncertain future, *Journal of Public Economics,* **85(2),** pp. 149–166.

Gorman, W.M., 1955: The intransitivity of certain criteria used in welfare economics, *Oxford Economic Papers,* new series, **7,** pp. 25–35.

Gowdy, J.M., 2004: The revolution in welfare economics and its implications for environmental valuation and policy, Rensselaer Polytechnic Institute, Working paper. Available online at http://www.rpi.edu/dept/economics/www/faculty/gowdy/fff/Gowdy2004.pdf.

Grimble, R., M. K. Chan, J. Aglionby, and J. Quan, 1995: *Trees and Trade-offs: A Stakeholder Approach to Natural Resource Management,* Gatekeeper series no. 52, International Institute for Environment and Development, London, UK.

Guha, R. and J. Martinez-Alier, 1997: *Varieties of Environmentalism: Essays North and South,* Earthscan, London, UK.

Guijt, I., 1998: *Participatory Monitoring and Impact Assessment of Sustainable Agricultural Initiatives: An Introduction to Key Elements,* SARL discussion paper number 1, International Institute for Environment and Development, London, UK.

Habermas, J., 1975: *Legitimation Crisis,* Beacon Press, Boston, MA.

Haneman, W.M., 1994: Valuing the environment through contingent valuation, Symposium on Contingent Valuation, *Journal of Economic Persepctives,* **8(4)** (Fall), pp. 19–43.

Hanley, N. and C. Spash, 1993: *Cost-Benefit Analysis and the Environment,* Edward Elgar, Cheltenham, UK.

Hardin, G., 1968: The tragedy of the commons, *Science,* **162,** pp. 1243–1248.

Hausman, J.A., G.K. Leonard, and D. McFadden, 1995: A utility-consistent combined discrete choice and count data model assessing recreational use losses due to natural resource damage, *Journal of Public Economics,* **56,** pp. 1–30.

Heal, G., 2000: *Nature and the Marketplace: Capturing the Value of Ecosystem Services,* Island Press, Washington, DC.

Heal, G., 2002: Biodiversity and globalization, paper presented at 2002 Kiel Week Conference at Kiel University Institute for World Economics, 26 June 2002, Kiel, Germany.

Henriquez, B., 2004: Information technology: The unsung hero of market-based environmental policy, *Resources,* **152,** pp. 9–12.

Hicks, J.R., 1939: The foundations of welfare economics, *Economic Journal,* **49,** pp. 69.

Holland, J. (ed.), 1998: *Whose Voice? Participatory Research and Policy Change,* Intermediate Technology Publications, London, UK.

Holling, C.S. (ed.), 1978: *Adaptive Environmental Assessment and Management,* John Wiley, New York, NY.

Inglehart, R., 1995: Public support for environmental protection: Objective problems and subjective values in 43 societies, *PS: Political Science & Politics,* **28(1),** pp. 57–72.

IPPR (Institute of Public Policy Research), 1999: *Models of Public Involvement,* Institute of Public Policy Research, London, UK.

Irwin, E., 2002: The effects of open space on residential property values, *Land Economics,* **78(4),** pp. 465–81.

Journal of Economic Perspectives, 1994: Symposium on Contingent Valuation, with papers by P. Portney, The contingent valuation debate: Why economists should care, W.M. Hanemann, Valuing the environment through contingent valuation, and P. Diamond and J. Hausman, Contingent valuation: Is some number better than no number? *Journal of Economic Perspectives,* **8(4)** (Fall), pp. 3–64.

Kaldor, N., 1939: Welfare propositions. In: Economics and inter-personal comparisons of utility, *Economic Journal,* **49,** p. 549.

Kahneman, D., and J. Knetsch, 1992: Valuing public goods: The purchase of moral satisfaction, *Journal of Environmental Economics and Management,* **22(1),** pp. 57–70.

Kazis, R. and R.L. Grossman, 1991: *Fear at Work: Job Blackmail, Labor, and the Environment,* New Society Publishers, Philadelphia, PA.

Kidd, Q. and A-R. Lee, 1997a: Postmaterialist values and the environment: A critique and reappraisal, *Social Science Quarterly,* **78(1),** pp. 1–15.

Kidd, Q. and A-R. Lee, 1997b: More on postmaterialist values and the environment, *Social Science Quarterly,* **78(1),** pp. 36–43.

Kokkelenberg, E. and W. Nordhaus, 1999: *Nature's Numbers: Expanding the National Economic Accounts to Include the Environment,* National Academy Press, Washington, DC.

