WILD SPECIES AS COMMODITIES

Managing Markets and Ecosystems for Sustainability

CURTIS H. FREESE

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To
Mom and Dad,
Heather and Erica

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Preface

This book is the result of four-year study supported by the World Wildlife Fund (known as the World Wide Fund for Nature in Europe or, more generally, WWF) designed to provide a better understanding of the link between the commercial consumptive use of wild species and nature conservation. The central question addressed in the study was how to manage consumptive use in order to minimize its negative effects on biodiversity and maximize its potential as a conservation tool. The effort began with fifteen case studies commissioned by WWF to examine this link in various uses of wild species, from forest management and fisheries to big game and waterfowl hunting. These studies have been published as a separate volume, *Harvesting Wild Species: Implications for Biodiversity Conservation*, ed. C. H. Freese (Baltimore: Johns Hopkins University Press, 1997). A general synopsis of the results of the overall study, including a discussion of the fifteen case studies, is presented in a WWF discussion paper, *The Commercial, Consumptive Use of Wild Species: Managing It for the Benefit of Biodiversity,* by C. H. Freese (Washington, D.C.: WWF Unites States, 1996), which was widely distributed for comment to individuals and institutions working on the issue. A key output developed by the WWF task force for this study and presented in the discussion paper was the "WWF Guidelines for the Commercial, Consumptive Use of Wild Species." These guidelines have been slightly modified, based on feedback we received, and are included as an appendix to this volume.

The idea for this book took shape as I reviewed what emerged from the case studies and the volumes of other information and ideas gathered in the course of the study. Although much has been written about consumptive use and conservation within the various individual disciplines—economics, sociology, ecology, forestry, fisheries, recreational hunting, and so on—few publications have looked at the issue by integrating the work of these disciplines. Despite the generally recognized need for interdisciplinary approaches to natural resource management, ecologists and economists, forest and fisheries managers, and temperate zone and tropical zone natural resource managers still seldom compare notes. The goal of this book is to facilitate such comparisons and to propose interdisciplinary ways of thinking about and managing wild species use within the context of biodiversity conservation. I have also tried to provide an introductory framework for students and others who are new to the subject and to shed new light on questions and challenges that face both policy and field practitioners regardless of their particular disciplines.

My first thanks go to WWF, particularly WWF United States, WWF International, WWF United Kingdom, and WWF Canada, for the strong and consistent financial and institutional backing provided to me in carrying out the study and writing this book. A major benefit of working with WWF has been the ability to tap the knowledge of a multidisciplinary staff that is dispersed around the world and working from the policy to the field level. I owe special thanks to Ginette Hemley for her support and enthusiasm from the very beginning of the study to the completion of this book. Many other people, both within and outside WWF, contributed along the way by providing advice, information, and ideas and critiquing drafts of the manuscript. I am particularly grateful to the following (asterisks indicate members of the WWF task force for the study): Tundi Agardy, Cleber Ahlo,* Robert Buschbacher, John Butler, Jason Clay,* Barry Coates,* Steve Cornelius,* David Cumming, Dominick DellaSala, Holly Dublin, Anton Fernhout,* Pamela Hathaway, Barbara Hoskinson,* Monte Hummel, Jon Hutton, Kevin Lyonette,* Nick Mabey, Tom McShane,* Rowan Martin, James Martin-Jones, Filemon Romero, David Schorr, Fulai Sheng,* Gordon Shepherd,* Francis Sullivan, Michael Sutton,* Jennifer Swearingen, Magnus Sylven,* Michael 't Sas-Rolfes, Alice Taylor, Caroline Taylor, Russell Taylor, David Trauger, and Niall Watson. I also give my sincere thanks to Maria Boulos and Kimberly Doyle for their attentive and meticulous administrative support throughout the study and to Nanci Davis for applying such skill and care to preparing the figures. The Department of Biology and Renee Library at Montana State University provided invaluable support for bibliographic research. I am extremely grateful to Pat Harris, whose thorough and sensitive copyediting saved me from more embarrassing mistakes than I care to admit and saved readers from more tortuous sentences than they would care to endure.

At Island Press, I thank Barbara Dean for her early interest in the book and for encouraging me to pursue it. I owe very hearty thanks to

my editor, Todd Baldwin, who provided insightful suggestions and tactful prodding from beginning to end.

Finally, a big hug of thanks goes to my wife, Heather, and daughter, Erica, for breaking up my morning routine of research and writing. I knew it was time to wrap up this book when Erica came home from first grade with a drawing that depicted her image of Dad—the back of me facing a computer screen, with a tall file cabinet and stack of books to one side.

Wild Species as Commodities

CHAPTER 1

Commercial Consumptive Use of Wild Species: Conservation Issues

> *Between the extremes of deep wilderness and the private plots of the farmstead lies a territory which is not suitable for crops. . . . This area, embracing both the wild and the semi-wild, is of critical importance. It is necessary for the health of the wilderness because it adds big habitat, overflow territory, and room for wildlife to fly and run. It is essential even to an agricultural village economy because its natural diversity provides the many necessities and amenities that the privately held plots cannot.*
>
> —Gary Snyder (1990)

The consumptive use of wild species directly and indirectly shapes the livelihood of every human community on earth. Most of this use involves trade, which ranges from small-scale and local to massive and international. Local medicinal plant collectors and artisanal fishers sell their daily take in the local village, perhaps earning enough to buy some rice and sugar to round out their evening meal. A publicly owned transnational corporation harvests thousands of cubic meters of wood or thousands of kilograms of fish on the same day and sells the timber or fish on the opposite side of the world, yielding dividends to shareholders in Toronto, Tokyo, and elsewhere around the globe. All but the most remote ecosystems on earth are affected by such use. All the world's seas and significant bodies of freshwater are fished, and virtually all its terrestrial ecosystems (except for Antarctica) are logged, grazed, collected from, and hunted. The interconnectedness of ecosystems ensures that those few places not used are affected by those that are.

In a world where the human population continues to grow in both numbers and per capita consumption, pressures on wild species and natural ecosystems are becoming increasingly severe. We are reaching a point at which traditional means of conservation, in the familiar guise of protected areas and endangered species recovery programs, are no longer adequate. Most natural and seminatural ecosystems and their inhabitant species lie beyond the reach of such efforts. Meanwhile, commerce in wild species constitutes the main source of revenue from most of the world's remaining natural and seminatural ecosystems, ranging from boreal, temperate, and tropical forests to virtually all marine ecosystems. The total economic value of harvested wild species probably exceeds $500 billion annually (U.S. dollars throughout unless noted otherwise) (see chapter 2)—at least twenty times greater than the most reliable estimates for global revenues from nature tourism (Brandon 1996; Goodwin 1996).

The question this book addresses is whether or not the harvest of wild species for monetary gain can be turned to the advantage of conservation. Proponents of the "use-it-or-lose-it" strategy argue that more biodiversity will be conserved by making diverse and full use of natural and seminatural ecosystems and their living resources than by designating more protected areas. D. H. Janzen (1994, p. 4) stakes out this position with respect to tropical habitats when he claims that the use-it-or-lose-it strategy "envisions 80–90 percent of tropical terrestrial biodiversity conserved on 5–15 percent of the tropics," as compared with continuing a traditional approach to protected areas, in which 10–30 percent of biodiversity will be conserved on 1–2 percent of the lands. Janzen advocates establishing a multitude of uses, both consumptive and nonconsumptive, to give value to natural ecosystems. Such an approach is a cornerstone for biodiversity conservation programs undertaken by many multilateral and bilateral development agencies, government-run natural resource agencies, and major nonprofit conservation organizations (e.g., IUCN, UNEP, and WWF 1991).

The principal foundation of the use-it-or-lose-it argument is that commercial consumptive use (CCU) is often crucial for making natural ecosystems sufficiently profitable to economically outcompete alternative uses that would greatly degrade or entirely alter them (e.g., monocrop agriculture replacing forestland, estuaries filled in for coastal development, oceans used as sinks for industrial pollutants) (Benson 1992; McNeely 1988; WWF 1993). In economic terms, CCU may be able to offset the opportunity cost of the next most profitable use of the land or water. Where this is the case, the argument goes, the profit-conscious resource owner should decide to maintain the commercially valuable wild species and its habitat. Moreover, CCU potentially provides many other conservation benefits. Intensive commercial harvesting from one site may relieve pressures to harvest from other sites of higher conservation priority. The use of some wild species as commodities may be more environmentally friendly than the manufacture and use of substitutes. Commercial use may at times provide a tool for managing populations and ecosystems to meet biodiversity conservation goals. And the use of wild species may provide a bridge to connect a society that is increasingly isolated from nature with the importance of natural ecosystems and wild species and the need to manage them wisely.

Nonetheless, CCU is the proverbial double-edged sword for conservationists. If well managed, it can be a tool for biodiversity conservation; if poorly managed, it can lead to overexploitation and biotic impoverishment. Many believe that commercial use of wild species in natural ecosystems has not yet been broadly demonstrated as a sustainable land-use option that maintains biodiversity and other nature-based values such as "wilderness," seeing it instead as generally leading to biotic impoverishment. R. F. Noss (1991, p. 121) takes this skeptical view of the conservation benefits of sustainable use when he states that "The confidence that we can manage landscapes sustainably for multiple uses is no less arrogant than the humanistic assumption that every environmental problem has a technological solution." Similarly, J. Terborgh and C. P. van Schaik (1997, p. 31) state that they are "skeptical that the extensive use of multispecies tropical forests is anything other than a transitory phase in the land-use cascade" to degraded lands, concluding that only the establishment and maintenance of protected areas will be effective in maintaining biodiversity. J. G. Robinson (1993, p. 24), while recognizing the potential conservation benefits of sustainable use, believes that "any use of a biological community will ultimately involve a loss of biological diversity" and that under many socioeconomic conditions, "sustainable use will be impossible." Based on the history of wildlife commerce and conservation in North America, V. Geist (1994, p. 491) concludes that "The important lesson is to keep wildlife out of the marketplace." Finally, animal rights groups have been quick to attack the new paradigm of sustainable use of wild species as not only ethically indefensible but also "unworkable in application, and tragic in effect" (Hoyt 1994, p. 9).

Differing views on the sustainability of CCU are not simply the result of people comparing apples and oranges by looking at different socioeconomic or ecological systems. D. Ludwig, R. Hilborn, and C. Walters (1993, p. 17), in a widely cited article based primarily on the operations of marine fisheries, claim that "There is remarkable consistency in the history of resource exploitation: resources are inevitably overexploited, often to the point of collapse or extinction." However, A. A. Rosenberg and colleagues (1993, p. 828) also reviewed the evidence from marine fisheries and conclude that "There is a sound theoretical and empirical basis for sustainable use . . . exploitation is not inevitable or necessarily irreversible."

To judge the value of the presumed conservation benefits of consumptive use, a second major question must be answered. Even if the offtake is sustainable and the socioeconomic benefits of CCU outcompete alternative uses of land or water, what trade-offs do offtake and management imply for biodiversity and ecosystem integrity in the ecosystems under management? This goes to the heart of the question of ecological sustainability—the maintenance of biodiversity while using species and their ecosystems. M. Mangel and co-workers (1993, p. 575) claim that "Apparently sustainable exploitation can have profound effects on genetics, species, and ecosystem diversity." Besides the incidental effects of offtake and harvest methods on biodiversity, the forces of economic specialization can create a slippery slope for biodiversity as marketed species and their ecosystems are manipulated to increase productivity or commercial quality (Freese 1997a). In contrast to the more obvious link between overexploitation and biotic impoverishment, the effects of sustainable offtake and management have received relatively little attention. The only significant exception is recent work in forest management in North America (e.g., Aplet et al. 1993; Hansen 1997) and, to a lesser extent, in Nordic countries (Sjöberg and Lennartsson 1995).

Although viewpoints differ, the issues raised thus far have in common the environmental or land ethic that places the conservation of species, biodiversity, and ecosystems as the ultimate goal (Holmes 1988; Leopold 1949). Animal rights and animal welfare activists, however, insert distinctly different ethics into the debate. The animal rights ethic emphasizes the rights of the individual organism and considers it morally impermissible to exploit animals (wild or domesticated) for human benefit of any kind. Proponents of animal welfare, as opposed to animal rights, have a less rigid ethic that allows animals to be killed or subjected to suffering only when there is substantial human benefit (Francione 1996; Garner 1996). In the eyes of many animal protectionists (both animal welfare and animal rights advocates), hunting and trapping cannot be justified on the basis of cultural or utilitarian value as a tool in the use-it-or-lose it strategy (King 1991). Animal protectionism, largely in the West, has had an increasing influence on some commercially important uses of wild species.

One example of this influence is the collapse in the early 1980s of the European market for seal products as the result of a public awareness campaign employing images of white-coated harp seal (*Phoca groenlandica*) pups being killed by clubbing (MacKenzie 1996). The moratorium on commercial whaling by the International Whaling Commission and the conservation groups that support it have been criticized for focusing too much attention on animal rights and not enough on human rights (Einarsson 1993). Recreational hunting, a major source of revenues for conservation in southern Africa and North America, is coming under increasing pressure from animal rights organizations (Maveneke 1996; McTaggart-Cowan 1995). Three of the world's largest conservation organizations acknowledged animal welfare as a legitimate consideration when they jointly agreed that "People should treat all creatures decently, and protect them from cruelty, avoidable suffering, and unnecessary killing" (IUCN, UNEP, and WWF 1991, p. 14). Inevitable conflicts arise as individuals and cultures act on their respective interpretations of what constitutes "avoidable suffering" and "unnecessary killing." Common ground will seldom be found here.

While these debates go on, researchers, managers, and owners of natural resources are confronting an array of on-the-ground problems in managing the harvest of wild species. Many wild species populations are intensively harvested as part of humankind's day-to-day economic activities and we simply are not able to entertain the question of whether or not use should occur. In many, if not most, cases, the primary need is to improve the management of already overexploited populations and degraded ecosystems. These range from wild species uses that constitute major international commercial enterprises, such as large-scale forestry and the operations of open marine fisheries, to rapidly growing local and regional markets for bush meat and many nontimber plant products. Thus, in much of the world, the practical question is not *whether* to use wild species but *how* to improve current uses that are clearly unsustainable and causing biotic degradation.

In other cases, conservationists are attempting to employ CCU as a conservation tool by searching for new uses and opening markets for already well managed or previously unexploited populations. Examples include fee hunting in North America, southern Africa, and central Asia, "rain forest–friendly" products such as Brazil nuts (*Bertholletia excelsa*) and tagua (vegetable ivory) (*Phytelephas aequatorialis*), and lesser-known timber species in Neotropical forests. A high-profile example is the successful push by Botswana, Namibia, and Zimbabwe at the 1997 CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora) conference to downlist their African elephant (*Loxodonta africana*) populations to allow limited trade in existing ivory stockpiles. Ivory trade, it is argued, is important for the revenues it generates to fund wildlife conservation programs in these countries, including the culling of excessively high elephant populations.

Progress in biodiversity conservation will often depend, ultimately, on how well the socioeconomic benefits derived from the biodiversity-based values of these ecosystems are able to outcompete and deter alternative forms of land and water use. For many natural and semi-natural ecosystems, CCU still provides the single most tangible socioeconomic benefit, and it will continue to play a pivotal role in averting conversion of these ecosystems to other uses. However, managing these ecosystems for such commercial purposes may often require sacrifices of some naturalness and native biodiversity. Employing CCU as a conservation tool thus poses a dilemma for biodiversity conservationists: get the incentives wrong and populations will be overexploited, but institute what seem like the right incentives (e.g., secure tenure and resource rights) and the resource owner may economically specialize in the most valuable resource, whether wild or domestic, and create a monoculture at great cost to biodiversity (Freese, n.d.).

Managing this dilemma requires that we bring other biodiversity-based values to the decision-making table and incorporate them into a broader framework for biodiversity management that extends far beyond protected areas. In recent years, management of protected areas has increasingly focused on extractive uses and ecological services and other environmental amenities that surround them (Wells, Brandon, and Hannah 1992). This is the philosophy behind biosphere reserves. As ecologists and conservationists have learned that the maintenance of natural ecosystems and biodiversity requires areas much larger than a protected area can typically cover, protected areas have increasingly become only one element in a much broader approach to conservation. W. V. Reid (1996, p. 448) notes that "Increasingly, policy makers are viewing human uses of resources within the context of regional and national conservation needs rather than relegating conservation only to the domain of national parks and protected areas." Integrated conservation-development projects (Wells, Brandon, and Hannah 1992), ecosystem management (Salwasser et al. 1996), and bioregional management (Miller 1996a) are three management concepts that represent this broader approach. The Convention on Biological Diversity epitomizes this trend by recognizing the need for biodiversity conservation to operate well beyond the bounds of protected areas, with sustainable use of wild species as an integral component (Convention on Biological Diversity 1994).

Over the past two decades, we have witnessed a transformation in focus from creating and managing protected areas to managing buffer zones around protected areas to managing landscapes and ecosystems, of which protected areas are often one small component. Protected areas still play a crucial conservation role but within a broader landscape context. This evolution is the result of two factors. First, we now have a better understanding and appreciation of the large geographic scales required for biodiversity conservation. Second, growth of the human population, technology, and consumption has enabled humankind to exploit and affect the natural world at unprecedented levels. The result is that biodiversity conservation must be conducted at much greater scales than heretofore imagined. This transformation has increasingly placed the production of commodities from natural and seminatural ecosystems at the center of, and often at odds with, biodiversity values and conservation strategies. More than ever, we now realize that we must make room for biodiversity in human-dominated landscapes outside protected areas where commodity production goals often dictate decisions about the management of lands and waters.

What Is at Stake?

At stake here is the future management of terrestrial and marine ecosystems that are not yet fully converted to urbanization and domestic production or that are not secured in protected areas—a major portion of the earth's land and water surface (figure 1-1). Pressure to convert the remaining natural and seminatural ecosystems to more intensive and specialized forms of monoculture production will continue as some 90 million more people each year compete for resources to feed, clothe, medicate, warm, and shelter themselves.

The negative effects on biodiversity will be direct and severe wherever natural ecosystems are converted to monocultures. As of 1987,

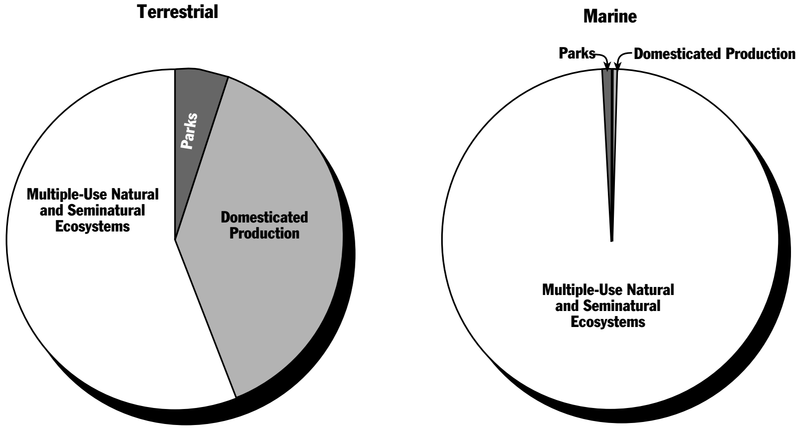

Figure 1-1. *Relative proportion of terrestrial and marine realms devoted to parks and protected areas, domesticated production, and multiple use of natural and seminatural ecosystems.*

cropland occupied roughly 11 percent of the total land area of all the world's continents combined (this and subsequent figures exclude Antarctica) (WRI 1990), and deforestation had affected roughly another 3 percent of the total land area beyond the forest and woodland estate already converted to cropland (calculated from figures in Williams 1990). Of the world's dry rangelands, some 24 million square kilometers, accounting for approximately 18 percent of the earth's total land area, are at least moderately desertified (Durning and Brough 1992). Losses of wetlands (included here as a component of terrestrial systems) have been estimated at 6 percent of the total land area over the past century alone (Turner et al. 1994); of this loss, only a small fraction is apparently due to creation of cropland (Williams 1990). These figures suggest that at least 38 percent of the world's terrestrial biomes have been either converted to alternative land uses or altered in such a way as to seriously affect native biodiversity. Although there are vast regional differences, the global trend is toward this percentage moving higher as forests, grasslands, and wetlands continue to be converted or degraded.

Based on these calculations, approximately 62 percent of the earth's land area is still in a natural or seminatural state. This figure may be conservative compared with one estimate that 48 percent of the world's land area has been subject to low human disturbance and 28 percent to medium human disturbance (WRI 1994) and with another estimate that at least 75 percent of the land on every continent except Europe is available for potential wildlife use (Martin 1993, cited in Edwards 1995).

In the estimated 62 percent of remaining natural or seminatural areas, parks and similar reserves (IUCN protected area categories I–V) now cover 6 percent of the earth's land area (WRI 1992). Despite the rapid growth in parks and reserves over the past three decades, it is questionable whether that rate of growth can continue and, if it does, whether management of these areas will be effective. Management is nonexistent or ineffective in protecting biodiversity in many existing parks (van Schaik, Terborgh, and Dugelby 1997). Only limited and isolated gains for biodiversity conservation seem possible via this avenue in the future.

This leaves roughly 56 percent of the earth's land area still in natural or seminatural ecosystems and open to multiple potential uses. In addition, many ecosystems that have been degraded or altered (and thus are included in the 38 percent figure of converted land), such as forests and grasslands, can recover if given sufficient protection and time. Leaving aside the significant differences in these percentages among regions and biomes, the maintenance of much of the earth's biodiversity clearly will depend on how these natural and seminatural ecosystems and recoverable lands are managed, as parks and reserves are far too small and poorly distributed to do the job (Miller 1996b).

Marine ecosystems are affected by human activities that generally decrease in intensity from nearshore systems to the open marine realm. Coastal areas, at the interface between marine and terrestrial systems, are magnets for various habitat-altering human activities. Mangroves and salt marshes are dredged, filled, and channelized; natural shorelines are replaced with jetties, marinas, and resort and industrial developments; coral reefs are destroyed by anchoring, cyanide fishing, and land-based sources of siltation; aquaculture is expanding in both estuarine and nearshore systems; and exotic species are introduced both purposefully for aquaculture and accidentally via oceangoing vessels—all with negative ecosystem effects that often extend far beyond the immediate area (J. R. Clark 1996; Norse 1993). Although intensive and semi-intensive aquacultural production (the marine equivalent of agricultural production) covers only a small fraction of the total marine area, the fact that it is concentrated in coastal ecosystems means that it can have significant local effects. For example, an estimated 5 percent of the world's mangroves have been cleared for aquaculture (mostly shrimp production), with the loss for some countries and islands in

Southeast Asia exceeding 50 percent (Clay 1996). Open marine systems are altered by such diverse disturbances as deep-sea mining and noise pollution from boat traffic and other sources. Perhaps most seriously, marine systems are biogeochemical sinks that are used, purposefully or incidentally, as a communal dump for all forms of human-produced wastes and chemicals (Norse 1993).

Despite these various assaults, most coastal marine areas remain in a natural or seminatural state and the use of wild species within them is still of major socioeconomic and ecological importance; virtually the entire open marine realm is still essentially a natural ecosystem in which wild species will predominate in human use for the foreseeable future. Protected areas, however, cover only approximately 0.6 percent of the marine realm (calculated from figures in WRI 1994), an order of magnitude less than those in the terrestrial realm.

What Is Meant by *Commercial* and *Consumptive*?

I broadly define commercial use as any use of a wild species that is driven or greatly influenced by a revenue-generating motive for one or more stakeholders. Such stakeholders may include harvesters, managers, owners, and others, whether individuals or public agencies, who directly profit from the use. Thus, for example, although the term *commercial fisheries* is generally equated with the capture and marketing of fish for food, this definition also includes almost all recreational fishing because government agencies that charge a fee for fishing permits, private landowners who collect fishing fees, and individuals who sell fishing gear or operate charter boats either directly manage or strongly influence the fisheries. Similarly, although the term *market hunting* is generally used when the purpose is to sell the animal or its parts, commercial use as defined here includes both market hunting and any recreational hunting that involves the purchase of permits, the payment of fees, or significant commercial involvement in hunting operations.

This is a much more inclusive definition of commercial use of wild species than is often given (e.g., see Robinson and Redford 1991b), and it leaves few circumstances of consumptive use in which the term *commercial* would not apply. Subsistence use in its purest forms may be excluded, but few subsistence hunters or fishers do not also engage in the sale or barter of their harvest. This distinction, however, is not particularly important here, as many of the issues and questions raised are relevant regardless of whether or not the use is purely subsistence oriented.

Consumptive use occurs when an entire organism is deliberately killed or removed or any of its parts are utilized, either as a goal in itself (e.g., recreational hunting and fishing) or for a product (e.g., pets, timber, food, leather) (table 1-1). The focus here is on wild species harvested from natural or seminatural conditions, as opposed to captively raised or cultivated organisms. A special case of consumptive use is the employment of domestic stock to harvest forage and convert it into products more palatable and useful to humans. Recreational hunting cannot generally be considered purely consumptive, since much of its value, commercial or otherwise, is based on the act of the hunt itself and the environmental setting in which it is conducted (i.e., it is partly nature tourism). CCU is used throughout this book as shorthand for the commercial consumptive use of wild species. Any species subject to

Table 1-1. *Examples of Consumptive Use of Wild Species*

REMOVAL OR KILLING OF WHOLE OR NEARLY WHOLE ORGANISM EXCLUSIVELY FOR PRODUCT
- Capture of fish and wildlife for food
- Harvest of timber for construction material and fuelwood (trees not always killed)
- Harvest of wild ginseng for roots and of bears for gallbladders, for medicinal uses
- Harvest of fruits, nuts, and eggs for food and other uses
- Capture of live birds and ornamental fish for pets; collection of live plants as ornamentals

REMOVAL OF PART OF ORGANISM
- Harvest of foliage and flowers for food or ornamentation
- Grazing of grass by domestic livestock
- Tapping of trees for rubber and maple syrup

CAPTURE OR KILLING OF ORGANISM LARGELY FOR RECREATION
- Recreational hunting of waterfowl and big game (may also be valued for meat)
- Recreational fishing, including catch and release (fish may also be valued for food)
- Trophy hunting and fishing (the trophy itself also has significant value)

such use is referred to here as a wild species commodity or CCU product.

In contrast to *consumptive,* the term *nonconsumptive* is applied when use does not involve such direct and deliberate killing or removal (e.g., bird-watching and other forms of nature tourism). The distinction between the two terms, however, is more blurred than first meets the eye in ways that are significant for conservation. The labels *consumptive* and *nonconsumptive* are applied according to the effect of human action on individual organisms, whereas biodiversity conservation is concerned with ultimate effects on populations and ecosystems. Neither term is a useful indicator of these ultimate effects. For example, whereas the killing of a postreproductive buck may have no effect, or even a positive effect, on a deer population, overzealous bird-watchers may crowd the nesting colony of a rare bird species, resulting in abandoned nests and a lower population. Yet the first is labeled consumptive and the latter nonconsumptive. Thus, although the term *consumptive* correctly places the burden of proof on demonstrating that removal of an organism has no undue consequences for a population, nature tourism, though labeled as nonconsumptive, may also affect wild species populations with impoverishing effects for biodiversity (Cater 1993).

What Is Meant by *Sustainability*?

Three types of sustainability can be defined:

1. *Sustainable offtake.* In this type of sustainability, the removal of individuals from a population, known as offtake, is conducted at a rate and in a manner that can be continued indefinitely. This includes the harvest of parts or products of organisms, such as tree sap and foliage.

2. *Ecological sustainability.* Here, offtake from the target population(s) and associated management practices do not lead to notable changes, particularly degradation, in natural (native) biodiversity at any level (genetic, species, ecosystem) or in natural ecosystem processes and functions (e.g., nutrient cycling and watershed protection).

3. *Socioeconomic sustainability.* In socioeconomic sustainability, the socioeconomic rewards from CCU create incentives for sustain-

able offtake and, more broadly, for ecological sustainability by the primary stewards of the target species and ecosystem.

Sustainable offtake, at the most basic level, is any harvest that does not drive the harvested population toward extinction. More generally, however, sustainable offtake implies that the population size is not reduced or the ecosystem degraded to the point at which harvest levels are greatly diminished. Natural fluctuations in population levels may cause sustainable levels of offtake to vary substantially from year to year or from decade to decade.

By definition, offtake and management that are ecologically sustainable are compatible with biodiversity conservation. Although ecological sustainability implies that offtake is sustainable, if the species being harvested is naturally rare and of little ecological significance in its community, ecological sustainability may be little affected by overexploitation of the species. In addition, management that specializes in the production of a commercially important wild species, such as a single-species forest plantation, may be sustainable in terms of offtake but not ecologically sustainable. This definition of ecological sustainability is largely identical with the definition for sustainable use in the text of the Convention on Biological Diversity: "The use of components of biological diversity in a way and at a rate that does not lead to the long-term decline of biological diversity, thereby maintaining its potential to meet the needs and aspirations of present and future generations" (Convention on Biological Diversity 1994, p. 5). This definition differs, however, from that recently proposed by J. B. Callicott and K. Mumford (1997). Although these authors also discuss ecological sustainability in terms of conserving ecosystems that are inhabited and economically exploited, they describe ecological sustainability as aiming at "preserving ecosystem health; that is, normal ecological processes and functions, irrespective of which species perform them" (p. 39). Their definition, then, describes the ecological conditions necessary for sustainable offtake as defined earlier.

Implicit in the definition of socioeconomic sustainability is that the incentives provided by CCU engender support for sustainable management both from the primary stewards of the resource, such as landowners, and from other key decision makers, such as government agencies whose policies affect management. Socioeconomic sustainability must be defined in terms of how successfully the socioeconomic incentives generated by a given use meet the conservation goals for the area under management. Natural forest management is socioeconomically sustainable if it provides socioeconomic incentives that result in the forest owner maintaining the natural forest cover. If, however, despite the fact that natural forest management is highly profitable for a forest owner, the owner decides to convert the natural forest into a plantation forest because the latter is even more socioeconomically rewarding, natural forest management is not in this case socioeconomically sustainable. In other words, the socioeconomic sustainability of a given form of land or water management depends on its socioeconomic competitiveness with alternative uses of the land or water.

Biodiversity Conservation: Standards, Goals, and Monitoring

Throughout this book I will discuss how biodiversity and biodiversity conservation goals are influenced, both positively and negatively, by CCU. Articulating biodiversity conservation goals and assessing how successfully they are met, however, is a difficult challenge for biodiversity conservationists. This is because of both our general ignorance regarding the planet's biodiversity (Wilson 1992) and confusion in the conservation community regarding how to define biodiversity conservation goals (Reid 1996). In the broadest sense, I adopt here R. F. Noss's (1996, p. 574) suggestion that "Biodiversity exists and must be conserved at genetic, species, and ecosystem levels of organization . . . and across many spatial and temporal scales," including the need to consider landscape diversity that incorporates the spatial arrangement of habitats and communities. A fundamental, and often more elusive, aspect of this general goal is that "natural" or "native" biodiversity and ecosystems are to be conserved, as opposed to obviously artificial ecosystems in which human interventions and/or domesticated species dominate. I will generally use the term *natural ecosystem* (as opposed to the frequently used but less inclusive term *wildland*) to indicate ecosystems in which human influence is currently not a dominating factor, even though it may have been in the past. The postulated human-caused extinctions of megafauna in, for example, Madagascar to North America thousands of years ago surely changed dramatically and forever the ecosystems and biodiversity of these regions, but this does not prevent us from identifying natural ecosystems in these regions today. Thus, *natural,* as used here, includes moderate levels of human use and activities that do not lead to significant alterations or declines in biodiversity and ecosystem function.

Natural, and many of its counterparts, such as *native* and *integrity,* are fickle terms when used in defining biodiversity conservation goals. Aldo Leopold (1949, pp. 224–225) had these concepts in mind when he wrote that "A thing is right when it tends to preserve the integrity, stability, and beauty of the biotic community." M. E. Grumbine (1994, p. 31) conveyed the notion of naturalness when he cited "native ecosystem integrity" as the goal of ecosystem management, as did J. R. Karr and D. R. Dudley (1981, p. 56) in defining "biological integrity" in terms of a community of organisms that is comparable to that of "the natural habitat of the region." J. E. Anderson (1991, p. 347) proposed three indices for assessing naturalness: "(1) the degree to which the system would change if humans were removed; (2) the amount of cultural energy required to maintain the functioning of the ecosystem as it currently exists; and (3) the complement of native species currently in an area compared with the suite of species in the area prior to settlement." This provides a useful framework for *natural* as used in this book, though it begs the obvious question of when "settlement" occurred on most continents, particularly Africa.

From a practical perspective, because human influence has been long-term and pervasive over most of the terrestrial and nearshore realm, including previously labeled "pristine" ecosystems (Botkin 1990; Denevan 1992; Dublin 1991; Sprugel 1991), it is difficult if not impossible to define precise standards of naturalness in terms of biodiversity conservation objectives for most areas (Freese 1997a; Reid 1996). As fishing fleets and environmental contaminants reach all parts of the open marine realm, we are also losing the opportunity to understand what native ecosystem integrity may look like in that vast environment. At the extremes, we can separate the more or less natural from the clearly artificial, the native grassland from the cornfield. In between, however, a large portion of the earth's biotic communities defy being labeled natural or artificial; they are the "seminatural" ecosystems in which most, though generally not all, native species still exist, but in numbers and conditions substantially altered by humans. Despite such far-reaching human influence, for much of the earth we can still define a very large part of what constituted the native biota and ecosystems up to a few hundred years ago, and this continues to serve as a reasonable first standard for setting biodiversity conservation goals (Dinerstein et al. 1995).

What we define and accept as natural is more a function of human values than it is of tabulating the number of native species in an area.

At least on a local scale, management that creates clearly unnatural conditions may have no significant effect on biodiversity or may even create greater biodiversity. Logging gaps in the forest may mimic natural gaps and cause no noticeable change in biodiversity in that forest. Within a few years, except for the decaying stumps, all evidence that it was a logging gap rather than a blowdown gap may be largely unnoticeable and thus more acceptable to those who desire a more "natural" or "pristine" look. However, there is a clearer trade-off in values when an expansive natural wet meadow, whose very size and homogeneity are valued, is blasted to create potholes for waterfowl. Biodiversity within the meadow has surely increased with the creation of a more heterogeneous habitat, but at a cost to naturalness. Such issues beg the question of how prepared we are to accept humans and their influence on ecosystems as part of the natural process. Can humans be viewed in the same light as other predators when fishing induces genetic change in a fish stock? Can we accept some introductions of exotic species as part of the larger, long-term process of species being shuffled around the earth via various processes, but recognize that the current scale and distance of translocation of species by humans is probably unprecedented, with major consequences for biodiversity? In many cases, the answer to such questions will depend on the degree of human influence involved and on how much human-induced change stands out against the backdrop of other factors that determine the structure and function of biological communities.

Biologically diverse and largely natural ecosystems display considerable adaptiveness to change and resilience to disturbance (see chapter 5). Maintaining the adaptiveness and resilience of ecosystems should be another standard against which we measure the influence of use and management. Managing for stability and high production from commercially important species often creates "brittle" ecosystems that are nonadaptive and not resilient (Holling et al. 1995), ultimately compromising the goal of biodiversity conservation.

Articulating a longer-term view, G. K. Meffe (1996, p. 42) argues that "Evolutionary concepts must form the core of conservation management at all levels," with an eye to maintaining both biodiversity and long-term adaptiveness. He suggests that we "should not adopt a *preservationist* mentality of keeping populations as they now are . . . but a *conservationist* mentality of allowing populations to continually change and adapt." This requires not only that we pay close attention to ecosystem- and species-level interactions and conservation, but also that we employ genetically based management systems that give special consideration to patterns of genetic diversity within populations and species so as to maintain their evolutionary potential. Thus, use and management that erode genetic diversity and reduce evolutionary potential compromise biodiversity conservation goals.

CCU management often requires trade-offs with biodiversity conservation goals. This implies that biodiversity conservationists and managers know what is being traded and how to measure it. Marketers of CCU products use revenues to measure the level of sacrifice they may have to make for biodiversity conservation, but what criteria or measurements do conservationists use to understand the biodiversity trade-offs that commodity uses of wild ecosystems may entail? Decisions about trade-offs between commodity production goals and biodiversity conservation in natural ecosystems require that we have standards or benchmarks against which to measure change and conservation goals defining how much human-induced change will be acceptable within a given management unit. This, in turn, depends on decisions regarding the best role for that management unit—intensive commodity production, biodiversity protection, or something in between—within a larger global framework. For highly degraded ecosystems, biodiversity benchmarks may have to be inferred from historical data and from observations of similar areas. Our goals will then be more focused on how much we wish to restore an ecosystem to some previous, more intact, biologically diverse state.

Development of both benchmarks and goals will require a basic understanding of ecosystem and evolutionary processes and changes as much as static measures of biodiversity within a given area. Moreover, our goals will require a heavy dose of human values and subjectivity, which will vary from culture to culture (Freese 1997a). This task will be easier in some ecosystems and under some conditions of use than in others. In many cases, our efforts will be hampered by ignorance of the ecosystem and the history of human influence on it. Further, to the extent that we can define biodiversity benchmarks, current levels of research and monitoring are wholly inadequate to enable us to measure and understand changes caused by harvesting and management regimes, whether at the genetic, species, or ecosystem level (Hansen 1997; Kremen, Merenlender, and Murphy 1994). Managing CCU so that trade-offs between socioeconomic development and biodiversity conservation can be minimized, or so that the two goals may be mutually reinforcing, requires that much more attention be given to defining biodiversity standards, setting conservation goals within the context of those standards, and monitoring and understanding human-induced change.

Summary

Most of the earth's lands and waters are still in natural and seminatural conditions. The future disposition of these remaining natural and seminatural ecosystems, most of which are outside protected areas, will determine how much of the earth's biodiversity will be conserved. In almost all of these ecosystems, wild species are being harvested for trade in markets ranging from local to international.

Given this situation, a major issue facing biodiversity conservationists is how to manage the commercial consumptive use of wild species so that it serves the purposes of biodiversity conservation in the world's remaining natural and seminatural ecosystems as well as in the restoration of already degraded lands and waters. One approach, often called the "use-it-or-lose-it" strategy, advocates making full and extensive use of these ecosystems. This usually implies commercial use, with consumptive use often assuming a major role. The rationale is that unless wild species and their ecosystems convey tangible economic rewards to those responsible for their management, wild species will be eliminated and natural ecosystems will be converted to other uses. Yet because commercial markets have often led to either overexploitation of species or overspecialization in their production, both usually with negative consequences for biodiversity, this strategy presents many pitfalls. For much of the world and for many forms of use, however, the question is not whether to use wild species but rather how to improve management of currently unsustainable uses.

This book examines how the interaction of economic, social, and ecological factors determines whether the commercial consumptive use of wild species leads to biodiversity conservation or degradation. In the process, it identifies the questions and issues that must be addressed to improve the management of CCU, not only to minimize its negative effects on biodiversity but also to employ it as a useful conservation tool.

C H A P T E R 2

A Global Overview

The fish come from the river. . . . The supply is infinite.

—Nsamonie, an African fisherman (Harms 1987)

Probably all the great sea-fisheries are inexhaustible; that is to say that nothing we do seriously affects the number of fish.

—Thomas Huxley (1884)

Until plants were first cultivated and animals were first domesticated a few thousand years ago, humans were entirely dependent on wild species for food, fiber, shelter, and medicines. Hunter-gatherer societies lived for millennia in regions of both low and high biotic diversity, and many continue to do so today. In many cases, they and the ecosystems they inhabit have arguably evolved to some level of harmonious coexistence. Given the current overexploitation of wild species under systems of commercial use, one is tempted to assign problems of overexploitation uniquely to the perverse incentives of the profit motive.

The path to coexistence, however, has not always been harmonious. The "ecologically noble savage" (Redford 1991) is not particularly evident even in prehistoric, precommercial times, during the first wave of human colonization of the earth. Human migration into new lands often led to major extinctions of species before some semblance of ecological balance with the remaining biota was reached. The massive extinctions of large mammals and flightless birds in North America, Madagascar, and New Zealand coincided closely with, and was almost certainly caused by, the arrival of humans who hunted them (Ward 1997; Wilson 1992). The spread of Polynesians across the islands of the Pacific beginning some 8,000 years ago led to the extinction of at least half of the endemic species they encountered. As E. O. Wilson (1992, p. 245) declares, "The voyagers ate their way through the Polynesian fauna." Early human history appears to have foreshadowed some of the problems faced in the modern era of consumptive use.

The Modern Era of Consumptive Use

A sevenfold expansion of the human population during the past 300 years (Demeny 1990), coupled with rapid growth in per capita material expectations, technologies for harvesting wild species (Headrick 1990), and commercial trade in wild species products, produced the modern era of wild species use. Large-scale commercialization of both wild and domestic species products is a fundamental aspect of this trend, although significant trading of wild species products occurred in some parts of the world, especially Asia, before 1700 (Chisholm 1990). The modern era is marked by the commercial harvest of species, often on a massive scale and linking distant markets, in virtually every corner of the globe, both terrestrial and marine.

Wild species are used around the world to meet basic requirements for human survival and for socioeconomic development based on diverse essential and nonessential uses. From native peoples living in the Arctic circle to those in equatorial regions, people often depend on wild species of plants and animals as their only source for meeting most dietary and medicinal needs. In Africa, wild sources of food often become even more crucial for survival during periods of drought or crop failure from other causes (IIED 1995; I. Bond 1993). Moreover, the value of many consumptive uses extends far beyond any measure of subsistence or monetary worth. The acts of managing, harvesting, preparing, and using many wild species constitute an important part of the cultural and spiritual fabric of societies around the world. Whether the annual tradition of cutting the family Christmas tree, the ceremonial taking of a hallucinogenic plant extract, or recreational hunting and fishing, consumptive uses represent and uphold an array of cultural values. This concept is captured in a comment by J. G. Robinson and K. H. Redford (1994, p. 303) in their review of hunting in the Neotropics: "Wild game has a high social value; by securing game and sharing it with other members of the community, the hunter builds debts, acquires allegiances, and contributes to social cohesiveness." Various studies, they note, suggest "a link between the increasing dearth of wild game and a breakdown of the traditional village social structure."

At the other end of the spectrum, wild species commodities, ranging from basic foodstuffs to luxury items, support multimillion-dollar industries through national and international markets. The sale of wild species products is the major source of revenue for many households in both developing and industrialized countries, and the number of people employed by enterprises utilizing wild species is enormous. Taxes, royalties, and fees based on wild species commodities also generate substantial revenues for governments.

Many wild species resources for which there are significant markets have been overexploited and their ecosystems highly altered and generally simplified. As in prehistoric times, overexploitation has occasionally led to extinctions. Hunting, for example, has caused 23 percent of all documented animal extinctions (WRI 1994). Where extinction has been avoided, good management has often emerged only after a period of obvious overuse and population decline, regardless of socioeconomic and ecological conditions (Freese 1997a; Hilborn, Walters, and Ludwig 1995). Forest management from Scandinavia (Hytönen and Blöndal 1995) and Switzerland (McShane and McShane-Caluzi 1997) to Mexico (Kiernan and Freese 1997) and India (Singh et al. 1997) displays this trend. Improved management and recovery of populations have followed bad management and overfishing of the halibut (*Hippoglossus stenolepis*) on the western coast (Knauss 1994) and the striped bass (*Morone saxatilis*) on the eastern coast (Upton 1997) of North America; this same trend can be found in the oxbow lake fisheries of Amazonian Peru (Bodmer et al. 1997). Crocodilian populations, widely overharvested earlier in the twentieth century for their hides and in some cases because of their danger to human life, have bounced back under improved management in Australia, Zimbabwe, and the United States (Hutton 1996; Joanen et al. 1997; Webb and Manolis 1993). Several big game species exhibit this same trend, among them the markhor (*Capra falconeri*) in Pakistan (Johnson 1997) and the elk (*Cervus elaphus*) in North America (Gill 1990).

A few examples drawn from diverse uses and regions of the world indicate the magnitude of consumptive use of wild species, its socioeconomic significance, and its level of sustainabililty.