Kolstad, C.D., 2000: *Environmental Economics,* Oxford University Press, Oxford, UK.

Koopmans, T.J., 1960: Stationary ordinal utility and time preference, *Econometrica,* **28(2),** pp. 287–309.

Krasner, S. (ed.), 1983: *International Regimes,* Ithaca, NY.

Kremen, C., J. Niles, M. Dalton, G. Daily, P. Erlich, et al., 2000: Economic incentives for rainforest conservation across scales, *Science,* **288,** pp. 1828–1832.

Langholz, J.A., J.P. Lassoie, D. Lee, and D. Chapman, 2000: Economic considerations of privately owned parks, *Ecological Economics,* **33(2),** pp. 173–183.

Laumann, E.O. and D. Knoke, 1987: *The Organizational State: Social Change in National Policy Domains,* University of Wisconsin Press, Madison, WI.

Leach, M., R. Mearns, and I. Scoones, 1999: Environmental entitlements: Dynamics and institutions in community-based natural resource management, *World Development,* **27(2),** pp. 225–247.

Leftwich, A., 1994: Governance, the state and the politics of development, *Development and Change,* **25(2),** pp. 363–386.

Little, I.M.D., 1950: *A Critique of Welfare Economics,* Cambridge University Press, Cambridge, UK.

Lowndes, V., L. Pratchett, and G. Stoker, 1998: *Guidance on Enhancing Public Participation in Local Government,* Department of the Environment, Transport and the Regions, London, UK. Available at http://www.odpm.gov.uk/stellent/groups/odpm_localgov/documents/pdf/odpm_locgov_pdf_023830.pdf.

Lukes, S., 1974: *Power: A Radical View,* MacMillan, London, UK.

Mäler, K-G. and J. Vincent, 2004: *Handbook of Environmental Economics, Volume II: Valuing Environmental Changes,* North Holland, Amsterdam, The Netherlands.

MA (Millennium Ecosystem Assessment), 2003: *Ecosystems and Human Well-being: A Framework for Assessment,* Island Press, Washington, DC, 245 pp.

Mishan, E.J., 1981: A reappraisal of the principles of resource allocation. In: *Economic Efficiency and Social Welfare: Selected Essays on Fundamental Aspects of the Economic Theory of Social Welfare,* E.J. Mishan (ed.), Allen and Unwin, London, UK, pp. 3–13(a).

Muir-Leresche, K. and R.H. Nelson, 2000: *Private Property Rights to Wildlife: The Southern Africa Experiment,* Center for Private Conservation, Competitive Enterprise Institute, Washington, DC.

Newell, R.G. and W.A. Pizer, 2003: Discounting the distant future: How much do uncertain rates increase valuations? *Journal of Environmental Economics and Management,* **46,** pp. 52–71.

Nordlinger, E.A., 1981: *On the Autonomy of the Democratic State,* Harvard University Press, Cambridge, MA.

Oates, J.F., 1999: Myth and reality in the rain forest: How conservation strategies are failing in West Africa, University of California Press, Berkeley, CA.

O'Brien, K.L. and R.M. Leichenko, 2003: Winners and losers in the context of global change, *Annals of the Association of American Geographers,* **93,** pp. 89–103.

Odum, H.T., 1981: *Energy Basis for Man and Nature,* McGraw-Hill, New York, NY.

Olson, M., 1965: *The Logic of Collective Action; Public Goods and the Theory of Groups,* Harvard University Press, Cambridge, MA.

Olsson, P., C. Folke, and F. Berkes, 2004: Adaptive comanagement for building resilience in social-ecological systems, *Environmental Management,* **34(1),** pp. 75–90.

Ostrom, E., 1990: *Governing the Commons: The Evolution of Institutions for Collective Action,* Cambridge University Press, Cambridge, UK.

Pagiola, S., J. Bishop, and N. Llandell-Mills (eds.), 2002: *Selling Forest Environmental Services: Market-based Mechanisms for Conservation and Development,* Earthscan, London, UK.

Parry, I., W.H., Lawrence, H. Goulder, and D. Burtraw, 1997: Revenue-raising vs. other approaches to environmental protection: The critical significance of pre-existing tax distortions, *RAND Journal of Economics,* **28(4),** pp. 708–731.

Pattanayak, S.K. and R.A. Kramer**,** 2001: Worth of watersheds: A producer surplus approach for valuing drought mitigation in eastern Indonesia, *Environment and Development Economics,* **6,** pp. 23–146.