Timber Use and Forest Management

The world's forests provide diverse timber products, both to the 500 million people who live in or near them (Sharma et al. 1992) and to faraway markets. The annual value of fuelwood and wood-based products worldwide is estimated at $418 billion, or nearly 2 percent of the world's gross domestic product (table 2-1) (FAO 1995b). In the developed world, the single most important commodity from forests is timber for construction and wood products. A greater diversity of forest

Table 2-1. *Worldwide Annual Value of Fuelwood and Wood-Based Industrial Products, 1991*

	Annual Value (U.S. $, Billions)			
	Fuelwood	Wood-Based Industrial Products	Total	% of GDP
Developing countries	70	63	133	4.1
Developed countries	26	259	285	1.4
World	96	322	418	1.8

Source: FAO 1995b.

products is generally used in developing countries, but their single most important timber product is firewood (including charcoal) for heating and cooking. In developing countries, 80 percent of wood is consumed as fuel, and fuelwood supplies 20 percent of all energy needs (FAO 1995b; Sharma et al. 1992). Worldwide, an estimated 3 billion people depend on wood for household energy (FAO 1995b). From 1961 to 1991, fuelwood consumption nearly doubled and use of industrial roundwood increased by 50 percent, so the amount of fuelwood consumed now exceeds industrial roundwood use. However, the projected rate of growth in industrial wood consumption over the first decade or two of the twenty-first century (2.5 percent) is greater than that for fuelwood (1.6 percent). The forestry sector provides annual subsistence and wage employment equivalent to 60 million work years worldwide, 80 percent of which is in developing countries (FAO 1995b).

Forty-six percent of the world's closed forests have been converted to other uses during the past 8,000 years. Of the 54 percent that remain, less than half represent forest areas that are sufficiently large and undisturbed to be considered natural ecosystems. Forty-eight percent of these natural forests are boreal, and 44 percent are tropical (Bryant, Nielsen, and Tangley 1997). Plantation forests, usually dominated by a single species, account for approximately 3 percent of the world's total closed forest cover (Sharma et al. 1992). Major forest conversions occurred centuries and even millennia ago in regions such as eastern Asia, western Europe, the Middle East, and Mesoamerica (Bryant, Nielsen, and Tangley 1997; McNeely 1994; Williams 1990). Developed countries, with 90 percent of the world's temperate forests, went through a period of extensive deforestation in the eighteenth and nineteenth centuries, a conversion fueled by both agricultural expansion, particularly in North America, and timber needs generated by war and the industrial revolution. The industrial revolution brought about a significant transition from multiple-product forestry to single-use wood production (Williams 1990).

Forest cover in Europe, the Nordic countries, and North America has increased, often significantly, during the twentieth century (Hytönen and Blöndal 1995; Koch and Kennedy 1991; McShane and McShane-Caluzi 1997; Williams 1989, 1990). From an ecological perspective, however, much of the forest that has been restored in temperate regions is ecologically distinctive and biotically depauperate compared with earlier forests (Hytönen and Blöndal 1995; McShane and McShane-Caluzi 1997; Noss and Cooperrider 1994).

Developing countries, where most tropical forests are found, continue to lose forests at a rate of nearly 1 percent annually (Botkin and Talbot 1992; Sedjo 1995a). Of the three major tropical regions, Southeast Asia has suffered the most severe loss (Gillis 1992a, b; Poffenberger 1992). As Southeast Asia's forests are depleted, international markets are turning to Latin American and African forests (Dudley, Jeanrenaud, and Sullivan 1995).

Although much, if not most, forest clearing to date in both temperate and tropical regions has been due to agricultural expansion, the major threat to the world's remaining natural forest ecosystems is unsustainable commercial logging (Bryant, Nielsen, and Tangley 1997). Forest management for the production of wood reveals a dismal record in terms of sustainable offtake and ecological sustainability. Misguided policies, government subsidies, and corruption are causative factors. Roughly 10 percent of temperate and boreal forests and 5 percent of tropical forests worldwide are actively managed (Allan and Lanly 1991; Winjum, Meganck, and Dixon 1993). Many temperate forests, including plantations, are being sustainably managed for offtake (Hytönen and Blöndal 1995; Koch and Kennedy 1991; McShane and McShane-Caluzi 1997), but their ecological sustainability is low (see chapter 6). Of productive tropical forests, D. Poore and colleagues (1989) estimate that as of 1985, only 200,000 hectares out of a total of 828 million hectares, or 0.02 percent, were under sustained-yield (i.e., sustainable offtake) management, with no sign of improvement. In fact, N. Johnson and B. Cabarle (1993, p. 19) conclude that "The gap between the principle and practice of natural forest management on most humid tropical forestlands has never been wider." This problem is especially acute for fuelwood, which is widely being cut faster than it is produced in the world's drier forest and woodland regions (Panayotou and Ashton 1992; WRI 1986).

Fisheries

Fisheries yield the second most important wild species commodity in the world, with global revenues from marine food fisheries exceeding $70 billion per year (Sissenwine and Rosenberg 1993). Marine fisheries, however, like much timber harvesting, are highly subsidized enterprises. The total annual operating cost of the global fishing fleet was $92.2 billion in 1989, resulting in an operating deficit of $22 billion without accounting for the cost of capital and a deficit of $54 billion including the cost of capital. Approximately half of the total fishing revenues come from exports, with the global value of fish and fish product exports increasing sixty-three-fold from $571 million in 1950 to more than $36 billion in 1990. Exports from developing countries now nearly equal exports from developed countries in total economic value (FAO 1993).

Many regions depend on fisheries for protein and employment. Globally, food fisheries employ 15–21 million fishers, of whom more than 90 percent are small-scale operators who use traditional equipment and small boats (Weber 1994). Ninety percent of fishery landings in tropical developing countries come from shallow coastal waters, providing 40–95 percent of national animal protein consumption (Holdgate 1993). Worldwide, fish account for 19 percent of the total human consumption of animal protein (Botsford, Castilla, and Peterson 1997).

The consumptive use of fish also has a major recreational component. Although there are no statistics on the global economic significance of this activity, in the United States recreational fishers spent $24 billion on their sport in 1991 (U.S. Department of the Interior 1993).

Although fisheries have sustained coastal and riverine human populations for millennia, extensive exploitation of the world's fisheries, unlike the situation in the world's forests, is a much more recent phenomenon (figure 2-1). Overfishing of the world's oceans has been exacerbated in recent decades by rapid expansion of the world's fishing fleets and technologies and by rapid increases in the price of fish as compared with the price of foods and with the cost of fishing inputs (FAO 1993). The harvest of marine fish increased from roughly 3 mil-

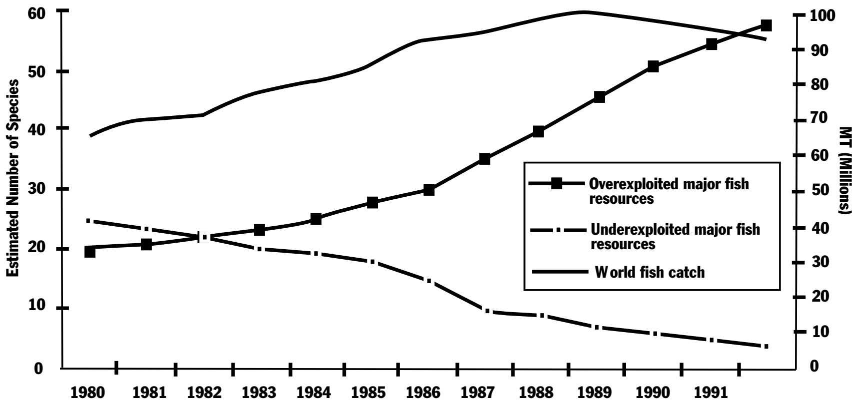

Figure 2-1. *World fish catch and numbers of overexploited and underexploited species, 1980–1991* Source: *Alverson et al. 1994.*

lion metric tons at the turn of the twentieth century to a peak of 86 million metric tons in 1989 (FAO 1993; Hilborn 1990). In the mid-1980s, the catch from developing countries surpassed that from developed countries, with the developing countries' portion reaching nearly 60 percent in 1991 (Sissenwine and Rosenberg 1993). From 1989 to 1994, the world catch declined by 5 percent (Adams 1994), probably due in part to the fact that an estimated 70 percent of the world's fish stocks are overfished (FAO 1995a). The decline in some whale stocks due to commercial fishing preceded the trends in marine fish. For example, whalers had nearly exterminated Atlantic populations of the northern right whale (*Eubalaena glacialis*), bowhead whale (*Balaena mysticetus*), and Atlantic gray whale (*Eschrichtius robustus*) by 1800, with the latter now extinct (Norse 1993). Populations of probably all species of marine turtles have declined due to excessive exploitation for diverse turtle products, from meat and eggs to leather and ornamental turtle shells (Bjorndal 1981). Marine fisheries are also far from ecologically sustainable. Fisheries operations in coastal marine waters have caused wide-ranging losses and alterations of biodiversity at the genetic and ecosystem levels (Dayton et al. 1995).

Amidst this sea of unsustainability, however, are examples of sustainably managed stocks, including some that have been rehabilitated after overexploitation (e.g., see Butler et al. 1993; Sissenwine and Rosenberg 1993; Upton 1997). Globally, rehabilitation of overexploited stocks could raise sustainable yields by perhaps 20 million metric tons annually above the current world catch (FAO 1993).

The harvest for food from inland fisheries has also increased during the twentieth century, reaching more than 14.4 million metric tons in 1990, or approximately 17 percent of the marine take (FAO 1993). In contrast to the situation in marine fisheries, habitat degradation from pollution, dams, dredging, and other factors has probably been the major cause of decline in many inland fisheries, beginning most notably with the effects of the industrial revolution on streams and populations of Atlantic salmon (*Salmo salar*) during the twentieth century (Hilborn 1990). Excessive fishing pressure, however, has also caused serious declines in both salmon and strictly freshwater species (Crivelli 1992; Hilborn 1990). Other important examples of freshwater groups whose populations have been depleted during the twentieth century include several species of crocodilians (Messel, King, and Ross 1992), valued for their hides, and Amazonian river turtles (*Podocnemis* spp.) (Alho 1985), valued for their meat.

Wildlife Hunting for Meat and Recreation

Wild game is a crucial source of protein and an important factor in local economies of developing countries, although its market value is small compared with those of timber and fishery products. Africa depends on bush meat more than does any other continent. For example, game meat provides 75 percent of the protein consumed in Liberia, with an estimated total value of $42 million (Anstey 1991), and in Equatorial Guinea, it constitutes 50 percent of the protein consumed in urban areas (Fa et al. 1995). In Gabon, where the average person consumes 17.2 kilograms of game meat per year, the estimated value of game meat passing through formal markets is $3 million, and its estimated value in rural consumption is $21 million (Steel 1994). In Côte d'Ivoire, the value of game meat eaten each year (about 80,000 metric tons) is estimated to be $200 million (Lamarque 1995). Furthermore, the production of meat from domestic livestock has dropped over the past two decades in several African nations, and thus the demand for wild sources of protein may be increasing (Chardonnet et al. 1995). In Sarawak, Malaysia, each person consumes an estimated twelve kilograms of wild meat per year (Caldecott 1988). J. G. Robinson and K. H. Redford (1994) estimate that the half million residents of the state of Amazonas in Brazil annually hunt and consume at least 3 million mammals, 0.5 million birds, and several hundred thousand reptiles.

In Sweden, the total value of meat obtained from hunting of wild game in 1987 was estimated at $61 million, most of which is represented by moose (*Alces alces*) (Mattsson 1990). In the 1970s and early 1980s, the annual harvest of more than 550,000 ungulates in the former Soviet Union produced meat and hides collectively valued at $40 million (Sokolov and Lebedeva 1989). Game meat, as well as fish, is still of critical importance as a source of protein and income to the people of northern Canada (McTaggart-Cowan 1995). In Canada's Northwest Territories, for example, about 2.5 million kilograms of game meat are harvested annually, with an estimated value of $25 million in the mid-1980s (Usher 1987).

As in the case of fisheries, recreational hunting of terrestrial game is a major commercial enterprise. Where both meat and recreational value are components of hunting, the economic value of recreation dominates. Of 6.6 million cervids (e.g., deer and elk) harvested in 1985 in Canada, the United States, and Europe, no more than 3 percent were taken by subsistence hunters (Gill 1990). In Sweden, two-thirds of the economic value of all game hunting in 1987 was recreational and one-third was for meat (Mattsson 1990). Recreational hunting is a major economic activity throughout Europe and North America. In Germany in 1985, hunters spent $559 million harvesting game whose meat was valued at $123 million (Gill 1990). In the United States in 1991, hunters spent $12.3 billion (U.S. Department of the Interior 1993). The economic importance of big game hunting in southern Africa and parts of Asia is also well established. Revenues from safari hunting in Zimbabwe have risen steadily over recent years, totaling $1.5 million in 1995 on communal lands under the CAMPFIRE Program (CAMPFIRE Collaborative Group n.d.) and $3.1 million in 1990 on private commercial ranches (I. Bond 1993). In Pakistan, trophy fees in excess of $25,000 per markhor are generating significant revenues for local people (Johnson 1997).

High demand for game meat has frequently led to depletion of game populations. Game meat was marketed in cities throughout the United States during the nineteenth century, leading to major declines in many species of birds and mammals and extinction of the passenger pigeon (*Ectopistes migratorius*) (Shaw 1991; Tober 1981). The most striking declines in wildlife populations due to meat markets during the twentieth century are probably among the large to medium-sized mammals of tropical forest regions (Alpert 1993; Bodmer et al. 1997; Caldecott 1988; Fa et al. 1995; Fitzgibbon, Mogaka, and Fanshawe 1995; Glanz 1991; Steel 1994).

One of the most sustainable forms of consumptive use during the twentieth century, under varying degrees of commercialization, has been recreational hunting of large game, particularly cervids, in North America and Europe. Populations of all cervids except the caribou (*Rangifer tarandus*) (subject to mostly traditional harvesting) have increased since 1900, and barren-ground caribou numbers are now increasing in some areas as well (Gill 1990). Similarly, offtake from recreational hunting and commercial culling of big game in some regions of southern Africa is also sustainable (Crowe et al. 1997; Cumming 1989; Luxmoore 1985). The demands of the big game hunting market, however, may compromise some aspects of ecological sustainability (Geist 1995; Mcnab 1991; Teer 1997; chapter 6 of this book).

Harvest of Nontimber Plant Products

Apart from the use of forests for timber production and of grasslands for livestock grazing, multiple products from wild plants enter local, national, and international markets. Perhaps most prominent among these is the use of wild plants in traditional medicine. The current world market for medicinal herbs, most of which are apparently harvested from the wild, is estimated at $10 billion annually (Blass 1993). Traditional medicine is the basis of primary health care for approximately 80 percent of the population of developing countries, or some 3 billion people, and about 85 percent of traditional medicine involves the use of plant extracts (Farnsworth 1988). Nontimber plant products also occupy national and international markets for various other uses. Brazil nuts, with $33 million in sales in 1987, and palm heart (*Euterpe* spp.), with nearly $300 million in sales in 1993 in the region of the Amazon estuary alone, are two familiar foods harvested from the wild that have global markets (Clay 1997a, b). Another example of the diverse uses and markets of nontimber forest products is illustrated by Indonesia, which in 1986 earned $134 million from exports of rattan (*Calamus* spp.), resin, essential oils, kapok (*Ceiba pentandra*), and Indian fever bark (*Cinchona* spp.) for quinine (Sharma et al. 1992). In India, the estimated value of nontimber forest products exceeds $1 billion annually (Poffenberger 1990).

The diversity and significance of uses of nontimber plant products is often best understood at the local scale. In one study of sal (*Shorea robusta*) forests in West Bengal, India, 155 of 214 species recorded were used by local communities for food, fuel, fiber, fodder, medicine, construction, household articles, religious use, ornamental use, or recreation. Nontimber forest products accounted for 55 percent of the total income of forest fringe dwellers in West Bengal (Singh et al. 1997).

The importance of nontimber plant products is not limited to developing countries in the tropics. Berries and mushrooms, for example, are a staple in the diet of many Nordic countries. In Finland, sixteen species of wild berries are picked, with more than fifty kilograms per person per year harvested in some regions, and thirty species of wild mushrooms are eaten, with 68 percent of all households engaged in collecting them in one region (Salo 1995).

Little is known about the sustainability of most harvesting of nontimber plant products. For some, offtake is apparently sustainable without strong management because demand simply does not exceed productivity, as with berries in the Nordic countries (Kardell 1986; Salo 1995). However, there is mounting evidence that many nontimber plant products are being overexploited, even in highly traditional systems such as those of Southeast Asia (Edwards and Bowen 1993; Hall and Bawa 1993). Where strong national and international markets have developed, as for palm heart (Clay 1997b) and rattan (Peluso 1992), the often open-access nature and thus poorly regulated harvesting of nontimber plant resources has often led to major declines in their populations.

Livestock Grazing of Rangelands

Human use of livestock to convert largely unusable and unpalatable products—grass and browse—into something usable—food, milk, hides, and wool—represents perhaps one of the least sustainable forms of consumptive use, with widespread impoverishing effects on biodiversity. The United Nations Environment Programme estimates that 73 percent of the world's 3.3 billion hectares of dry rangeland is at least moderately desertified (Durning and Brough 1992). Overgrazing is estimated to have caused from 35 percent (WRI 1994) to more than half (Durning and Brough 1992) of the world's rangeland degradation

Other Forms of Consumptive Use

A miscellaneous array of other species uses, too diverse to cover adequately here, frequently occupies center stage in the use-it-or-lose-it debate. The most publicized have often involved the decline and endangerment of populations of charismatic species. Most prominent, perhaps, is the trade in elephant ivory and rhinoceros horn, mostly illegal, with a combined annual value of more than $50 million during the 1970s and 1980s (Hudson and Cumming 1989). Bear gallbladders and bile are important in traditional oriental medicine. Although estimates of the total trade are unavailable, in South Korea the import price of bile was $7,337 per kilogram in 1993 (Mills 1995). Also of importance in oriental medicine is tiger bone (as well as many other tiger parts), which often wholesales for more than $1,000 per kilogram (Mills and Jackson 1994).

Animal skins are also a significant product in international trade. The annual export of kangaroo skins from Australia, for example, oscillated between approximately 1.0 million and 1.7 million from 1982 to 1992, with the total 1992 value of raw and pickled skins at around $11.9 million. The harvest is from wild populations, and offtake is well monitored and regulated (de Vos 1997). Seal skins, as well as seal oil, have been another important product in global trade for more than two centuries (Bonner 1997). CITES reported 44,810 cat skins and more than 9 million reptile skins in international trade in 1990 (WRI 1994). The largest trade in reptile skins involves species of *Tupinambis* from Argentina, averaging 1.83 million skins annually from 1983 to 1987, and monitor lizards (*Varanus* spp.) and several species of snakes from Asia and Africa. These earn millions of dollars in foreign revenues for the exporting countries. Almost all trade in lizard and snake skins depends on wild, unmanaged populations, with the effects on these populations largely unknown (Whitaker 1997).

Among the most economically valuable reptile products are the so-called classic crocodilian skins (skins from the American alligator, *Alligator mississippiensis*, and several species of crocodiles, *Crocodylus* spp.), whose CITES-documented trade increased from 65,245 skins in 1984 to 245,082 in 1991. American alligators accounted for 60 percent of the 1991 trade. Documented trade in the less valuable skins of *Caiman* species was around 800,000 in 1988 and fell to roughly 340,000 skins in 1991, though the latter figure may be twice as large if undocumented trade is included (Luxmoore and Collins 1994). Almost all production in crocodilian skins comes from a combination of wild-caught animals and animals captively raised from wild-collected eggs and hatchlings (Roth 1997a). In the state of Louisiana in the United States, the largest producer of alligator skins, the total value of wild- and captively produced skins was $19 million in 1990, before world prices dropped as a result of overproduction. Alligator meat has also grown in popularity, with its total value exceeding $5.5 million in 1992 in Louisiana (Joanen et al. 1997). Strong markets for crocodilian skins, beginning near the end of the nineteenth century, led to overharvesting and rapid declines of many populations. Concerns about the fate of several species led to strong protection efforts beginning in the 1960s and the recovery of populations of economically important species such as *C. porosus* in Australia (Webb and Manolis 1993), *C. niloticus* in southern Africa (Hutton 1996), and *A. mississippiensis* in the United States (Joanen et al. 1997). Populations of several species, however, continue to be overexploited (Luxmoore and Collins 1994).

Ornamental animals and plants, both dead and alive, also support major commercial markets. Marine ornamental fish are captured almost entirely from wild populations. Global figures are lacking, but imports of marine ornamental fish into the United Kingdom in 1996 were valued at $1.3 million (Davenport 1997), and exports of ornamental fish from the Philippines during the first eight months of 1995 totaled $800,000 (Pratt 1996). Although the low volume of trade in some species is probably sustainable, the use of explosives and cyanide to capture both ornamental and food fish has caused extensive damage to coral reefs in the Philippines and in many other regions of Southeast Asia (Pratt 1996).

CITES registered nearly 1 million birds in international trade in 1990 (WRI 1994). From 1982 to 1986, trappers of wild birds in Neotropical countries earned an estimated average of $6.6 million per year from the average annual export of 280,000 parrots, while intermediaries received $22.8 million per year on the sale of the same birds. The estimated retail value of these birds was $320 million, or more than $1,000 per bird (Thomsen and Brautigam 1991). In 1986, the declared value ($1.4 million) of bird exports from Guyana to the United States accounted for roughly 0.6 percent of Guyana's total export revenues. The declared value of bird imports to the United States from all countries in 1987 totaled about $8.5 million (Thomsen, Edwards, and Mulliken 1992). A highly specialized niche in wild bird trade is the supply of birds for falconry. In the Middle East, an estimated 3,000 birds, valued at $15–$30 million, are traded annually (Cade 1997).

Finally, CITES reported more than 900,000 cacti and nearly 1.3 million orchids in international trade in 1990. The sustainability of harvest from wild populations of these plants is largely unknown (WRI 1994).

Summary

The activities of prehistoric humans presaged modern patterns of wild species use. Whether the first colonization of new lands by prehistoric peoples or the development of new markets for wild species in modern times, such changes are often accompanied by significant ecological adjustment and biotic impoverishment. Overexploitation has caused extinctions in both prehistoric and modern times. Commercial harvest programs often go through a stage of overexploitation before more sustainable management practices are implemented. Many of the causes of overexploitation by prehistoric colonists of new lands were probably similar to some of today's causes—lack of knowledge about newly encountered resources; open-access systems that prevail before territorial claims are well established; a frontier mentality wherein the resource is viewed as inexhaustible or there is always thought to be more just over the horizon; and the "have-to-eat-today" principle, associated with extreme poverty, which precludes a long-term view of resource management.

The early history of human colonization and use of the world's ecosystems suggests that commercialism is not a prerequisite for unsustainable wild species use. The simple demand for more resources by a mushrooming human population, exacerbated by weak social incentives and controls for sustainable resource use and ever more efficient harvest technologies, must be considered a more fundamental cause of overharvesting. Currently, humans directly and indirectly appropriate some 40 percent of the primary production of the earth's land area (Vitousek et al. 1986). Although only 8 percent of global aquatic primary production is required to sustain the world's fisheries, this figure jumps to 24–35 percent in areas of ocean upwelling, along continental shelves, and in freshwater systems (Pauly and Christensen 1995). Human consumption is therefore bound to have an overwhelming effect on the earth's wild living resources regardless of whether it occurs through commercial trade or through each of the planet's 4.5 billion people toting a gun, net, or chainsaw to harvest his or her own.

Commerce, however, often at corporate and international scales, is now a central feature in almost all forms of consumptive use. Market forces tend to dominate the levels and patterns of consumptive use of wild species and, more broadly, the ways in which natural ecosystems are managed, if they are maintained at all. Any progress toward more ecologically sustainable management of the earth's biosphere will require a clear understanding of these market forces and of the broader set of economic and social factors that influence them.

CHAPTER 3

Economic Issues

Insofar as there is a dominant belief in our society today, it is a belief in the magic of the marketplace.

—George Soros (1997)

Opinions vary widely within economics regarding the ability of the marketplace to bring about efficient and sustainable use of natural resources (Sagoff 1995; Tietenberg 1996). Neoclassical economics focuses on the microeconomic issue of getting prices right through efficient operation of the marketplace. This, it is argued, will result in resources being allocated in the best possible way for total human benefit. An efficient market, then, should internalize the external environmental costs of resource use and ensure that depletion of a resource is reflected in its price. The proper role of government is to ensure that the full social and environmental costs of consumption are reflected in prices and not ignored by consumers. Thus, the price of timber should include (internalize) the costs to society (the negative externalities) of resulting soil erosion and downstream siltation. Similarly, government subsidies should not maintain artificially low consumer prices for increasingly scarce resources, since higher prices will reduce consumption and stimulate the production of substitutes (Nordhaus and Tobin 1972; Vincent and Panayotou 1997). A corollary of this view is that incentives for people to act for the common good are best achieved through private ownership of natural resources (Anderson and Leal 1991; Smith 1981).

More broadly, neoclassical economics views the resource base as essentially limitless because, if markets operate efficiently, technological progress can substitute for resource depletion. Resource shortages will be automatically corrected through market-mediated changes in resource allocation—through changes in resource production and consumption. There is therefore no need for policies that place limits on levels of consumption of a given resource (Nordhaus and Tobin 1972; Vincent and Panayotou 1997).

The relatively new school of ecological economics also recognizes the importance of optimal allocation of resources engendered by efficient markets and the importance of well-defined property rights. However, it views these conditions as far from sufficient for maintaining sustainability. Equal or greater weight is given to the optimal *scale* of resource use—the total amount of resources consumed. This is a macroeconomic issue. Scale is considered crucial because human-made capital provided by technology is viewed as incapable of substituting for depleted natural capital—the stock of natural assets that yields a flow of resources and services (Daly 1991b). In short, it is argued, any goal of sustainability must incorporate the finite carrying capacity of the earth, determined by the level of resources and services provided by the earth's stock of natural capital and the capture of solar energy through photosynthesis. Further, although markets may efficiently allocate resources at any given scale of consumption, they will not lead to a sustainable global level of consumption. The thoughtful guidance of government, acting on behalf of current and future societal interests, is seen as essential for setting the proper scale of use. Sustainable growth, in terms of a continual increase in resource consumption, is considered an impossibility (Bromley 1991; Costanza, Daly, and Bartholomew 1991; Daly 1991a, b; Farber 1991).

How to manage questions of both allocation and scale of resource use are key issues in examining the link between CCU and biodiversity conservation. Will efficient markets alone lead to sustainable levels of offtake? To ecological sustainability? Where is the thoughtful guidance of government needed?

Failure of Markets to Internalize Costs and Benefits

The failure of markets to internalize the benefits of biodiversity and natural ecosystems and the costs of their use is a widely recognized problem (McNeely 1988; Swanson and Barbier 1992). There are two key aspects to this problem with regard to CCU:

1. The negative effects of CCU on the societal benefits of biodiversity are not fully reflected in prices established by conventional markets, and there are inadequate market mechanisms to compensate those affected. For example, the price of lumber generally does not include the cost of sustainably managing the forest from

which it is harvested. More broadly, it does not reflect the costs that increased runoff and siltation may impose on downstream fishing communities, and even if it did, mechanisms to reimburse downstream communities for lost revenues are generally lacking. In short, revenues from CCU seldom cover the costs of sustainable offtake or ecological sustainability.

2. Those values of biodiversity and natural ecosystems that are not based on CCU, such as watershed protection, carbon sequestration, and aesthetic appreciation, are poorly captured by conventional markets. Thus, there are limited and inefficient mechanisms for beneficiaries of these amenities of biodiversity to pay for their maintenance. Further, because these values are largely considered to be public goods, individual beneficiaries have a limited incentive to pay even if mechanisms for payment exist. Under the limitations of conventional market systems, CCU, by default, emerges as the most productive use of the natural ecosystem, and it is left to shoulder, for better or worse, most of the burden of biodiversity conservation.

Economists have created a lexicon to deal with these market problems and values. *Externalities* refers to those values not captured by markets. The uncompensated negative effect on downstream communities created by logging of a watershed is a "negative externality;" the reforestation of a watershed that benefits downstream communities is a "positive externality." Those who benefit from positive externalities but do not pay for them, either because there are no mechanisms to pay or because they refuse to do so, are called "free riders."

Economic Values of Biodiversity

The two aspects of market failure previously noted are part of the broader problem of estimating the diverse economic values that biodiversity and natural ecosystems provide to society. A system for classifying these economic values according to the type of human benefit they provide is presented in figure 3-1. At the broadest level, benefits can be divided into use and nonuse values. Use values are further subdivided into three categories: (1) Direct-use values are those most readily captured by markets. They include two forms of use—consumptive use, as already defined, and nonconsumptive use, such as nature

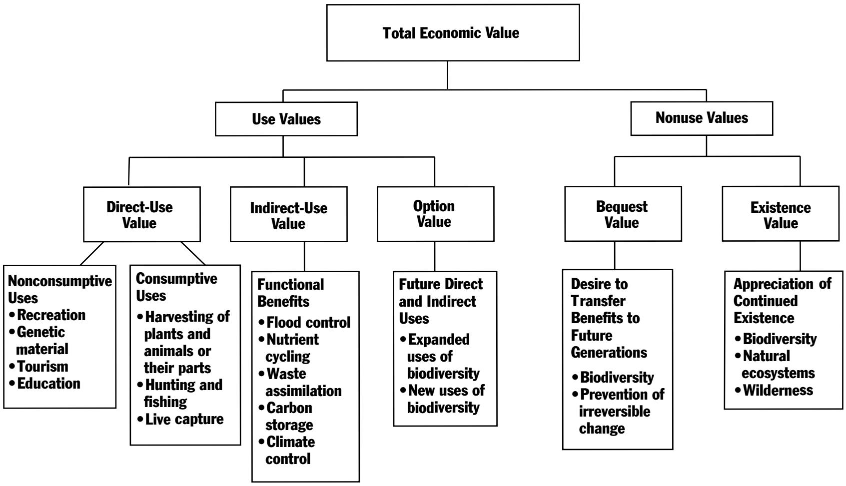

Figure 3-1. *Economic values of an ecosystem.* Sources: *Atlantic Coastal Action Program, n.d.; Barbier 1992; Pearce and Moran 1994.*

tourism and the provision of genetic material. (2) Indirect-use values, as in the foregoing watershed example, generally include the functional services provided by ecosystems and are in fact often referred to as functional values. (3) The option value accounts for an actual or potential resource user's desire to retain the option to benefit in the future from either the direct or indirect use of biodiversity. Nonuse values account for two additional categories: (4) the bequest value is based on the desire to pass on the resource to future generations; (5) the existence value is based on the knowledge that the resource simply exists. Thus, consumptive use is only one, albeit usually the most visible, of several values that can be attached to biodiversity and natural ecosystems. A challenge lies in how to better estimate non-CCU values, and once estimated, how to give them visibility and incorporate them into decision making about resource use so that an ecosystem's value does not depend exclusively on its marketable wild species commodities.

A key feature in understanding CCU and its role in biodiversity conservation is the disproportionately high tangibility of direct-use values, particularly CCU values, to owners of natural areas compared with the total economic value the natural areas provide to society at large (figure 3-2). Stated another way, resource owners who maintain biodiversity provide broader public benefits at private expense. The reason is that there are few mechanisms to pay owners for the other values (indi-

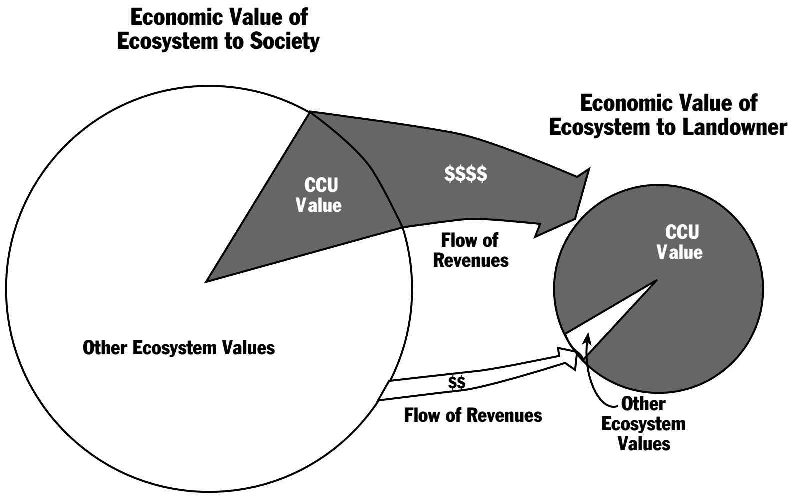

Figure 3-2. *Relationship between economic value of CCU and economic value of other natural ecosystem values of a unit of land in terms of their importance to society and to the landowner.*

rect use, option, bequest, existence) of the land or water they manage. Consequently, rather than managing for biodiversity, resource owners manage land and water to produce those direct-use values for which markets exist. These may be products and amenities based on natural ecosystems, such as timber and nature tourism, or agricultural commodities from an ecosystem converted to domesticated production.

However, what is good for individual producers may impose costs—negative externalities—on the rest of society. In a hypothetical freely functioning market, the individual who clear-cuts a watershed has no incentive to consider those who use the water and fisheries resources downstream. Even less tangible to this individual are the costs to society in terms of lost global benefits that the forest ecosystem provides—biogeochemical cycling, carbon sequestration, habitat for birds that control crop pests, potential pharmaceuticals from wild plants, and a diversity of wild plants and animals that have existence value for people around the world. Even though these societal benefits also accrue to the individual resource owner, their loss due to cutting of the forest will generally be small for the owner compared with the gains to be made from selling the timber and potentially converting the land to agriculture. This market failure arises not from the functioning of markets, but from the fact that there are no markets at all for these societal benefits. They are "missing markets" (Pearce and Moran 1994).

On a global scale, market failure and missing markets accentuate trends that lead to biodiversity loss. At best, if the CCU of a natural ecosystem is financially competitive with alternative uses of land and water, such as agriculture, the ecosystem is maintained. The effects on biodiversity in this case, as explored in chapter 6, range from minor to severe. At worst, revenues from CCU are not competitive with alternative uses and the natural ecosystem is converted to domesticated production systems or other uses with generally devastating consequences for biodiversity. In either case, increasing compensation to the resource owner for non-CCU ecosystem values should increase incentives to manage for biodiversity. In the first case, such payments may tilt the financial scale toward leaving the natural ecosystem intact rather than converting it to other uses. In the second, a more diverse and natural ecosystem may result because management efforts are not focused exclusively on the production of wild species commodities.

Forest management is a prime example in which because of the ease of calculating the monetary value of timber and the difficulty of quantifying nontimber values, timber harvesting emerges by default as the best and most productive economic use of the forest. P. P. Appasamy (1993, p. 258) illustrates this situation in India, where nontimber forest products used by poor people "do not enter the system of national accounts, which results in the undervaluation of the forest wealth of the country." M. Gillis (1992a, p. 130) notes that because the "assignment of monetary values to the protective services provided by forests . . . is much more difficult than for productive services, . . . in dozens of nations from Southeast Asia to Latin America to Africa, the owners of property rights to the natural forest have placed far heavier value on the productive rather than the protective resources provided by the forest." The failure of governments to incorporate resource depletion and the externalities of resource use into national accounting systems makes this problem systemic in national and international policy making.

Ecosystem Valuation: Beyond Traditional Markets

Economic valuation of ecosystems is an attempt to economically quantify the various use and non-use benefits of biodiversity. Such valuation is difficult, however, because many of the goods and services provided by natural ecosystems and biodiversity are impossible to reasonably quantify. Consequently, there is widespread aversion to attempting such calculations and to incorporating them into decision making

(Bingham et al. 1995). Nevertheless, economic valuations are indicative of the values of some indirect, nonmarketed benefits of wild species and ecosystems. They can be a tool for improving ecosystem management by identifying free riders and estimating the monetary value of benefits they receive and by expanding the range of values considered by decision makers (Freese 1997a). Pearce and Moran (1994) provide an overview of how ecosystem valuations are conducted and of their potential and shortcomings.

The relative importance of CCU compared with other economic values of ecosystems is indicated by the economic valuations of the four different ecosystems presented in table 3-1. CCU contributed 2–27 percent of the total ecosystem value in these cases. None of the valuations,

Table 3-1. *Results of Economic Valuations Conducted in Four Ecosystems*

NET PRESENT VALUE OF ALL FORESTS, SWITZERLAND, 1990S
(DISCOUNT RATE NOT INDICATED)

	U.S.$, BILLIONS	%[a]
Sustained timber production[b]	0.37	5
Game production[b]	0.01	0
Recreation	1.33	18–19
Protection function	2.91–3.41	42–46
Species diversity	2.33	31–34
TOTAL	6.95–7.45	

NET PRESENT VALUE OF FOREST IN KORUP NATIONAL PARK, CAMEROON, 1980S
(8 PERCENT DISCOUNT RATE)

	U.S.$, MILLIONS	%
Sustained timber production[a]	5.05	27
Replaced subsistence production	1.50	8
Tourism	2.09	11
Genetic value	0.74	4
Watershed protection of fisheries	5.82	32
Flood control	2.43	13
Maintenance of soil fertility	0.82	4
TOTAL	18.45	

(*continues*)

Table 3-1 (*continued*)

NET PRESENT VALUE OF DIRECT USES OF LOUISIANA WETLANDS, UNITED STATES, 1983 (8 PERCENT DISCOUNT RATE)

	U.S.$/HA	%
Commercial fishery[b]	783	13
Fur trapping[b]	373	6
Recreation	114	2
Protection from storms	4,731	79
TOTAL	6,001	

TOTAL ECONOMIC VALUE OF L'ETANG ESTUARY, NEW BRUNSWICK, CANADA (YEAR NOT INDICATED; 7 PERCENT DISCOUNT RATE)

	CAN$	%
Shellfish harvesting[b]	40,909	2
Recreation	1,184,047	66
Waste disposal	250	0
Option value	525,993	29
Existence value	39,768	2
TOTAL	1,790,967	

Sources: For net present value of all forests, Switzerland: Rauch-Shwegler 1994, in McShane and McShane-Caluzi 1997. For net present value of forest in Korup National Park, Cameroon: Ruitenbeek 1989. For net present value of Louisiana wetlands, United States: Costanza, Farber, and Maxwell 1989. For total economic value of L'Etang Estuary, New Brunswick, Canada: Atlantic Coastal Action Program n.d.

[a] Percentages do not necessarily add up to 100 because of rounding.
[b] CCU value of ecosystem.

however, covered all economic values, and thus the contribution of CCU to total economic value is probably overestimated.

Such ecosystem valuations indicate that the failure of markets to capture the spectrum of biodiversity values poses a major challenge for biodiversity conservation. Recognition of these broader values is beginning to guide the development of economic and resource-use policies that benefit biodiversity. In some cases, national-level policy changes have occurred as the public and decision makers have become more aware of functional and nonuse values. In Switzerland, for example, where wood production represents only 5 percent of the total forest value, forest management policies are being restructured to give greater weight to functional and nonuse values of biodiversity (McShane and McShane-Caluzi 1997). In other cases, global environmental markets are being created to provide a means for biodiversity beneficiaries to pay biodiversity managers for maintaining these other values (Pearce and Moran 1994). Many such markets involve payment from developed countries to maintain biodiversity in developing countries. Given that most of the world's biodiversity and threats to it are in the tropical regions of developing countries, markets for payments from the developed world to the developing world are especially needed. Mechanisms for such payments now exist, and new ones are being created. An example is the German Development Assistance Agency's efforts to negotiate, as a stakeholder interested in biodiversity, a buyout payment to the government of the Central African Republic to stop logging in the Dzanga-Sangha Dense Forest Reserve (Telesis 1991). Debt-for-nature swaps are another form of North-to-South transfers for setting aside lands of high conservation value. The Global Environmental Facility provides a mechanism for developed countries to fund the conservation of areas important for biodiversity conservation in the developing world (Pearce and Moran 1994). Nonprofit conservation organizations provide a quasi-marketplace through which donors contribute to (pay for) the conservation of biodiversity.

Such mechanisms, however, appear to fall far short of the potential for much larger payments for biodiversity conservation. For example, a recent valuation study indicated that citizens in the United States would be willing to make a one-time contribution of $2.18–$2.82 billion for rain forest conservation (Kramer, Sharma, and Munasinghe 1995), yet the actual level of giving is only a fraction of that.

A look at one particular functional value of tropical forests, carbon sequestration, provides another insight into the potential importance of non-CCU values. Based on an estimate by S. Fankhauser (1994) of $20 of global environmental costs due to global warming for every metric ton of carbon released, Pearce and Moran (1994) estimate that conversion of a primary forest to agriculture would cause damages of about $4,000–4,400 per hectare. If we apply the lower end of this estimate to the 126,000-hectare core area of Cameroon's Korup National Park, the park's total carbon sequestration value equals $504 million, more than twenty-seven times the estimate of the park's total net present value in table 3-1 (see next section for definition of net present value). Global markets for the carbon sequestration value of forests are developing in various ways, through both intergovernmental agreements and private sector initiatives (Pearce and Moran 1994).

Despite our ability to approximate economic values for the societal benefits of biodiversity and to create mechanisms for payment, getting the world's beneficiaries to pay is another matter. Global biodiversity values that are public goods, such as carbon sequestration, the option value of new pharmaceuticals, biogeochemical cycling, and aesthetic appreciation, invite free riders. Why should I pay if there is a chance, if I wait long enough, that others will? Thus, we cannot rely on the private sector and markets to produce public goods such as biodiversity (Tietenberg 1996). This constitutes another major economic and ethical issue for the biodiversity conservation community.

Another issue is the considerable inertia in existing political and economic systems that caters to well-established economic stakeholders at the expense of new ones, particularly those stakeholders concerned with public goods and services provided by natural ecosystems. A. M. Rivlin (1993, p. 257) observes that "Forces opposed to policy change tend to have the resources and organization to capture the political system, while the beneficiaries of change do not." Decision making is thus bound by "the tyranny of the status quo" (Vatn and Bromley 1994). The result is that decision makers, whether local landowners or government policy makers, continue to make land-use decisions based largely on commodity values and inimical to the broader societal benefits of biodiversity and ecosystem services (Freese 1997a).

I will return in chapter 8 to the questions of non-CCU biodiversity values, the degree to which they are important in decision making regarding biodiversity conservation, and how to incorporate them into decision making. At this point, it should be clear that conventional markets fail to capture the full array of biodiversity values of natural ecosystems, and as a consequence, decisions about the management of natural ecosystems tend to be driven by commodity-based interests.

Valuation of Future Benefits

Under traditional market mechanisms, the economic value of a wild species or its products is greatest for current benefits and decreases for future benefits. The rate of decrease over time is called the discount rate. Thus, whereas \$10.00 received today for a cubic meter of wood is worth \$10.00, at a discount rate of 5 percent, \$10.00 received for that wood ten years from now is worth only $6.14 today. And $10.00 received for that cubic meter of wood in 100 years is worth only $0.08 today. At a discount rate of 10 percent, waiting 100 years to harvest and receive $10.00 for the wood yields a present value of just $0.001. The discounted sum of all future returns on a harvested resource is termed the net present value. Clearly, the higher the discount rate, the less likely one will forgo revenues from harvesting a resource now for future revenues from later harvests (Barbier 1992; Randall 1981). As K. N. Lee (1993a, pp. 191–192) contends, "Efficient markets tend to allocate resources to the current generation at the expense of later ones," and "If resources are traded in markets, the value of conserving them for ecologically significant lengths of time is set by markets, not by biology; usually, biological conservation turns out to be worth very little."

C. W. Clark (1973) demonstrated the perversity of the discount rate in wild species management by showing that for species with a low annual growth rate (e.g., whales, rhinos, and trees), the rational economic decision is to harvest the entire population and put the revenues in an investment with a higher annual return. If, on the other hand, the timber in the previous example grows (through reproduction and growth of individual trees) at a rate greater than 5 percent or 10 percent (depending on which discount rate is used), it might be worth forgoing the harvest today because the rate of increase in harvestable wood would more than make up for the discount rate. Thus, "high interest rates encourage transformation of natural ecosystems toward faster-growing species or other uses of land" (Norgaard 1995, p. 453).

The discount rate is, at its most basic level, equal to the prevailing rate of interest or return on capital investments (Norgaard and Howarth 1991). However, for the individual or company deciding about when and at what level to harvest a wild species commodity (or other resource, for that matter), uncertainty about the future generally pushes the discount rate higher (Ciriacy-Wantrup 1952). Uncertainty in the commerce of wild species comes in many forms, all of which tend to place a premium on exploiting resources and earning revenues now rather than later. Consumer preferences and demands for wild species products fluctuate over time. If there is reason to believe that the demand for a species product may greatly decline or disappear, the owner of the resource may decide that the best economic strategy is to harvest the entire population now. Wild species commodities based on luxury uses, and particularly fads, are especially prone to this. The collapse during the early 1980s of the market for fur seal skins, from $40 to $9 per pelt, is a recent example (MacKenzie 1996). The issue of product demand is treated in more detail later in this chapter.