Pearce, D.W., 1998: Auditing the Earth, *Environment,* **40(2),** pp. 23–28.

Pearce, D.W., 2004: Environmental policy as a tool for sustainability. In: *Scarcity and Growth in the New Millennium,* R.D. Simpson, M.A. Toman, and R.U. Ayres (eds.), Resources for the Future, Washington, DC.

Pearce, D.W. and Turner, R.K., 1990: *Economics of Natural Resources and the Environment,* Johns Hopkins University Press, Baltimore, MD.

Perrings, C. and B. Walker, 1997: Biodiversity, resilience and the control of ecological-economic systems: The case of fire-driven rangelands, *Ecological Economics,* **22(1)** (July), pp. 73–83.

Pezzey, J.C.V., 1997: Sustainability constraints versus "optimality" versus intertemporal concern, and axioms versus data, *Land Economics,* **73(4),** pp. 448–466.

Pezzey, J.C.V and M.A. Toman, 2002: Progress and problems in the economics of sustainability. In: *Yearbook of Environmental and Resource Economics* 2002/3, H. Folmer and T. Tietenberg (eds.), Edward Elgar, Cheltenham, UK, pp. 165–232.

Pierce, J.C., 1997: The hidden layer of political culture: A comment on "post-materialist" values and the environment: a critique and reappraisal, *Social Science Quarterly,* **78(1),** pp. 30–35.

Pigou, A.C., 1932: *The Economics of Welfare,* 4th ed., MacMillan and Company, London, UK.

Portney, P., 1994: The contingent valuation debate: Why economists should care, Symposium on Contingent Valuation, *Journal of Economic Persepctives,* **8(4)** (Fall), pp. 3–17.

Prabhu, R., C.J.P. Colfer, and R.G. Dudley, 1999: *Guidelines for Developing, Testing and Selecting Criteria and Indicators for Sustainable Forest Management,* Centre for International Forestry Research, Jakarta, Indonesia.

Putnam, R.D., 1988: Diplomacy and domestic politics: The logic of two-level games, *International Organization,* **42,** pp. 427–460.

Raustiala, K., 2000: Compliance and effectiveness in international regulatory cooperation, *Case Western Reserve Journal of International Law,* **32,** pp. 387–440.

Ready, R.C. and R.C. Bishop, 1991: Endangered species and the safe minimum standard, *American Journal of Agricultural Economics,* **72(2),** pp. 309–312.

Rees, W., and M. Wackernagel, 1994: *Our Ecological Footprint: Reducing Human Impact on the Earth,* New Society, Philadelphia, PA.

Reid, J. and R. Rice, 1997: Assessing natural forest management as a tool for tropical forest conservation, *Ambio,* **26(6),** pp. 382–386.

Richardson, A., 1998: Health Panels. In: *Panels and Juries—New Government, New Agenda,* R. Sykes and A. Hedges (eds.), Social Research Association, London.

Roy, A., 1999: *The Greater Common Good,* India Book Distributors, Bombay, India.

Sagoff, M., 1994: Four dogmas of environmental economics, *Environmental Values,* **3(4)** (Winter), pp. 285–310.

Samuelson, P.A., 1951: Abstract of a theorem concerning substitutability in open Leontief models. In: *Activity Analysis of Production and Allocation,* T. Koopmans (ed.), Wiley, New York, NY, pp. 147–154.

Samuelson, P.A., 1954: The pure theory of public expenditure, *Review of Economics and Statistics,* **36,** pp. 387–389.

Scher, S.J., A. White, and D. Kaimowitz, 2002: *Making Markets Work for Forest Communities,* Policy brief, Forest Trends, Washington, DC.

Scitovsky, de, T., 1941: A note on welfare propositions in economics, *Review of Economics and Statistics,* **9,** pp. 77–88.

Seppanen, J. and E. Valiverronen, 2000: Nature in newspaper photographs: On the relations between text and image in environmental discourse, *Sosiologia,* **37(4),** pp. 330–348.

Shogren, J.F. and T.D. Crocker, 1999: Risk and its consequences, *Journal of Environmental Economics and Management,* **37(1),** pp. 44–51.

Simpson, R.D. and R.A. Sedjo, 1996: Paying for the conservation of endangered ecosystems: A comparison of direct and indirect approaches, *Environment and Development Economics,* **1,** pp. 241–257.

Simpson, R.D., R.A. Sedjo, and J.W. Reid, 1996: Valuing biodiversity for use in pharmaceutical research, *Journal of Political Economy,* **104(1)** (February), pp. 163–185.