Uncertainty about the future supply of a resource will also increase the discount rate. The future supply, in this case the size and accessibility of a wild species population, can be highly uncertain for several reasons. First, as explored more fully in chapter 5, boom-and-bust population fluctuations are exhibited in many economically important wild species resources, from seed production of tropical trees (Clay 1997a; Janzen and Vásquez-Yanes 1991) to marine fish stocks (Francis 1997; Upton 1997; Wise 1991). The causes differ, but the effect is the same when the harvester is faced with highly migratory species—many birds, large ungulates, marine species—whose populations may be here today and gone tomorrow. In effect, such mobile species are an open-access resource, since no one can effectively own or control access to them.

Unstable or poorly defined social and political conditions also create high discount rates. For example, civil strife and the possibility of major policy changes regarding land use will create uncertainty about land tenure and resource use rights. Such conditions do not encourage long-term resource stewardship.

Finally, overcapitalization and the "ratchet effect" commonly seen in marine fisheries (Ludwig, Hilborn, and Walters 1993) place a premium on maximizing current harvests and revenues at the expense of future returns. Fishers frequently expand their fishing capabilities by buying new gear during periods of good fishing. When fishing yields decline, government subsidies may help the fishers continue their operations and meet payments on the new gear until fish populations rebound (if they do). The rebound may lead to another round of equipment capitalization and resulting new debt. Where harvesting has been unsustainable, debt and poverty often preclude the investments necessary for conversion to sustainable practices (see the section on poverty in chapter 4).

Forestry, because of the relatively slow growth of trees and the length of time between planting and harvesting, is often cited as a case in which economic logic based on the discount rate undermines sustainable practices. R. J. A. Goodland and colleagues (1990, p. 310), for example, note that logging concessionaires in tropical forests are able to turn a profit in large part because they follow unsustainable logging practices. Even tropical tree plantations on prime sites, they point out, generally yield only 3–7 percent increments in annual volume. Thus, "the rate of return on investment at today's high discount rates makes long rotation forestry and 20- to 30-year tree plantations economically less attractive than faster yielding alternatives, such as a two-year rice project."

I. Bateman (1992) provides a useful illustration of this problem and how it can affect alternative land-use decisions in a temperate forest region, the Highlands of the United Kingdom. Although the Highlands region has the lowest agricultural land values in the United Kingdom and 44 percent of the land could be reforested, forestry is uncompetitive even at low discount rates. At a preferential 3 percent discount rate, forestry produces a better commercial return than agriculture on just 28 percent of the land, and at a 5 percent discount rate the area is reduced to 11 percent. Although the incorporation of agricultural subsidies into these calculations would improve the position of forestry, Bateman concludes that "Accepting the validity of the discounting means that the gap between the economic value of timber production and that of agriculture still seems significant."

The Tamshiyacu-Tahuayo Communal Reserve in Amazonian Peru provides another example of how discount rate considerations may affect sustainability. Commercial tree species, palm fruits, and some wildlife species have been overexploited in the reserve, and thus a no-use recovery period has been recommended. Over the short term (0–5 years), stopping use would cause an estimated 21 percent decline in income for the communities compared with continuing the current unsustainable system. Over the long term (6–30 years), however, the transition to a sustainable system would yield an estimated 25 percent increase in income because the recovery of previously overharvested species would provide higher yields than are currently possible. If people of the reserve have a discount rate higher than 12 percent, which is likely, given their poverty, the unsustainable system is more financially profitable. The communities have mitigated the short-term costs by staggering the implementation of management programs—instituting fisheries management in 1984, reducing timber extraction in 1989, and initiating game management in 1994 (Bodmer et al. 1997).

A positive discount rate means that effectively no weight is given to resource conservation or human welfare beyond a generation into the future. Yet in forest management, for example, there may be two or three human generations between the time of investment in the resource and the time when it yields benefits, and during that interim what is valued in the forest may change. G. L. Baskerville (1995) points out, with regard to forest management in New Brunswick, Canada, that conventional approaches to estimating the net present value assume that the present and future generations agree on what is of value. He concludes that "An economic valuation carried out in the 1930s would have decided not to protect fir stands from fire or insect attack, since these stands had little value to the generation making the decision. As time has shown, the generation of the 1950s valued the same fir stands highly and has invested heavily in crop protection."

The perverse effect of the discount rate is another case in which markets, left unfettered, do not lead to sustainability and biodiversity conservation. Further, the effects of discount rates on investments pose a dilemma for conservationists. Although high discount rates work against the sustainability of wild species use, high rates may also curb investments in development projects that may be destructive to biodiversity. These conservation implications of the discount rate have attracted considerable attention in resource economics and among conservation and development agencies regarding how to manipulate the discount rate to favor sustainability and intergenerational equity. However, if we are concerned about passing on the full array of benefits and options provided by biodiversity to future generations, then trying to change the way people invest in resource use through ad hoc manipulation of discount rates is the wrong game (Norgaard and Howarth 1991). R. B. Norgaard and R. B. Howarth (1991) suggest that if we are concerned about the distribution of welfare across generations, then we should transfer wealth by protecting biological diversity, educating the young, and developing methods for the sustainable management of renewable resources. Such transfers may occasionally be good investments as well, but they should not be evaluated as investments, and the benefits they provide to future generations should not be discounted. In short, because biodiversity is unique and irreplaceable, money cannot substitute for it now with the option of recreating the lost biodiversity in the future.

A broader and deeper ethic than society has displayed to date will be required for society to transfer today's biodiversity wealth and knowledge to the future. Developing such an ethic will be no small challenge, particularly in light of R. M. Solow's (1991, in Levin 1993) comment that "There is something faintly phony about deep concern for the future combined with callousness about the state of the world today." The problem of the discount rate is indeed its failure to fully value future benefits of biodiversity, but that problem is a function of society's failure to fully value future beneficiaries. No economic theory of incentives can help society correct that failure.

Economic Incentives and Property Rights

At the most basic level, a piece of land, a body of water, or a wild species that occupies the land or water is subject to one of two possible types of ownership: It is owned or it is not owned. If no one owns or controls access to the land or water or the resources therein (whether legally or via de facto ownership), it is an open-access resource and all who wish to harvest the resources are free to do so. Under such conditions, there is no incentive for any one individual to forgo harvesting part of a population because any part of the population that remains can be immediately harvested by other individuals. Nor is there an incentive to invest in management of the resource because others may reap the dividends of the investment (Randall 1981).

Under conditions of open access, the population of a commercially important species will continue to be harvested as long as there is an immediate profit to be made. The harvest will stop only when the population becomes so scarce that revenues no longer cover the costs of harvesting and marketing and the value of the harvester's time. For poverty-ridden harvesters, for whom the opportunity cost of labor is extremely low or zero (i.e., there is no alternative employment opportunity), or where heavy capital investments in harvesting capability have already been made (i.e., dedicated sunk capital), there may still be a financial advantage in pursuing the last stragglers of a depleted population. For species that are easily located and harvested and whose commercial value rises with increasing scarcity, such conditions may readily push the population to endangerment or extinction.

The label of no ownership best applies to marine organisms of the oceans beyond the 200-mile exclusive economic zones of the world's nations. No individual, community, corporation, or government owns the resources of the high seas. However, open-access conditions can also prevail under conditions of ownership when the owner fails to control, whether purposefully or not, access to the resource. For example, although national governments claim ownership of most of the world's forests, many forest resources are subject to open-access exploitation. In some cases, the government may exercise its ownership rights over the land and its resources by closely controlling the harvest of trees, but it may exercise no control over the harvest of nontimber forest products which results in a de facto open-access regime for the latter resources. In other cases, an owner of private land may simply be unable to exercise control because of inadequate resources for doing so and because the state fails to help enforce the owner's rights. Also, private ownership may apply to only some resources in a given area. In many countries, for example, an individual may own land and the vegetation on it, but wild animals belong to the state.

In short, it is only when ownership is clearly defined and enforced that economic incentives become a potentially viable conservation tool (Tietenberg 1996). Owners come in many forms—individuals, corporations, communities, local and national governments—each with distinct implications for sustainability of species use and biodiversity conservation (see chapter 4).

Consumer Demand and Sustainability

Consumer demand for a wild species commodity affects both the price received for the product and the quantity and quality of the product harvested. The price, quantity, and quality of wild species commodities are key factors affecting ecological and socioeconomic sustainability. The relation between consumer demand and sustainability is thus a key management tool.

The Effect of Price on Sustainability

How does price or, more precisely, profit (price minus costs) affect ecological sustainability? Figure 3-3 provides a simplistic model of the key issues regarding this question. Figure 3-3 (a) illustrates the case in which ownership of the resource (both for the land or water and the species therein) is established and there is a threat of economically attractive alternative uses of the land or water. The alternative use may mean total alteration of the ecosystem, as when a wetland is converted to a parking lot, or it may be more specific with regard to particular parts of the ecosystem, as when a pest species (e.g., elephants or geese that eat crops) is extirpated (i.e., an alternative use of a forest with elephants is a forest without elephants). Figure 3-3 (b) illustrates the case in which either there is open access to the resource, with no controls on harvest level, or ownership exists but there is no threat of an alternative use. In figure 3-3 (a), low prices confer low sustainability because the natural ecosystem is economically uncompetitive with an alterna-

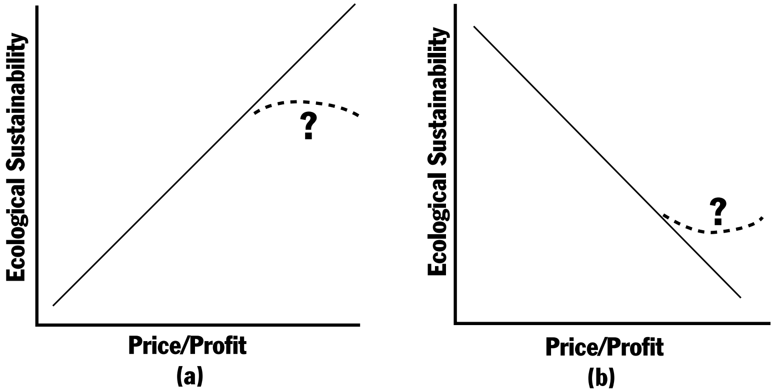

Figure 3-3. *Relation between the price of, or profit from, a wild species commodity and ecological sustainability under three different conditions: (a) established tenure rights, where CCU is important for offsetting the opportunity costs of alternative land or water uses; (b) open access regardless of the threat of alternative uses, or established tenure rights where there is no threat of alternative uses.*

tive use of the land or water. That competitiveness increases, however, as profits from the natural ecosystem increase.

Low profits may also preclude sufficient investment in research on and management of the target species, with an increased risk of overharvesting and bad management practices. Many nontimber forest products, for example, bring relatively low prices but are extensively harvested. The revenues from any one nontimber product are often insignificant, particularly relative to timber revenues, and thus owners (often government agencies managing public forestlands) ignore research and management needs for such species. Ignorance about the ecology and management of such common nontimber forest products as chicle (*Manilkara zapota*) from Mesoamerica (Kiernan and Freese 1997), Brazil nuts from Amazonia (Clay 1997a), and ginseng (*Panax quinquefolium*) from North America (Anderson et al. 1993; Nantel, Gagnon, and Nault 1996) are three examples.

Further, low prices may facilitate overharvesting when the harvest of the low-value species is incidental to, or a condition for, the harvest of high-value species. This is probably most problematic in marine fisheries, where many species in the bycatch are retained for sale rather than discarded. For example, in Singapore, 48 percent of the landed bycatch were low-value species used for direct consumption or processed into fish paste and other products (Abdullah, Ismail, and

Rahimah 1983, cited in Alverson et al. 1994). In the African and Asian shrimp fisheries covering the east-central and southeastern Atlantic and the western and eastern Indian Ocean, more than half of the estimated 2.5 million metric tons of bycatch is retained by artisanal fishers (Alverson et al. 1994). If these species had no value, they would be returned to the sea, and though there is high mortality in discards, at least some survive. If they had high value, greater attention might be given to their management. A variation of this problem is found in the Plan Piloto Forestal management project in Quintana Roo, Mexico. Contracts for the purchase of high-value woods such as mahogany (*Swietenia macrophylla*) have included an obligation to purchase low-value species in an effort to add value to the forest, but revenues are insufficient to provide adequate research and population monitoring of these species, with the result that some may be unknowingly overharvested (Kiernan and Freese 1997).

Higher prices per organism harvested under secure ownership can increase sustainability by allowing greater investments in managing the species and by offsetting the opportunity cost of alternative land and water uses. Higher prices and, if harvest costs do not increase too much, high profitability per organism harvested often result only when the organism is scarce relative to demand (G. Child 1996). How to manage scarce resources often presents a dilemma. A common tactic is to prevent use, though this then nullifies use of the scarcity value of the species as an economic tool for conservation. B. Child (1996, p. 372) argues that "If resources are 'endangered' they are scarce, and scarcity is the essence of value. People conserve valuable things."

Unusually high prices, however, may also jeopardize ecological sustainability for several reasons. First, the financial returns of harvesting become so high that clandestine harvesters are willing to assume high risks and thus enforcement of property rights becomes difficult. Widespread poaching of protected wildlife populations, from elephants in Africa to black bears (*Ursus americanus*) in North America (Rose and Gaski 1994), is a well-known example of this problem. Depending on local circumstances, this problem may be assuaged by the fact that higher profits enable the property owner to invest more in protection.

Second, government agencies may weaken, ignore, or nullify the tenure rights of communities (often traditional communities) in order to provide access, if not outright ownership, to more politically powerful interests that wish to mine resources and reap profits for reinvestment elsewhere. In Thailand, for example, P. Hinton (1987) found that nontraditional commercial fishers, with new technologies and international markets for their catches, had begun to encroach on the domain of traditional fishing communities. He concluded that "Coastal fishermen have become resigned to such intrusions, and to the loss of traditional fishing grounds, for government enforcement of a one mile coastal strip devoted to village fisheries has been ineffectual."

Third, if the price of a wild species commodity is viewed as unusually high and thus likely to fall again, the rational economic decision (i.e., to maximize long-term profits) for the resource owner may be to harvest the entire population now. This decision may also reflect concerns by the presumed resource owner that his or her property rights may become more difficult to enforce because high prices will attract and embolden clandestine harvesters.

Fourth, high profits may provide the incentive and financial ability for the owner to invest in specialized production of the commercial species. Thus, rather than being overexploited, the target species becomes superabundant. A plantation forest managed for a high-value species is a common example. Specialization generally leads to ecosystem simplification and the erosion of biodiversity at the site (see chapter 6).

Under conditions of open access (figure 3-3 (b)), as in much of the marine realm, or where tenure rights are poorly defined, as in many tropical forest regions, higher prices simply invite an increasing number of people to enter the harvest and to spend more going after a dwindling population, a sure recipe for overexploitation. Poverty exacerbates this situation because the opportunity cost for an individual's time is so low.

Overharvesting can create a self-escalating environment of scarcity. As the resource species becomes increasingly scarce its commercial value increases, which enables greater investments in going after the remaining population. The rapid buildup of the marine fishing fleet in developing countries during the rapid price increase in fish products over the past two decades may be a result of this effect (FAO 1993). I. E. Strand, V. J. Norton, and J. G. Adriance (1980) point out that in general, if revenues are rising faster than costs, fishers will continue to fish despite declining abundance. H. F. Upton (1997) examined commercial harvesters' response to declining striped bass populations along the Atlantic coast of the United States and concluded that "Revenues were maintained because price increased as product decreased, and thus a strong incentive remained for fishermen to continue fishing as stocks declined." During a thirty-year period in the Alaska salmon fishery, rising prices made it possible for twice as many fishers to be employed, catching half as many fish (FAO 1993). In Zimbabwe, B. Child (1996, p. 372) avers that "The alienation of people through, for example, colonial legislation, meant that at the very time population growth made resources critically scarce and therefore valuable, access to them became open. . . . This, more than anything, is the cause of the massive conservation problems facing us today."

The probability that a de facto open-access regime will exist may in large part be a function of resource profits. When profits from natural resources are low, an open-access regime for their extraction may reflect an explicit or tacit decision by the government that it is not worth investing in management of the resource by better defining and enforcing property and use rights (Bromley 1991; Swanson 1994). Although profits from the mining of the resource are worth capturing, it may be financially impractical for government to control the extraction process. Thus, increasing profitability may eventually provide the incentive and financial means for government to establish and protect property rights, thereby potentially leading to improved management. Yet even where demand and harvest of a resource are high, as long as productivity exceeds demand, open access may be an explicit policy that is compatible with sustainability. This would seem to be the case in Sweden and Finland, where, despite the massive amounts of wild berries harvested, the level of productivity is so high that the governments maintain a policy of open access for harvesting, even on private lands (Salo 1995).

Figure 3-3 (b) also suggests that despite secure tenure rights, increasing profitability from CCU will cause a decline in ecological sustainability where there is no competing alternative use of the land or water and where the target species itself is not a pest and thus subject to control or extirpation. For example, except for the more localized effects of mining, much of the boreal forests of Canada and Siberia are not threatened by conversion to alternative land uses; the region is unsuitable for large-scale agriculture or livestock grazing. Nor is the spruce (*Picea* spp.) of this region a pest species. Thus, higher prices for spruce lumber provide no benefit in terms of outcompeting alternative land uses (there are none) or for saving the spruce from extirpation (it is not a pest). Without these benefits, the four potential problems noted in the discussion of figure 3-3 (a) dominate the relationship between price/profit and sustainability.

Green Markets: Linking Consumer Demand with Producer Incentives

"Green marketing" provides a mechanism for internalizing some of the costs of sustainably managing CCU and enables consumers to pay for non-CCU values of biodiversity through the marketplace (Johnson and Cabarle 1993). The premise is that products identified as ecologically (or environmentally) safe (that is, "green labeled") will have an advantage in the marketplace over those that are not, and that the resource owner will be rewarded for the additional costs that ecological sustainability often entails. The reward may come from an increased market share that a green-labeled product commands, and/or from a premium that consumers are willing to pay for a green-labeled product. A key proviso is that a significant part of revenues realized at the retail level must make it back to the producer as a socioeconomic incentive for, and to cover the additional costs of, good management. J. Clay (1997b, p. 311) notes that "The values generated by the harvesting, processing and sale of wild species are traditionally captured far from the ecosystem that produced it. Green marketing is one way to return at least a portion of the value added to wild-harvested products to the producers themselves."

Green marketing can thus be a mechanism for helping meet the necessary conditions for all three types of sustainability—offtake, ecological, and socioeconomic. At a minimum, green markets may encourage producers to practice sustainable offtake. However, consumers of green-marketed products generally assume that the natural environment from which the species is taken is also being conserved—the essence of ecological sustainability. In this case, the additional revenues a green-labeled CCU product generates represent in part the otherwise unappropriable non-CCU values of biodiversity (Swanson 1994). The utilized species is simply a market vehicle for capturing some of these global values of biodiversity—the positive externalities of good management. Finally, to the extent that green marketing provides additional profits to the resource owner, it increases the socioeconomic sustainability of the natural ecosystem relative to competing uses of land or water.

A small green premium paid by the consumer can be very large for the producer. For example, harvesters of Brazil nuts in Amazonia receive approximately $0.07 per kilogram, or just 0.3 percent of the specialty store price of $22.00 per kilogram in the United States (Clay 1997a; see table 4-1). A premium of $0.10 per kilogram charged to consumers for ensuring that the Brazil nuts they buy are sustainably managed would add 0.45 percent to the specialty store price. If, however, one-half of this premium were passed on to harvesters in the forest, their income would increase by 58 percent from $0.07 to $0.12 per kilogram (Freese 1997a).

The resource owner or harvester will seldom be able to receive the full value of the premium because green labeling involves two other costs in addition to those directly assigned to on-the-ground management of the resource. First, costs are incurred in handling and tracking the green-labeled product through the marketing chain to keep it differentiated from nongreen products. These costs can be especially large at the early stages of product development, when volume of the green-labeled product is small. However, a study of forestry certification in Indonesia indicates the costs of certification may be more than offset by increased revenue capture and by savings from improved inventory control in the marketing chain that green labeling requires (Cabarle et al. 1995). The second major expense is the need to verify for everyone involved, particularly the consumer, that the product is in fact "green." This requires that an independent third party certify the sustainability of both the resource management practices and the chain-of-custody handling of the product.

The greatest certification efforts to date have occurred in the wood products industry. Environmental groups in Europe and the United States issued "good wood" guides in the late 1980s to educate consumers about linking the purchase of wood products with sustainable forestry. The first independent environmental certification program for wood products was the Smart Wood program, launched in 1990 by the Rainforest Alliance, a United States–based nonprofit organization (Cabarle et al. 1995). Many other certifying organizations, both nonprofit and for profit, have since been established. Meanwhile, wood product companies wasted no time in employing often spurious claims of sustainability as a marketing tool for an increasingly environmentally conscientious public. A 1991 study in the United Kingdom of more than 600 companies that marketed wood products as sustainably produced found that only three were able and willing to substantiate such a claim (Read 1991). Although no mass market for certified wood products has yet developed, significant niche markets, particularly in North America and Europe, have emerged (Lyke 1996).

The need for universal standards and accountability led to the creation in 1993 of the Forest Stewardship Council (FSC). The primary role of the FSC is to act as an independent, objective body that can certify certifiers, a crucial task if consumers are to have confidence in the veracity of certification claims. To do this, the FSC has established guidelines for certifiers and a set of principles and criteria for natural forest management. The principles and criteria address not only ecological factors but also social and economic considerations, such as respect for indigenous people's rights and the social and economic well-being of forest workers and local communities. The key criteria with regard to the CCU-biodiversity link are that management should conserve biodiversity, unique and fragile ecosystems, and the ecological functions and integrity of the forest (Cabarle et al. 1995). The FSC system provides a credible, voluntary mechanism that allows consumers to selectively purchase wood products with confidence that they were taken from well-managed forests. An important benefit of the voluntary nature of the program is that the system is immune from regulations of the World Trade Organization (WTO), which makes it difficult for governments to impose environmentally oriented restrictions on internationally traded products.

The unknowns of forest certification loom fairly large at this early stage of development. Certification programs based on different standards continue to proliferate independently of the FSC, at national, international, industrial, and government levels (Kiker and Putz 1997; Lyke 1996). Without coordination of efforts, this trend threatens to further confuse both producers and consumers, and some in the forest products industry will promote less rigorous certification programs to undermine the effectiveness and growth of the FSC movement. Financial and technical challenges exist at various levels: (1) Funding mechanisms for certifiers remain uncertain. Ideally, if objectivity and the public's trust are to be maintained, funding should come from sources that are independent of those being certified. (2) Many questions remain in terms of how to consistently and accurately define ecologically sustainable management practices for natural forests ranging from the tropics to the boreal region, for diverse types of wood products, and under diverse social, economic, and market conditions. (3) A dilemma exists in setting standards flexible enough to lure timber companies into experimenting with certification, yet rigorous enough to ensure consumer confidence. If a large timber company agrees to initiate sustainable management on 10 percent of its land, with the intent to increase that percentage if successful, should that 10 percent be certified despite unsustainable practices on the rest? (4) Tracking certified products through the market, from forest manager to consumer, presents formidable challenges. (5) Although it seems likely that consumer demand for green products will continue to grow in the developed world, how fast it will grow and how much of a premium consumers will be willing to pay for such products remain big unknowns.

How consumer demand develops will be determined in large part by the level of exposure and education consumers receive regarding the need for ecological sustainability in the production of wild species commodities and by how well and with what confidence green-labeling programs enable consumers to distinguish good from bad management (Kiker and Putz 1997). In contrast to the largely bottom-up evolution of forest product certification, the first green-labeling initiative for marine products began with one of the world's largest buyers and producers of frozen fish products, Unilever, when the company signed an agreement with the World Wide Fund for Nature (WWF) in 1996 to create the Marine Stewardship Council (WWF and Unilever n.d.). Executives of Unilever obviously believe that consumer awareness of and concern about overfishing are, or soon will be, sufficient to give the company a marketing advantage in being the first major fish product company to occupy the green-market niche. It remains to be seen whether this will be the case and how the rest of the marine fisheries industry will respond.

Recreational Hunting and Fishing: A Special Case of Green Markets

Recreational hunting and fishing occupy a unique position in the CCU sector. The value that consumers attach to most wild species commodities, whether fish in the marketplace, hides made into boots, or boards at the lumberyard, is based primarily on utility. The environmental setting from which the product is harvested becomes a factor only when green labeling is applied. In contrast, the value hunters and fishers attach to a recreationally killed duck or trout is not exclusively, or even primarily, based on the act of killing the animal or having the dead duck or trout in hand. If it were, hunters would go to game ranches in Texas to shoot exotic African ungulates in small corrals, and there would be a major market for large commercial fish-rearing tanks, where fishers could be guaranteed to land big fish. Markets do, in fact, exist for variations of both of these scenarios, but there is a much larger market serving hunters and fishers who want to pursue game in more or less natural ecosystems. Aldo Leopold (1936, p. 394) captured this notion with the theorem that "The recreational value of a head of game is inverse to the artificiality of its origin."

Those who pay to hunt and fish in a natural environment are expressing a consumer preference similar to that of consumers who buy certified timber products. Just as consumers of certified timber are paying for biodiversity conservation in the area where the timber is harvested, hunters and fishers are paying, in large part, for the opportunity to pursue game in natural surroundings. The primary difference is that the consumer of certified timber is potentially paying for various non-CCU values of biodiversity—functional, option, bequest, and existence. In contrast, much of the economic value of recreational hunting and fishing is probably based on nature tourism, with the harvest of game a value-added activity. Hunters in particular argue that it is the low-impact, high-value feature of recreational hunting that makes hunting revenues so important in justifying the conservation of wildlife habitats (Morrill 1995).

This does not mean that all hunting and fishing, or all management for hunting and fishing, is "green" (i.e., ecologically sustainable). Although offtake may be sustainable, recreational hunters and fishers display a wide range of attitudes, understanding, and values regarding what is "natural." Catch-and-release fishing, in which barbless hooks are used and fish mortality is low, might be the extreme case of nonconsumptive use being the main value being paid for. Where these values are important for maintaining instream flow and contaminant-free waters, the conservation benefits are important. Yet, as examined in chapter 6, fishers often are not discriminating regarding the provenance of the species they catch. Streams around the world have been stocked with nonnative sport fish, to the detriment of native species and the naturalness of otherwise wild streams and lakes. Waterfowl hunting has generated revenues for wetland conservation, but native biodiversity has been compromised by species transplants and manipulation of wetlands for increased waterfowl production (Callaghan, Kirby, and Hughes 1997; see also chapter 6).

Although certification programs have been suggested for recreational hunting in both southern Africa (Lewis and Alpert 1997) and North America (Rasker and Freese 1995), neither the hunting nor the recreational fishing communities have undertaken any formal or widely accepted certification efforts. The seeds for such programs, however, exist. Safari Club International, for example, apparently encourages hunters to hunt in countries with good conservation programs (Lewis and Alpert 1997). Some states in the United States (e.g., Colorado and California) have initiated hunting programs that provide, through the marketplace, economic incentives for habitat improvement and maintenance for ranchers who follow state-issued habitat management guidelines. Seasons are lengthened and game quotas are increased for ranchers who follow the guidelines, thus enabling those in compliance to host more fee-paying hunters (Rasker and Freese 1995). In this case, the incentive works through the increased offtake and number of fee-paying hunters rather than through any premium paid by hunters in explicit recognition of good biodiversity conservation practices. The ecological sustainability of this approach depends on how compatible management interventions to increase game populations (an obvious landowner incentive in this case) are with the broader goal of biodiversity conservation.

Macroeconomic Policies Affecting CCU

National and international macroeconomic policies affect the CCU-biodiversity linkage in complex ways. Three areas of particular importance are free trade agreements, structural adjustment programs, and national accounting systems.

Free Trade Agreements

Free trade agreements, such as the World Trade Organization (WTO; formerly the General Agreement on Tariffs and Trade or GATT) and the North American Free Trade Agreement (NAFTA), have far-reaching implications for attempts to manage CCU for the benefit of biodiversity conservation. Proponents of free trade argue that stronger national economies, to the extent that free trade creates them, enhance public attitudes toward environmental quality, better enable countries to carry out natural resource conservation programs, and result in more environmentally benign production systems (World Development Report 1992; Yu 1994). Others contend that rather than encouraging sustainability, free trade simply encourages countries to mine their natural capital in order to attract industry (Johnstone 1995). H. E. Daly and R. Goodland (1994) express concern that unregulated trade will result in the world's economic growth more readily overshooting sustainable levels of resource use on a global scale.

Free trade poses significant barriers to internalizing the externalities of CCU. It is difficult under free trade agreements to impose standards and sanctions on exporting countries with low environmental standards (Ekins, Folke, and Costanza 1994; Preeg 1995), as demonstrated by the problems the United States encountered under WTO when federal government attempted to ban the import of tuna from Mexico because of high incidental catches of dolphins by Mexican tuna fishers. Further, under free trade, a country that internalizes environmental costs will be at a disadvantage, at least over the short term, compared with a country that does not do so because the prices that must be charged to cover the costs of sustainability will be higher (Daly and Goodland 1994). P. Ekins, C. Folke, and R. Costanza (1994, p. 7) contend that "If the potential environmental benefits of free trade are to be realized, trading rules, such as those developed by WTO, must recognize that environmental externalities are, in effect, environmental subsidies." WTO provides no check against, and in fact protects, international trade in wild species products that are harvested unsustainably. Further, free trade rules may discourage nations from participating in international environmental agreements such as CITES because such agreements often involve trade sanctions for noncomplying nations (Reed 1996). A solution to this would be for WTO to recognize sanctions imposed for legitimate reasons under other international treaties (Pearce 1995).

Another issue is that global free trade enables each country to specialize in its most economically productive living resources. Thus, it is argued, rather than each country developing a diversified agricultural and forestry sector to meet its own commodity needs, more open trade should better enable one country to specialize in rice, another in bananas, another in beef, another in timber, and so on, depending on which product can be produced most efficiently. Such homogenization of a country's production system is predicted to result in decreased biodiversity (Ekins, Folke, and Costanza 1994; Johnstone 1995; Norgaard 1987). Economic globalization over the past century has significantly reduced the diversity of crops produced within a given country. This trend includes a large decline in genetic diversity within crops, as demonstrated by the dramatic reduction in the varieties of rice grown in southern and Southeast Asia (Johnstone 1995). Unregulated global trade reinforces this trend.

A distinction needs to be made, however, between how specialization affects the diversity of commodities and how it affects natural ecosystems and their biodiversity. The ability to locate commodity production in the ecologically most productive regions of the globe may, in some circumstances, minimize the total area devoted to the production of that commodity and thereby relieve pressures for commodity production in areas of significant biodiversity (see chapter 6). Further, free trade rules generally reduce subsidies for production of agricultural commodities, which may tip the scales in favor of managing some lands as natural ecosystems. For example, in southern Africa, where wildlife is often more profitable than cattle under subsidy-free production systems (I. Bond 1993; Crowe et al. 1997), free trade–induced specialization toward wildlife-based industries should benefit biodiversity. Free trade may also reduce financial subsidies for the exploitation of natural resources, particularly timber, which should lead to more sustainable management (Yu 1994). Thus, the effects of free trade on biodiversity will be highly contingent on the ecological and socioeconomic conditions of a particular region.

Another aspect of specialization is that economic gains from trade tend to be overstated, particularly in less industrialized countries. Specialization entails a reorientation in production from diverse subsistence resources, which are generally ignored in economic accounts, to a few fully accounted-for commodities (Ekins, Folke, and Costanza 1994). This is an important subset of the larger trend that global free trade will further encourage. The management and marketing of the world's wild species commodities, from timber to fish, will increasingly shift from local resource owners and communities, with oversight (though often weak) by national governments to protect societal interests, to transnational corporations that are largely unaccountable to national governments. With transnational corporations now controlling 70 percent of the world's trade (Daly and Goodland 1994), and with their ability to alter on a global scale where they invest and what they invest in, special attention must be given to the effects of increased unregulated trade on the management of major wild species commodities.

Structural Adjustment Programs

In the 1970s, governments in the developing world accumulated extreme levels of debt due to excessive borrowing and a decline in their ability to make debt payments because of declines in revenues from international trade. Led by the World Bank, bilateral and multilateral development institutions responded by instituting economic structural adjustment requirements for debt-ridden countries as part of their lend-ing and development assistance programs. The main components of structural reform were trade liberalization, export-oriented growth, and sectoral reform. Sectoral reform emphasized free market approaches through a heavy dose of government deregulation and privatization of state-owned enterprises. The ability of government agencies to manage and monitor the use of natural resources was further eroded by required cutbacks in government budgets (Reed 1996).

Although the social and environmental effects of these reforms are complex and far-reaching (Frickman Young and Bishop 1995; Reed 1996), the principal effect on CCU has been in the forestry sector. The additional emphasis structural adjustment placed on export-oriented growth exacerbated the negative consequences of liberalized trade policies. In Cameroon, for example, the government identified timber as a major source of foreign currency earnings. Adjustment and trade reforms increased the profitability of logging relative to the production of other export crops, which might have been advantageous for biodiversity had sustainable forestry practices been implemented. However, despite a new forestry code, corruption and a lack of administrative capacity for enforcement, combined with incentives for increased forest exploitation, resulted in rapid deforestation and a loss of revenues from the forest (Reed 1996).

Trade liberalization also affected forests in Tanzania and Zambia, where it resulted in increased prices of imported petroleum-based energy sources. The response was to substitute fuelwood and charcoal as energy sources, which in turn contributed to increased deforestation. Concern about this outcome does not constitute an argument against getting the prices right for imported fuels, but rather underlines the need to be prepared to manage the effects of such change on other natural resources (Reed 1996).

Some structural adjustment measures in the forestry sector, such as increased stumpage fees for standing timber and longer concession periods, are expected to lead to greater investments in sustainable forest management (Frickman Young and Bishop 1995). Thus far, however, attempts to implement such positive incentives have largely failed. Also, in contrast to other government-owned sectors, outright privatization of state-owned forestlands has generally not occurred in the adjustment process.

Although the effects of structural adjustment on fisheries are less researched, the depletion of marine fisheries in Thailand was accelerated by adjustment policies that emphasized growth in export revenues (Reed 1992). This does not, however, appear to be a general trend in developing countries. Although the rate of fisheries exports from developing countries increased by 10.5 percent during the 1980s, this is one-half the rate of increase during the 1970s, before structural adjustment programs were implemented (FAO 1993).

As in the case of free trade, some goals of structural adjustment programs have the potential to benefit biodiversity conservation, but these benefits will not be realized unless environmental safeguards are more effectively incorporated into adjustment policies. D. Reed (1996, p. 342) concludes, based on a series of country case studies, that "Internalization of economic activities' social and environmental costs has been set back . . . during the adjustment period. Regulatory activities were relaxed, capture of resource rents fell, and enforcement programs were cut back or eliminated." He further notes that a central component of adjustment programs was the consumption of natural capital to finance macroeconomic imbalances and that resulting revenues have not been reinvested in maintaining or rebuilding natural capital stocks. No provision was made in adjustment programs to monitor or mitigate such effects on natural resources. Thus, he concludes, "The country studies provide no ground for affirming that this dimension of the adjustment process has moved the countries to a more sustainable development path" (pp. 348–349).

National Accounting Systems

It is well known that traditional national accounting systems tend to ignore depletion of natural resources, environmental contamination, and other environmental effects of development (El Serafy 1991), but how to resolve this problem remains an area of considerable disagreement. Standard measures of economic activity, such as the gross domestic product (GDP) and gross national product (GNP), are guided by the United Nations System of National Accounts (SNA) and are well-embedded measures of economic success. GDP and GNP are no longer seen as simply measuring economic activity, but have assumed larger, and misleading, roles as indicators of prosperity, welfare, and progress (Peskin 1991; Sheng 1995). How misleading these indicators can be is obvious when national disasters result in a jump in the GDP or GNP. For example, the recent earthquake in Kobe, Japan, in which more than 5,000 people died, 15,000 were injured, and estimated costs exceeded $200 billion, produced a positive economic effect in traditional national accounting systems (Lyonette 1995).

Much more common and pernicious, however, is that the ongoing depletion of natural capital—loss of forests, fisheries, and other wild species commodities as well as various ecosystem functions—is totally ignored in the current SNA. The costs of cleaning up a major oil spill or toxic waste site will result in a higher GDP because of increases in the production of cleaning material and equipment. In short, the greater the environmental disaster, the greater the contribution to the GDP. Clear-cut logging is counted entirely as generating national income; the value of other forest products and the various functions of the forests that are lost are not subtracted from logging revenues. Similarly, the various products and services of biodiversity and natural ecosystems that are lost as agricultural frontiers expand do not show up as a debit in the GDP (Sheng 1995). In Brazil, for example, the cost of timber depletion caused by agricultural expansion during the period 1971–1980 ranged from 46 percent to 98 percent of the total value added by agriculture (Hamilton et al. 1992). Accounting for the loss of other forest values would surely have resulted in a negative balance for most years. Figure 3-4 shows the results of a study in Costa Rica that calculated a partially adjusted GDP for the period 1970–1989 based on the depletion of fisheries (decline of principal populations in one area), forestland (loss of immediate and future timber harvests), and soils (loss of key nutrients). The depreciation in value due to depletion of these resources during the two decades analyzed exceeded $4.11 billion, equal to the country's average annual GDP. This may have

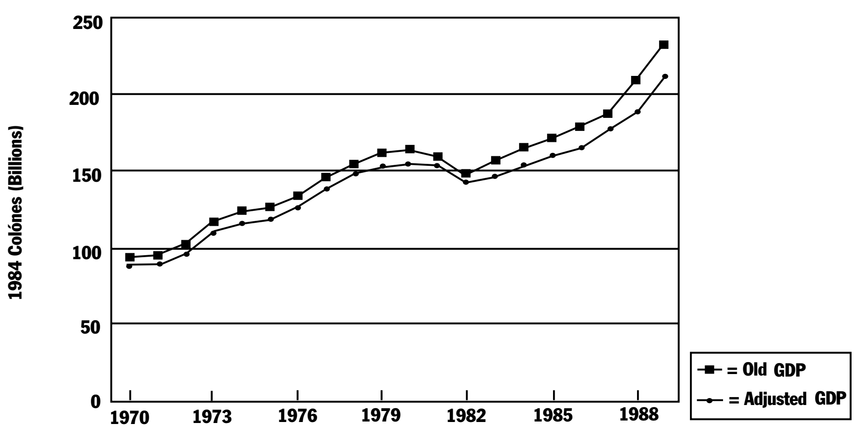

Figure 3-4. *Old gross domestic product (GDP) and gross domestic product adjusted to account for depletion of forestland, fisheries, and soil in Costa Rica, 1970–1989.* Sources: *Sheng 1995; Solórzano et al. 1991.*

reduced economic growth in the country by 25–30 percent over the period (Solórzano et al. 1991). The actual economic loss was surely much higher, as the calculations ignored the loss of many values provided by the three resources, such as the value of forests for wildlife habitat, tourism, and nontimber commodities.

Various international organizations and national governments are researching and experimenting with environmental accounting systems and with ways to reform the SNA (see the review in Sheng 1995). According to F. Sheng (1995), the primary objective of reforming the SNA is to identify the impacts of economic activities on nature, to estimate the monetary value of such impacts to the extent possible, and to adjust accordingly cost-benefit calculations to improve economic planning and decision making. Although estimating the monetary value of some impacts, such as the loss of non-CCU biodiversity values, presents significant challenges, these are surmountable and are no excuse for lack of reform (El Serafy 1991; Peskin 1991; Sheng 1995). The primary obstacle is a lack of political will on the part of many governments and major international institutions such as the World Bank. Concerns revolve around how an overhaul of the SNA will affect measures of economic success and progress and how the economic ranking of countries will be determined (Sheng 1995).

Summary

The economics of CCU is characterized in large part by the myopia and tunnel vision of markets vis-à-vis biodiversity values. Markets are myopic because discounting favors immediate consumption over resource conservation, and they have tunnel vision because only a fraction of the benefits provided by biodiversity, both current and future, are marketable. Market mechanisms are inadequate both in paying resource owners for the biodiversity benefits to society of good management (the positive externalities) and in ensuring that resource owners bear the ecological costs to society of bad management (the negative externalities). Thus, decisions about resource use and biodiversity conservation are generally dominated by commodity production goals, whether from natural or domesticated ecosystems.

Efficient markets are generally essential for sustainable management of wild species, but markets are often not efficient because property rights are not well defined. Where markets are relatively efficient and property rights are well defined, high discount rates may lead to overexploitation in some circumstances, whereas in others, specialization in production of the commodity species occurs. Both generally result in ecosystem simplification and biodiversity loss. Thus, there are few circumstances in which markets are sufficient in themselves to ensure that biodiversity is conserved.

The development of global markets for CCU products, combined with free trade agreements, raises additional questions about the ecological sustainability of CCU on a global scale. Although government management and policies affecting CCU have often been inept in protecting societal interests in biodiversity, enlightened government oversight is essential if CCU is to be compatible with, and at times a tool for, biodiversity conservation. Green-labeling programs for CCU products are emerging as a nongovernmental, market-based approach to protecting societal interests in biodiversity.

Attempts to estimate the monetary value of the societal benefits of biodiversity are useful in demonstrating to decision makers that non-CCU values of biodiversity are often much greater than marketable CCU values. Economic criteria, however, are only a subset of the information and considerations used in decision making. As A. Vatn and D. W. Bromley (1994, p. 145) caution, "There is nothing in economics in general—or in hypothetical valuation in particular—that can address the optimal level of air or water quality, or of land devoted to parks and wilderness." Although economic criteria may often be a useful tool in making decisions about the best use of natural resources, they are but one tool among several. Biodiversity conservation requires the consideration of cultural and ethical values that cannot, and should not, be subject to monetary measures of worth.

C H A P T E R 4

Social and Institutional Issues

There must be a mutuality of belonging: they must feel that the land belongs to them, that they belong to it, and that this belonging is a settled and unthreatened fact.

—Wendell Berry (1993), on what motivates people to care for the land

Markets and economies operate within a framework of social and institutional conditions. These conditions influence whether and how revenues from CCU act as an incentive or a disincentive for sustainability. Responsibility and authority for resource management ultimately rest with resource owners or, where ownership is lacking or not exercised, with resource harvesters. That responsibility and authority, however, are almost always tempered by government policies regarding resource ownership, management, and markets. Further, resource owners and harvesters differ greatly in how they are organized, their sociocultural environment and background, and the values and motivations they bring to CCU and ecosystem stewardship. These factors influence how CCU is managed and how ecological sustainability fares under different types of ownership.

Apart from resource owners, a diversity of other stakeholders have an interest in how the socioeconomic benefits and costs of CCU are distributed and how biodiversity is affected by CCU. These range from stakeholders for whom monetary returns are the primary factor to those who are motivated by nonmonetary social and biodiversity values.

Key Stakeholders and Their Motives

Human stakeholders in CCU and its sustainability can be broadly divided into four groups (Freese 1997a):

1. Those with a direct interest in the harvest of the wild species commodity and resulting revenues.

2. Those with values, economic and noneconomic, related to the target species, its ecosystem, and associated biodiversity that are affected by the externalities of its use and management.

3. Those concerned with humanitarian issues as broadly defined (Nash 1989), that is, human rights and social equity, particularly for local people, and the rights and humane treatment of animals.

4. Agencies and authorities that are outside the flow of benefits and externalities but that have management rights or substantial influence (legitimate or otherwise) concerning CCU programs and the ways in which the interests of the first three categories of stakeholders are addressed.

In addition, a biocentric perspective might argue for a fifth category—all nonhuman organisms—for which stakeholder interests could be defined. That perspective, however, is in part captured by stakeholders for biodiversity conservation and animal rights.

Direct-Use Stakeholders

Direct-use stakeholders are individuals, groups, or institutions whose property rights are clearly defined for the land being used and the product being harvested and sold (Randall 1981; van Kooten 1993). This group includes those in the market chain who benefit from the direct-use values of biodiversity—the resource owner or harvester, intermediaries, and end-use consumers. It also includes institutions, generally government agencies, that derive revenues from taxes and fees imposed on the product at any point in the market chain. Thus, direct-use stakeholders all have a direct involvement and interest in the flow of revenues resulting from wild species commodities. Except for the government agencies, all generally have ownership of the resource at some point. Monetary benefits resulting from CCU act directly on this group.