Smith, V.K., 1993: Arbitrary values, good causes, and premature verdicts, *Journal of Environmental Economics and Management,* **22,** pp. 71–89.

Smith, V.K., 1997: Mispriced planet, *Regulation,* **20(3),** pp. 16–17.

Smith, V.K., 1998: Pricing what is priceless: A status report on non-market valuation of environmental resources. In: *International Yearbook of Environmental and Resource Economics 1997/8,* H. Folmer and T. Tietenberg (eds.), Edward Elgar, Cheltenham, UK: pp.156–208. Available at http://biodiversity economics.org/pdf/topics-602-00.pdf, p.42.

Southgate, D., 1998: *Tropical Forest Conservation: An Economic Analysis of the Alternatives for Latin America,* Oxford University Press, Oxford, UK.

Strand, J. and K. Seip, 1992: Willingness to pay for environmental goods in Norway: A contingent valuation study with real payment, *Environmental and Resource Economics,* **2,** pp. 91–106.

Strange, S., 1983: Cave! Hic dragones: A critique of regime analysis. In: *International Regimes,* S. Krasner (ed.), Brooks/Cole, Pacific Grove, CA, pp. 337–354.

Striling, A. and S. Maher, 1999: *Rethinking Risk: A Pilot Multi-Criteria Mapping of a Genetically Modified Crop in Agricultural Systems in the UK,* Science Policy Research Unit, Brighton, UK.

Stroup, R.L., 2003. *Economics: What Everyone Should Know About Economics and the Environment,* Cato Institute, Washington, DC.

Taylor, S., 1984: *Making Bureaucracies Think: The EIS Strategy of Environmental Reform,* Stanford University Press, Palo Alto, CA.

Terborgh, J., C. van Schaik, L. Davenport, and M. Rao (eds.), 2002: *Making Parks Work: Strategies for Preserving Tropical Nature,* Island Press, Washington, DC.

Terborgh, J., 1999: *Requiem for Nature,* Island Press, Washington, DC.

Thorsnes, P.G., 2002: The value of a suburban forest reserve: Estimates from sale of vacant residential building lots, *Land Economics,* **78(3),** pp. 426–441.

Tietenberg, T., 1978: Spatially differentiated air pollution emissions charges: An economic and legal analysis, *Land Economics,* **54,** pp. 265–277.

Toman, M.A., 1998: Why not to calculate the value of the world's ecosystem services and natural capital, *Ecological Economics,* **25(1)** (April), pp. 57–60.

Varian, H.R., 1992: *Microeconomic Analysis,* 3rd ed., Norton, New York, NY.

Victor, D. K. Raustiala, and E.B. Skolnikoff (eds.), 1998: *The Implementation and Effectiveness of International Environmental Commitments: Theory and Practice,* MIT Press, Cambridge, MA.

Vogel, J.H., 1994. *Genes for Sale: Privatization as a Conservation Policy,* Oxford University Press, Oxford, UK.

Vossler, C. and J. Kerkvliet, 2003: A criterion validity test of the contingent valuation method: Comparing actual and hypothetical voting behavior for a referendum, *Journal of Environmental Economics and Management,* **45(3),** pp. 631–649.

Wade, R., 1988: *Village Republics,* Cambridge University Press, Cambridge, UK.

Warner, M., 1997: Consensus participation: an example for protected areas planning, *Public Administration and Development,* **17,** pp. 413–432.

Weitzman, M.L., 1998: Why the distant future should be discounted at the lowest possible rate, *Journal of Environmental Economics and Management,* **36(3),** pp. 201–208.

Wells, M. and K. Brandon with L. Hannah, 1992: *People and Parks: Linking Protected Area Management with Local Communities,* World Bank, World Wildlife Fund, and US Agency for International Development, Washington, DC.

Western, D. and R.M. Wright, 1994: *Natural Connections: Perspectives in Community Based Conservation,* Island Press, Washington, DC.

Wilson, J.Q., 1980: *The Politics of Regulation,* Basic Books, New York, NY.

World Bank, 1992: *Governance and Development,* World Bank, Washington, DC.

World Bank, 1997: *Investing in Biodiversity: A Review of Indonesia's Integrated Conservation and Development Projects,* Indonesia and Pacific Islands Country Department (IPICD), Washington, DC.

Young, O., 1999: *Governance in World Affairs,* Cornell University Press, Ithaca, 224pp.