Direct-use stakeholders exert their influence primarily via conventional market mechanisms, though their actions are not necessarily always legal (e.g., illegal trade, payoffs to government regulators). The degree of influence varies widely, however, from poor local harvesters who may work under debt peonage and have no influence to multinational organizations with considerable economic and political clout.

Within this group of stakeholders, ultimate responsibility for sustainability resides in the resource owners and harvesters. If the incentives are not right for them to harvest the population sustainably and maintain the integrity of the natural ecosystem, it matters little what the other stakeholders do. The only recourse is the use of negative incentives in the form of sanctions against resource owners who do not use sustainable practices. Sustainability then becomes an uphill battle. Resource owners and others with use rights, particularly if they have historical ties to the area in use, may attach noneconomic values to local species and ecosystems, values that generally favor conservation and may temper perverse commercial incentives (Gadgil and Berkes 1991; Richards and Creasy 1996).

End-use consumers, however, are almost equally important. Although their stake is in the end-use benefit they receive from the resource, they may promote social, humanitarian, and nature conservation values through their buying choices. Consumers ultimately decide which wild species have monetary value in the market, and thus economic incentives for resource owners flow from the choices made by consumers.

Intermediaries in the market include buyers, processors, and wholesalers involved in value-added activities and market links between producers and end-use consumers. Their stake is primarily economic, and they exert influence by making decisions regarding trading and processing, by managing the flow of information among producers and between producers and consumers, and by lobbying for favorable trade policies.

Stakeholders Affected by Externalities of CCU

Various stakeholders may be affected by the environmental externalities of CCU. Those most directly affected may be individuals whose economic well-being is jeopardized, such as the fisher whose harvest and profit are reduced by upstream logging activities or the dive shop operator whose business is affected by depletion of reef fish by commercial fishing. This group of stakeholders expands quickly as one considers the effects of CCU in a given area on the indirect-use and nonuse benefits of biodiversity. In the broadest sense, the effects of CCU on functions such as carbon sequestration and on the bequest value of biodiversity confer stakeholder status to everyone, including future generations.

Markets through which such stakeholders can pay for the biodiversity benefits they receive from the ecosystem under management are frequently absent or inefficient. The principal exception is nature tourism, for which markets are relatively well developed and rapidly growing (Brandon 1996; Goodwin 1996). Efforts to create new markets and payment mechanisms for indirect-use and nonuse values of global biodiversity represent an attempt to incorporate these values, via monetary incentives, into management decisions (see chapters 3 and 8). Because such payments pale in comparison with commodity markets for wild species, stakeholders who value biodiversity often have little influence over how an ecosystem is managed. Therefore, these stakeholders must often rely on their ability to influence, through lobbying, votes, civil disobedience, and other mechanisms, government policies and international agreements that affect resource use, both in the private sector and by government agencies on public lands.

Humanitarian Stakeholders

Humanitarian stakeholders bring in a set of values that is largely distinct from those held by the previous two groups. As noted earlier, they can be broken down into two major groups: (1) those concerned with issues of social justice and human rights and (2) those concerned with the rights and humane treatment of animals. These values are generally shared in various degrees by the first two groups of stakeholders.

Again, markets for these values are poorly developed. Consumers have a limited ability to express a preference for socially responsible management practices. The consumer of fish and timber, for example, generally cannot tell whether harvesters are treated in a socially responsible manner by those in charge of resource management. Nevertheless, questions of human rights and social equity clearly shape the management of wild species and natural ecosystems. The diversity of national and international development groups and human rights organizations testifies to the prominence of these values. For CCU and biodiversity, such values are particularly important in two major respects: (1) they encourage the protection of natural ecosystems and of traditional uses for communities with historical ties to the ecosystem; (2) they encourage the equitable distribution of benefits from CCU, particularly for local people and communities that depend most directly on the ecosystem under management (Holmberg, Robèrt, and Eriksson 1996).

Animal welfare and animal rights stakeholders are often at odds with the concept of CCU as a conservation tool (e.g., Hoyt 1994; see also chapter 1). Animal welfare and animal rights values are based primarily on the humane treatment and protection of individual animals, in contrast to the biodiversity conservationist's focus on the maintenance of species and ecosystems. Animal welfare and animal rights organizations can have a major influence on the use and trade of wild animals, especially for luxury and recreational purposes. Their influence is wielded primarily through the shaping of consumer awareness and buying choices and through the political process.

Agencies and Authorities That Influence Use

This category of stakeholder cuts across the other three. It includes mainly elected officials and other politicians who are not directly affected by the economic benefits or externalities of CCU as well as government resource management agencies whose budgets do not depend on taxes or fees from CCU. A common motivation for members of this group is the need for approbation by their constituency (the public) if they are to obtain or stay in office or to survive as resource management agencies. Their success, power, and whatever other benefits they obtain from holding public office thus depend on how well they attend to the interests of, and thereby earn the support of, the other three groups of stakeholders. Payoffs to government officials, however, corrupt this system. Many government resource management agencies also depend on revenues collected through fees and taxes levied on CCU, and thus their motives are often similar to the motives of direct-use stakeholders. Resource agencies and political figures have at their disposal the diverse policy and regulatory mechanisms by which governments influence natural resource management.

An Example of Diverse Stakeholder Interests

The interest and influence of all four groups of stakeholders is illustrated by Mexico's Plan Piloto Forestal, a communal forest management program in the state of Quintana Roo on the Yucatán Peninsula. *Ejidatarios*, communal landowners, use the forest for various purposes—ranging from provision of materials for home construction, hunting, and other traditional uses to the sale of chicle and mahogany—for their primary source of income. The Plan Piloto Forestal is virtually the only legal source of mahogany for the mills and craftsmen of Mexico, whereas chicle has a foreign market and is currently purchased only by companies in Japan. Thus, both national and international market players and consumers have an interest in and influence over forest management in the program. Other stakeholders hold interests in other components of the forest's biodiversity. Half of the birds found in the forest during winter are migrants from the north, and thus the health of these forests is important both for the thousands of bird-watchers in the United States and Canada and for farmers and foresters who benefit from the birds' control of insect populations. The carbon sequestration value of the forest further enhances its practical global significance.

The overall biodiversity of the forest is of intrinsic value to scores of conservation groups in Mexico and other countries. Major funding for forest management and research, supported by both foreign bilateral assistance agencies and nongovernmental conservation organizations in Canada, the United States, and Europe, can be viewed as payment for these forest values. In addition, consumers of hardwood products manufactured from FSC-certified timber produced by the project are paying the *ejidatarios* for practicing ecologically and socially responsible management. Others exert their influence through national and international policy instruments, as in the recent attempt to list mahogany under CITES. Support from bilateral assistance agencies for greater say and independence by *ejidatarios* in the management of their forests and the marketing of its products also reflects external interest and influence in issues of social equity. Although animal welfare and animal rights concerns are not a major issue in the region, proposals to trap monkeys for biomedical research or to institute trophy hunting of jaguars (*Pantera onca*) would surely cause such concerns to be raised. Finally, the livelihood of the *ejidatarios* gives a direct stake in forest management to state- and national-level politicians in Mexico, particularly given the high political visibility of the project due to its socioeconomic reforms (Freese 1996; Kiernan and Freese 1997).

Revenue Distribution among Direct-Use Stakeholders

The distribution of benefits, both monetary and nonmonetary, that CCU and biodiversity provide has a major effect on whether and how incentives for sustainability operate (Lonergan 1993; Loomis and Ditton 1993). Among those who directly participate in the harvest and trade of wild species commodities, a crucial issue in benefit distribution is the proportion of the total retail price captured by the resource owner or harvester. Various social and institutional arrangements affect this distribution. Much of the debate surrounding the issue is confused by two distinct motives: one concerns the revenue distribution pattern that best serves the goal of ecological sustainability; the other concerns the pattern that best serves the goal of social equity. Although these are not necessarily convergent interests, in most cases social equity is probably a prerequisite to creating the socioeconomic incentives and sociopolitical stability that good resource stewardship requires.

Harvesters of wild species commodities commonly receive but a fraction of the price received by intermediaries and retailers. This is particularly evident with CCU products from developing countries that are marketed internationally and harvested clandestinely or under conditions of open or semiopen access (see table 4-1). Harvesters in Amazonian Brazil receive, at best, 1 percent of the retail price and 0.3 percent of the specialty store price of Brazil nuts in the United States. Poachers of black rhinoceros (*Diceros bicornis*) horn in Zimbabwe receive roughly 2 percent of the price paid for the horn by consumers of traditional medicines in Asia (Milliken, Nowell, and Thomsen 1993). And trappers of psittacines (parrots and other species of the family Psittacidae) receive only 1 percent of what wholesalers receive in the United States (Swanson 1992a). In none of these cases are large costs involved in processing of the product, although transportation, storage, and other costs may be significant.

T. Swanson (1992a) provides a useful analysis of where resource revenues are captured in the trade of parrots from Irian Jaya. Exporters there paid an average of $7 per bird exported to the United States. Using assumptions that would tend to overestimate costs, Swanson calculated that exporters and importers pay an additional $47 per bird for transportation, quarantine, mortality premium (i.e., cost of birds lost in transport), and subsequent storage costs. Thus, the total cost of acquisition of the surviving birds would be $54. Given an average wholesale price of $256 per bird, $202 represents a minimum estimate of the profit captured by middlemen per bird. In contrast, the price received by the trappers covers little more than the value of time and materials used in capture because trappers tend to be unorganized and there is generally open access to the resource, with the result that

Table 4-1. *Revenue Distribution among Stakeholders in the Market Chain of Three Internationally Traded Species*

Brazil Nuts (early 1990s)		
	U.S.$/kg	% of Specialty Store Retail Price
Paid to harvester	0.07	0.3
Paid to forest intermediary	0.26	1
New York spot price	2.64	12
Bulk retail price	6.60	30
Specialty store retail price	22.00	

Psittacines from Irian Jaya, Indonesia (early 1990s)		
	U.S.$/bird	% of Wholesale Price
Paid to trapper	2.57	1
Paid to trader	7.09	3
Paid to exporter	49.43	19
Wholesale price	256.00	

Black Rhinocerus Horn, Southern Africa (early 1990s)		
	U.S.$/kg[a]	% of Retail Price
Paid to poacher	154	2
Paid to Africa-based intermediary	250	3
Paid by wholesaler	800	10
Wholesale price	2,300	28
Retail price	8,075	

Source: For Brazil nuts: Clay 1997a. For psittacines: Swanson 1992a. For black rhinocerus horn: Milliken, Nowell, and Thomsen 1993.

[a]Median of price range given; assumes 1.5 kg/horn.

many trappers are competing to sell birds. Thus, virtually no profit is captured from the resource at the local level, and consequently there is little incentive or financial capability for either local resource users or local government to institute more sustainable practices. This is a vicious cycle because without sufficient financial returns from the resource, neither the government nor the harvester is motivated or financially able to develop stronger property and resource-use rights or better management practices. Swanson suggests that this situation characterizes most wildlife trade.

Revenue distribution systems in the tropical timber trade create largely the same distorted incentives. In this case, governments, which are proprietors of roughly 80 percent of the world's closed tropical forest, fail to impose sufficient fees, royalties, taxes, and other charges to capture the full value of the resource (Jepma 1995). For example, from 1971 to 1974, the government of Ghana captured just 38.0 percent of logging rents, and from 1979 to 1982, the government of the Philippines captured only 16.5 percent of logging rents and the government of Indonesia captured just 37.3 percent. Such losses of potential public revenues result in a large profit incentive for timber companies, which rush in to obtain lucrative timber concessions, resulting in accelerated deforestation (Repetto 1988). The problem is exacerbated by concessions that often run for twenty or fewer years, by political and contractual uncertainties, and by other risks that lead concessionaires to realize their profits as early as possible and that provide no incentive to invest in long-term forest management (Panayotou and Ashton 1992; Repetto 1989; Repetto 1993). R. Repetto (1993, p. 105) concludes that "Few governments of tropical countries for which data exist have succeeded either in limiting timber exploiters to a normal rate of profit or in capturing the value of the forest resource for the public treasury." The same is true, he notes, in many temperate countries, including Australia, Canada, and the United States.

Not only are governments not receiving the full value of their forest resources, but also the local communities that depend most directly on the forest often receive little or nothing from timber revenues. In many tropical countries, revenues from logging accrue primarily to the relatively wealthy rather than to the poor, who depend on the forest for much of their livelihood (Gillis 1992a). Rural poor are often the laborers who work in logging activities. A greater return of logging revenues to local communities is justifiable from the perspective of social equity, but unless these communities have ownership or use rights in the forest, it will do little to improve forest management. Indeed, to the extent that timber harvesting is an open-access activity, increased returns to local forest workers may lead only to more rapid logging.

Reforms in India suggest that forest management can improve where new or strengthened proprietorship by local communities is combined with a greater return to these communities of forest-generated revenues. After decades of top-down, centralized control of forest management in India, in the 1970s and 1980s local communities began to reassert control of their forests because forest degradation was depriving them of diverse forest products and of services such as watershed protection. In most cases, communities initiated this process without any outside policy change or assistance from government. Largely because of this grassroots movement, both the national and state governments have instituted major policy changes involving forest management, and significant forest recovery is occurring on formerly degraded lands (Poffenberger 1995; Singh et al. 1997). For example, in the state of West Bengal, a program of participatory forest management that began in 1972 with 1,272 hectares and 618 households had spread by 1994 to 390,919 hectares and 2,423 forest protection committees, each composed of many households (Singh et al. 1997).

Forest management in India now largely entails comanagement by state forest departments and local forest protection committees that represent the communities. A key issue of such comanagement is how revenues are shared. In 1990 a change in national policy gave exclusive rights to revenues from nontimber forest products to those villages that actively protect their forests (Poffenberger 1995). This can be significant, since in West Bengal, nontimber forest products account for an estimated 55 percent of total income in the forestry sector (Singh et al. 1997). In contrast to nontimber products, the government policy for poles harvested from sal, the primary timber product of India's forests, was that 25 percent of net proceeds goes to the villages and 75 percent goes to the government. M. Poffenberger (1994) cites a case in West Bengal in which the forest department's costs ran to 53 percent of gross, and thus the villages' share was only 12 percent. Not surprisingly, such a revenue-sharing scheme is a point of contention, particularly because it is not clear to local villages how the forest department calculates its overhead. As S. Singh and colleagues (1997, p. 87) note regarding participatory management in India, "A transparent system of equity in terms of sharing responsibilities and benefits is a *sine qua non* for effective community participation in the long run." Perhaps not coincidentally, communities in West Bengal have increasingly managed their forests for nontimber products rather than for sal poles, since the former entail no revenue sharing with government. Although the biodiversity effects of such management are little researched, a preference for a diversity of nontimber products rather than for timber from one species might be expected to yield a more biologically diverse forest. Of 214 wild plant species found in regenerating sal forests, 155 were used for food, fuel, fiber, fodder, medicine, construction, commerce, household articles, religious purposes, ornamentation, and recreation. Seventy of these plants were used frequently and regularly (Malhotra et al. 1991, cited in Singh et al. 1997). This tendency for governments to secure revenues from timber harvested from government forestlands while allowing revenues from nontimber forest products to go primarily, if not wholly, to the harvester is a widespread phenomenon in both developing and developed countries.

Zimbabwe's CAMPFIRE program, in which villages on communal lands have been given an increased share of revenues from safari hunting, exemplifies how under the right conditions such changes can greatly improve incentives for good management. S. Metcalfe (1994) reports on the experience of villages in the Kanyarira Ward, where, when revenues from safari hunting went to the government treasury, attitudes toward wildlife were negative and communities tended to seek more community services from government by, in part, encouraging new settlers. When revenues from safari hunting began to be distributed directly to the communities, wildlife came to be seen as economically beneficial, a proprietary attitude toward the ward's wildlife was rekindled, poaching decreased, and the community began to question whether it wanted new settlers, with whom wildlife dividends would have to be shared. A major issue in CAMPFIRE regards what proportion of the wildlife-generated revenues is taken by local government institutions—the district councils—as their share for helping coordinate and manage the program. The interception by government agencies of revenues derived from natural resources that are meant to be distributed to local communities remains a problem in many regions of Africa (Kiss 1990).

The CAMPFIRE program also reveals the significance of another major factor that affects both total revenues and revenue distribution. Sparsely populated wards tend to have better wildlife resources and thus generate greater wildlife-based revenues than do more densely populated wards (figure 4-1). This disparity is magnified by the fact that the revenues are distributed among a smaller number of households in the low-density wards (CAMPFIRE Collaborative Group n.d.). Higher population densities therefore result in significantly lower socioeconomic rewards per individual, which jeopardizes socioeconomic and ecological sustainability. This effect is the bane of a successful resource management program, as people immigrate to the area to take advantage of the socioeconomic opportunities.

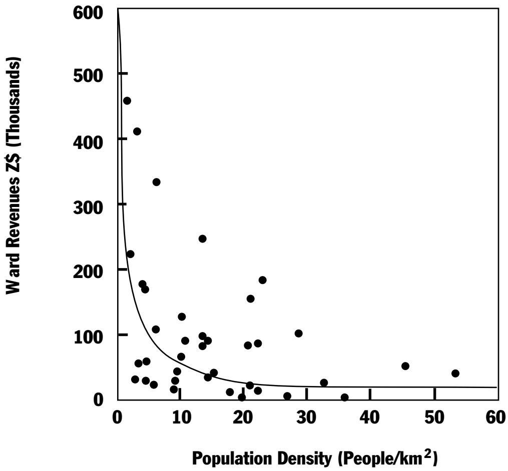

Figure 4-1. *Relation between human population density and revenues in the wards of the CAMPFIRE program, Zimbabwe, 1993.* Source: *I. Bond, pers. comm., 1997.*

The Balance between Local Resource Rights and Government Control

The management and harvest of virtually any economically significant wild species commodity involve a mixture of private enterprise and government involvement or oversight. The ecological and socioeconomic sustainability of CCU is greatly affected by specific components of this mixture. There is a need for the private sector to wield sufficient authority over, and realize sufficient rewards from, its investments that it is motivated to manage the resource sustainably, and there is a need for government oversight to protect societal interests. These two needs often conflict.

According to S. Jentoft (1989), the rationale for government involvement with regard to societal interests is threefold: (1) Government must protect society from the externalities of CCU. (2) Government must be involved to ensure equity among stakeholders. For example, government has a role in ensuring the fair distribution of fishing opportunities and incomes among individuals or communities that wish to fish in a certain coastal area. (3) Government must be involved for administrative reasons. Only government, it is argued, has the authority and resources needed to protect property rights and, at least for some resource species, to implement sound management.

Government involvement may range from outright ownership of the land or water and all its resources to a largely invisible presence that makes itself felt only when property or resource-use rights are illegally challenged. Government involvement may or may not be welcomed, depending on the stakeholder group. Landowners welcome government when it helps protect private property rights from intruders, but not when it imposes taxes or fees on a wild species commodity or when government regulations are seen as depriving the private sector of its right to use and manage a resource as it sees fit. Such regulations, however, often protect the broader societal interests of the other stakeholder groups. The concern of those with confidence in the market-based conservation approach is that excessive government fees or control may erode the private sector's incentives for good resource stewardship (Anderson and Leal 1991).

B. J. McCay and S. Jentoft (1996), in reviewing fisheries management, suggest that greater emphasis be given to the ecological attributes of the resource in order to define management structure. They propose (p. 244) that "To determine the proper domain of a management regime, one should attempt to map the decision-making regime onto ecosystem or fish stock boundaries. Thus, fish stocks with localized ranges could be subject to local-level management and migrating stocks could be the responsibility of regional, national, or transnational institutions." The difficulty with this prescription is that for many wild species, these ecological boundaries exceed the relevant social boundaries, as defined by cohesive communities of people that can most effectively cooperate to manage the resource. As M. W. Murphree (1996, p. 19) laments in examining community approaches to wildlife management in southern Africa, "This mismatch between social and ecological topography poses a major implementational problem." The solution may involve comanagement with other stakeholders, including adjacent groups of landowners and government agencies. The role of government in such situations, according to Murphree, is to confer "strong and enduring rights to units best placed to exercise them and a government commitment to provide the enabling policy and coordinative environment required for these rights to be translated into sustainable and productive systems of wildlife use."

The anadromous striped bass of the Atlantic coast of the United States exemplifies the management problems presented by migratory species and the need for a geographic scale of management authority that generally only government can provide. Severe declines of the striped bass population in the 1970s and early 1980s were largely attributable to three factors: (1) a strong demand for striped bass from both the commercial food industry and recreational fishers; (2) a lack of coordinated management throughout the coastal range of the species; and (3) the open-access nature of the fisheries. Incentives for individual fishers to reduce harvest were largely absent because they lacked ownership rights to fish stocks; individual ownership rights to such mobile resources are impossible to assign. Similarly, the fourteen jurisdictions (primarily states) through which the striped bass migrate were reluctant to introduce restrictions, since the interstate movements of bass precluded exclusive use and management rights by the individual states. Resolution of the problem and recovery of striped bass populations required regional coordination of regulation and enforcement through an interstate body, the Atlantic States Marine Fisheries Commission, and oversight of the process by the federal government. When the fishery was reopened in the state of Maryland, however, the number of fishers in the Maryland striped bass gill net fishery increased from 453 in 1991 to more than 900 in 1994. The number of fishing days in the striped bass gill net season was reduced by nearly half during the period because the quota was caught more quickly. Thus, although the total harvest is now well regulated, the open-access nature of the fisheries results in inefficient allocation of capital and resources. In addition, allocation issues remain between commercial food and recreational sectors of the fisheries (Upton 1997).

S. Jentoft and B. J. McCay (1995) reviewed the level and type of user participation in fisheries management in Canada, the United States, Spain, France, New Zealand, and the Nordic countries. In all cases, governments have a strong role in management of the fisheries. User group participation involved three alternatives: (1) At the minimum level of involvement, user groups may simply be informed of upcoming government decisions regarding management of the fisheries, often late in the planning process. Communication is one-way, from government to fishers. (2) Government may consult with user groups, as through public hearings or advisory groups, before setting regulations, though it may choose to ignore their suggestions. (3) Government and user groups may comanage the resource so that power, decision making, and management responsibilities are clearly shared between the two. These comanagement arrangements may develop by the central government's delegation of responsibilities to local institutions or by legal recognition of traditional management systems. Although Jentoft and McCay (1995, p. 227) claim that "Too much or too little involvement seems equally problematic," it remains unclear what form or level of cooperation between government and user groups is best for sustainability. What works best will depend in large part on the ecological, socioeconomic, and institutional context. Throughout Africa, for example, authority is commonly split because communal land resources are formally state owned and informally community owned. The result is that authority and management are compromised and open-access tendencies thrive (Metcalfe and Vudzijena 1996).

All stakeholder groups may not be comfortable with a cozy cooperative relationship between government management agencies and the principal user groups. Whether in forestry, fisheries operations, or recreational hunting, close involvement by those user groups that directly benefit monetarily is seen by many as part of the problem rather than the solution. Allowing commercial interests, such as harvesters and processors, to occupy a central seat at the management table is seen as akin to allowing the "fox in the henhouse." Jentoft and McCay (1995, p. 235) note, for example, that environmental groups in the United States are critical of the regional management council system "for being 'captured' by commercial fisheries interests." Similarly, in Iceland, the influence of commercial user groups has led to a trend to reestablish parliamentary authority in fisheries management (Pálsson 1991, cited in Jentoft and McCay 1995). Because of these concerns, groups with environmental and animal rights interests are securing greater roles in determining fisheries management policy. They currently have representation on fisheries councils in Norway and the United States (Jentoft and McCay 1995).

The perception that strong centralized government control is necessary, particularly in fisheries, may in large part be a result of the current fisheries management paradigm that focuses on quotas. J. A. Wilson and co-workers (1994), citing the general failure of the highly centralized form of fisheries management common in the industrialized nations, contend that the complexity and probable chaotic nature of marine systems and of the biology of fish stocks pose major problems in terms of managing stocks through quota systems. Information needs are considered overwhelming if management operates at the scale of a fishery stock's range. Wilson and co-workers examined fisheries regulations in thirty-two societies, most but not all in the nonindustrialized world, all of which represented some form of communal, community-based, or traditional management scheme. Their conclusion was that these communities tended to manage their fisheries more sustainably than did communities employing the government-controlled, top-down approach so prevalent in the industrialized world. They attribute this to two major factors: (1) Rules and practices in these societies regulate *how* fishing is done rather than the *amount* of each species that can be caught. Only one case was found in the nonindustrialized world in which people used quotas as a management tool, although in several cases fishers were expected to catch only what they could consume. (2) The communities or local institutions and the rules and regulations they promulgated covered very limited areas; management was highly decentralized. In many cases, there was essentially no government influence over the fisheries. Because of the complex and unpredictable nature of fish populations, government programs that attempt to manage stocks over large areas by limiting fishing efforts or by imposing quotas are likely to fail. Thus, Wilson and co-workers propose an alternative approach that rests heavily on decentralized management and community-based governance and on regulations that affect how, rather than how many, fish are caught.

Government has a crucial role to play in CCU programs in protecting both the monetary interests of resource owners and harvesters and the broader biodiversity interests of society. But as the foregoing discussion indicates, the difficulty lies in designing the right level and form of government involvement and, once defined, in implementing it. The costs of effective government oversight and of government-run management programs can be considerable. Such investments are often well beyond the means of government agencies, particularly in developing countries. Further, government policies and programs are inherently biased toward quick fixes that are designed and evaluated according to short-term socioeconomic indicators. Although politics and the marketplace can be equally myopic regarding the time scales on which populations and ecosystems should be managed, the spotlight currently seems to be on government failures in resource management. As indicated in chapter 3, the result is an expanding interest in the potential of free market systems as a foundation for sustainable CCU. The primary avenue for achieving this is privatization of resources.

Recent Trends in Privatization of Wild Species Use

The past three to four centuries were an era of rapid expansion of control by central governments and large government-sanctioned corporations over the use and management of wild species resources and natural ecosystems. In much of the world, this was part of a larger trend of colonization by foreign powers and immigrants whereby indigenous peoples and local communities lost historical rights to their resources.

This was most obviously the case with European colonization and colonial rule in the Americas, Africa, and much of Asia (Durby 1987; Wenzel 1991).

Government failures in resource management, combined with a broader societal trend toward confidence in free markets (concomitant with the collapse of centrally planned economies, as in the former Soviet republics), have stimulated interest in the conservation benefits of privatizing wild species resources. In a few places, greater rights are being invested in individual resource owners and corporations. Particularly in developing countries, however, the principal focus is on improving resource tenure rights for, if not entirely returning ownership to, local communities and communal management regimes (IIED 1994; Larson and Bromley 1990; Poffenberger and McGean 1993; Western and Wright 1994).

A new challenge, however, faces these communities: they are left with the despoliation of previous management regimes. Resource rights are being given back to local communities only after populations have been depleted and ecosystems degraded. The result is that communities must cope with the dual challenge of nursing populations and ecosystems back to health and enduring reduced harvest levels (and thus reduced income) during the recovery period. Most communities do not have the capital reserves to do this, and thus outside sources of support as a bridge to sustainability must accompany such changes in tenure (Tietenberg 1996).

A second challenge, which may be transformed into an opportunity, also faces communities that have reestablished their rights to valuable resources: instead of the subsistence and local markets community-based management regimes once served, in many cases the markets are now national and international. Although this may promise much larger monetary returns, it also brings these communities face-to-face with the competitiveness and complexity of world markets.

Increased privatization of wild species resources must be viewed as a global experiment to see whether, and under what conditions, private commercial interests and markets lead to better management than did government-controlled systems.

Forest Resources

The privatization trend in forestry began in the late 1970s, particularly in developing countries, with the so-called social forestry movement. In

India, as noted previously, local communities and tribal peoples recently regained partial control of now highly degraded forest resources after a period of British control of forests during the colonial period and subsequent nationalization of the forests following independence. Forestlands, however, are still under government ownership and considerable government control (Singh et al. 1997). Forest management has a similar history in Nepal, although there it required only thirty-five years of government mismanagement to trigger a devolution of rights back to local communities. In 1957, the Nepalese government enacted the Private Forest Nationalization Act in an attempt to improve forest management. D. A. Messerschmidt (1986, p. 458) notes, however, that "Partly because of this law, preexisting and traditional practices of communal resource management in the form of group control over local forests was upset, and existing local political structures in which communal control was embedded, with their customary rights and duties, became irrelevant." During 1964–1985, Nepal lost approximately 570,000 hectares of forestland. Although the exact causes of the loss are disputed, this dismal record resulted in the Forest Act of 1992, which returns management, but not ownership, of most forests to local user groups (Pardo 1993).

Reprivatization toward community management of national forestlands has followed a more grassroots path in Thailand, where rapid deforestation has occurred because of a "virtual management vacuum" by government (Poffenberger and McGean 1993, p. 1). Absent any official policy, but with increasing tacit approval from the Royal Forest Department, local communities are reestablishing control, with improved management, over government-owned forests. A 1992 survey documented more than 12,000 traditional rural community groups managing forest patches ranging in size from 1 to 4,000 hectares.

In Latin America, examples of the social forestry movement include the Plan Piloto Forestal in the state of Quintana Roo, Mexico, discussed earlier in this chapter, in which forests formerly managed by the government under concession agreements with a timber company have been converted to communal ownership and management (Kiernan and Freese 1997). In the department of Loreto in Amazonian Peru, the 160,000-hectare Tamshiyacu-Tahuayo Communal Reserve was created in 1991 for exclusive use by local communities (Bodmer et al. 1997). A major part of privatization in Brazil involves extractive reserves and settlements whereby government has granted to traditional extractive communities exclusive use rights to, but not ownership of, forestlands, with more than 3 million hectares designated or in the process of being designated (Environmental Law Institute 1995; Pinzón Rueda 1995). On a larger scale, beginning in the 1960s, indigenous groups in South America have achieved some success in reclaiming rights to their native lands. Nearly 110 million hectares in the eight member countries of the Amazon Cooperation Treaty have been set aside for indigenous populations, although substantial reforms are needed in most cases to give indigenous peoples greater security and authority over management of these lands (Davis and Wali 1994). Other Latin American countries, such as Honduras, Panama, and Argentina, have also taken significant measures to recognize native claims (Davis and Wali 1994; *New York Times* 1997). Apart from returning forestlands and use rights to local communities, Honduras, Peru, Colombia, and Brazil have also recently privatized many forest-based industries, such as pulp and paper operations. Based on forestry sector privatization and resulting rapid development of forest plantations in Chile, several other Latin American governments are using privatization and government incentive programs to encourage plantation forestry (Stedman-Edwards et al. 1997).

Among industrialized countries, the governments of the United Kingdom and New Zealand have divested ownership of significant forest areas to the corporate sector. New Zealand stands out in terms of corporate privatization; in a major shift in government policy in the 1980s, most of the government-owned forests managed for timber production were sold to large national and foreign companies. This included major purchases by foreign transnational corporations, and foreign control of New Zealand's forest industry has continued to increase since then. As discussed later, paralleling this trend, with significant implications for forest management, is the growing control by transnational corporations of timber markets, including on-the-ground harvest and management (Dudley, Jeanrenaud, and Sullivan 1995; F. Sullivan, pers. comm., 1996).

On a global scale, these and other efforts at forest privatization represent a very small share of all forest areas. With a few exceptions in developed countries, privatization has stopped short of outright granting of ownership. In most cases, especially in developing countries, the government has kept a strong hand in forest management and, occasionally, in maintaining a significant share of timber revenues. However, governments have seldom concerned themselves with meaningful control of, or revenues from, nontimber forest products. In cases such as India's participatory forest management movement and Brazil's extractive reserves, official actions have simply recognized the use by local communities of nontimber resources to which governments had previously paid little attention.

Wildlife Resources

Globally, one of the most significant trends in privatization of wildlife resources is found in southern Africa. Here, again, the history is one of traditional communal rights to wildlife being lost during colonial periods, when state and individual ownership regimes were emphasized. As M. Matemba (1996, p. 8) notes, "The concept that created the greatest disentitlement and disenfranchisement was that natural resources, particularly wildlife, were deemed God's domain and were to be administered by the state." There are two phases, corresponding to two ownership regimes, of wildlife privatization in the region. In 1967, Namibia was the first country to give individual farmers de facto ownership of wildlife, followed by Zimbabwe and South Africa (Cumming 1991a). Commercialization of wildlife on farmlands has prospered since then, particularly in Zimbabwe and South Africa (see chapters 6 and 7). The second phase began in the 1980s with the movement to return some level of wildlife proprietorship and responsibility to traditional communities. Botswana, Namibia, Zambia, and Zimbabwe have taken the greatest steps in this direction, though in no case have local communities been given full ownership of wildlife. According to A. S. Steiner and E. Rihoy (1995, p. 16), "In all four countries, resource managers and legislators have been struggling with the dilemma of how to give communities a 'sense of ownership' over a resource that is fugitive . . . and must be managed under a communal land tenure regime." The result, with mixed though promising results to date, is a system of comanagement by government and local communities. Substantial uncertainty and controversy remain regarding how and to what degree rights to wildlife resources should be further devolved to local communities.

Privatization of wildlife resources as a conservation tool has also become an issue in Canada and the United States (Rasker and Freese 1995; Rasker, Martin, and Johnson 1992). Wildlife is part of the public domain in both countries, which in their early history meant game was free for the taking, even on private lands (Gilbert 1993). During the nineteenth century, such public ownership, combined with large markets for game meat, hides, furs, and feathers, led to major declines in many wildlife populations (Geist 1993). Although many game populations have since recovered (Geist 1994; Kimball and Johnson 1978), there are concerns that trends in private land use will again lead to loss of game and biodiversity. The solution, according to some, is to enhance the ability of private landowners to profit from commercial wildlife markets (Benson 1992; Teer 1993). In the United States, although wildlife belongs to the state, landowners are increasingly exercising de facto ownership of wildlife by charging trespass fees to those who hunt on their lands (Gilbert 1993). This trend has occasionally been reinforced by government policy and legislation. In the states of California and Colorado, for example, landowners who follow state guidelines for managing land for wildlife are granted extended hunting seasons and increased quotas for game species, which enable them to increase revenues from hunting fees (Rasker and Freese 1995). In Canada, the provincial government of Alberta recently legalized the sale of wild venison (Hawley 1993). Some observers are concerned that such trends will undermine wildlife conservation and public support for wildlife by making hunting opportunities accessible only to the wealthy (Bunnell 1993; Geist 1993). F. F. Gilbert (1993, p. 30) cautions that "The 1980s have seen the strengthening of trespass law, the increased formation of private hunting groups, and the commercialization of hunting opportunities and of game itself. Restrictions on hunting opportunity, including rapidly increasing license fees, have begun to make hunting an avocation of the rich. The landowner again controls whether hunting will occur and who will hunt."

Fisheries Resources

The history of government versus private control of coastal fisheries offers a different perspective regarding the perception and recent evolution of use rights. Unlike the situation with land, property rights are generally not an institution people associate with the sea. However, sophisticated marine tenure systems for nearshore fisheries are found around the world (Acheson 1972; Andersen and Stiles 1973; Cordell 1974, 1989a, b; Forman 1967; Johannes 1978). As J. Cordell (1989a, p. 5) observes, "There aren't many places where an outsider can just walk into town and start fishing—hauling nets, setting traps and so on." Apart from ignorance by the general public and governments, most marine tenure systems are unwritten and thus have no official recognition—indeed, fishers often purposefully resist such official status.

The result of such obscurity regarding traditional marine tenure systems is that during colonial periods many governments superimposed common-property or public-domain fishery statutes on nearshore fishing claims held by local communities (Cordell 1989a; Johannes 1978). The customary tenure systems of many local fishing communities now fall under state-developed common-property laws that allow entrance, and often displacement, by outsiders, which in turn often leads to resource depletion (Cordell 1989a). In brief, customary tenure systems have been replaced by centralized bureaucratic policy making (Symes 1996). The development in 1982 of the 200-mile exclusive economic zone (EEZ) under the Law of the Sea Convention officially made extensive areas of coastal waters the property of the states. Although this helped protect the rights of local fishers by excluding foreign fishers, it also represented another step toward state control of coastal fisheries. According to A. Davis and C. Bailey (1996, p. 257), in many settings "governments have shown a marked resistance to empowering coastal peoples with management control over access to and use of coastal resources. Indeed, if anything, the state has risen to prominence in the pursuit and expression of its proprietorial claims and management prerogatives, particularly over the last couple of decades and culminating in the United Nations Law of the Sea conventions." They note that much of this is due to recognition by state and local elite of the economic gains to be made by expanding proprietorship, and states give legitimacy to this by claiming to protect public interests. The result is that "the coupling of state managerial proprietorship with industrial capitalist harvesting has been a definitive feature of global fisheries development over the past couple of decades" and that "it would be quite surprising to find governments initiating policies that legally recognize small-boat fisher use rights, let alone empowering them with management authority and responsibility."

Despite the existence of traditional marine tenure systems, many marine fisheries present a vexing problem regarding assignment of resource rights because of the migratory nature of many species and because territorial rights are harder to delineate, particularly farther from shore. Moreover, high prices due to strong national and international markets, as well as new fisheries technologies, can cause a breakdown in traditional systems and the sustainability of their fisheries (Weber 1994). In many cases, the intensive fisheries management required to meet the demands of the global marketplace is at odds with local fisheries customs. For example, K. Ruddle, E. Hviding, and R. E.

Johannes (1992, p. 255) report that traditional fishers of Marovo Lagoon in the Solomon Islands are resistant to increased commercialization of their fisheries, particularly to large-scale fishing operations by outsiders, because of their perception of a "limited resource base, and because few people are interested in long-term full-time fishing since their traditional economic strategies are based on a high degree of diversity and flexibility, alternating between subsistence and commercial production and exploiting different resource types." In a review of privatization of fishing rights for small-boat fishers on the Atlantic coast of Canada, A. Davis and C. Bailey (1996, p. 253) observe that "The logic and dynamics of such management policies give little recognition to the fact that for many in small-boat community settings fishing and ocean resources are as much expressions of social relations between kin and familiars as they are about economics and property."

Reestablishment of some form of marine tenure system has recently received increasing recognition from governments, with policies that amount to government-sponsored privatization to let market forces dictate fisheries management (Cordell 1989a, Mace 1993; McCay and Jentoft 1996). This usually entails a limited entry system whereby fishing rights are parceled out under a license system to individuals or vessels. Markets in private permits may then develop. One of the newest forms of limited entry is "individual transferable quotas" (ITQs), whereby a predetermined number of fishers individually receive a share of the annual quota, which they may then harvest as fish populations and markets dictate (Sissenwine and Rosenberg 1993). ITQs may be purchased, sold, or leased like property. Difficulties include decisions about who should receive the original allocation of ITQs and concerns that, without some restrictions on the number of ITQs per individual or company, highly capitalized fishing operations may buy out small holders and eventually control the fisheries (Weber 1994).

New Zealand has applied ITQ management on a more comprehensive scale than has any other nation, with mixed results to date (Sissenwine and Mace 1992). J. Cordell (1989a) warns that limited entry as imposed by ITQs ignores the customs and procedures that small-boat fishers use to establish sea rights. In a review of the effects of a new ITQ system on the sea clam fisheries of the eastern United States, B. J. McCay (1995, p. 107) concludes that "Ownership of vessels and of rights to fish is being concentrated in fewer larger firms, including banks and other 'outsiders,'" with many people losing their jobs in the process. The result is that the "big are getting bigger and the small smaller." The same trend toward concentration of ITQs in extremely large firms is occurring in Icelandic fisheries (Pálsson and Helgason 1996). The effects of these trends on the sustainability of the fisheries is yet to be determined.

Individual, Communal, and Corporate Ownership: Implications for Sustainability

What are the relative merits and liabilities of the three primary types of private ownership or use rights—individual, communal, and corporate—with regard to the ecological and socioeconomic sustainability of CCU? For the purposes of this discussion, individual ownership or use rights exist when a person and that person's immediate family have exclusive ownership of or rights to the use of a wild species. Communal (or common-property) ownership exists when a group of individuals, generally members of the same community, jointly own a resource or share rights to its use. Corporate ownership refers to ownership by a for-profit corporation, whether privately or publicly owned, and, as used here, generally implies a much larger financial operation than found in individual or communal ownership. There is obviously a continuum between these three categories, with great diversity of composition and structure within each. Further, a mix of individual and common-property ownership rights is often attached to different resources at different times of the year within the same area. This is particularly so in traditional community-based management systems (Lynch and Alcorn 1994). Finally, no property right is absolute, as the state always retains some use rights or regulatory control over private lands. Nevertheless, most private resource ownership and use rights can be readily assigned to one of these three categories.

Examples of both good and bad resource management can easily be found for all three forms of resource ownership (the term, as used here, also includes resource-use rights). Each form displays unique characteristics in terms of CCU management, with distinctive implications for sustainability. The main dichotomy is between communal ownership and individual or corporate ownership.

Communal versus Individual or Corporate Ownership

As noted previously, the traditional communal ownership and management systems first encountered by colonial powers were generally ignored and replaced with state and individual ownership. This occurred throughout much of the Americas, Africa, and southern Asia and affected both terrestrial and marine resources (Cordell 1989a; Lynch and Alcorn 1994). More recently, the tendency to lump or confuse communal management systems with open-access conditions, as done with Garrett Hardin's (1968) "tragedy of the commons" (Larson and Bromley 1990), reinforced the perception that communal systems were inferior for sustainability. The conditions under which communal management systems, as compared with individual ownership, create incentives for good resource stewardship remain an area of considerable debate (Berkes 1989; Bromley 1991; Hodson, Englander, and O'Keefe 1995; Larson and Bromley 1990; Mendelsohn and Balick 1995).

B. A. Larson and D. W. Bromley (1990) contend that individual private ownership has often been offered as the solution to resource degradation because it has been widely viewed as the only form of ownership that meets two axioms proposed as necessary for sustainable resource use: (1) the composition axiom states that complete control of a resource must be vested in a well-defined group for efficient use; (2) the authority axiom states that the well-defined group must act with a unified purpose. They note that the power of these two axioms "is legendary and well accepted; individual owners of natural resources will *not* use those resources in an inefficient—or antisocial—manner, while groups will *always* use resources inefficiently and at a rate that exceeds their natural regenerative capacity" (Larson and Bromley 1990, p. 236). This conclusion has been a major reason for efforts to establish individual (as opposed to communal) tenure rights and land registration in Africa since the colonial period as well as a justification for continued state control over communal lands. The composition and authority axioms, however, can be met under a range of property regimes, including communal ownership. More significantly, as suggested by the perverse effects of the discount rate discussed in chapter 3, these two axioms are insufficient to ensure that resource degradation is not an optimal response by the resource owner, whether individual or communal.

Despite these arguments, there is evidence that in some circumstances individual ownership can be more effective than communal ownership in providing socioeconomic incentives for natural resource conservation. S. Metcalfe and V. Vudzijena (1996), for example, suggest that in southern Africa communal tenure provides limited tenure security because it is based on community membership rather than individual title. This insecurity can discourage sound resource management as individuals externalize to the community the cost of conservation. I. Bond (1993, p. 39), in comparing wildlife management on ranches with that on communal lands in Zimbabwe, concludes that "Although considerable gains have been made in returning the proprietorship of natural resources to local communities, the financial and economic benefits have not reached a point where communities are prepared to make the trade-offs between agropastoralism and wildlife production systems. The rationale behind this is clear. The benefits of agropastoralism accrue directly to the individual, while the benefits from wildlife utilization are essentially communal." A key factor contributing to this difference is that government, in the form of district councils, keeps a portion of the revenues from wildlife use but not from agropastoralism.

Another factor that may undermine the sustainability of current communal systems is that communal ownership is most common among the poor in developing countries where customary tenure rules are still in effect, whereas individual or corporate ownership is more common under conditions of economic wealth (Larson and Bromley 1990). The consequence is that communal ownership often goes unrecognized by governments because those seeking recognition are financially poor and politically marginalized. O. J. Lynch and J. B. Alcorn (1994, pp. 378–379) state that "Without official recognition, communities do not have access to the formal legal structure to exclude those who encroach on their rights and overexploit their resources, be they local elites, multinational corporations, or landless migrants."

This link between poverty and communal systems may also make communal systems fragile in the face of strong commercial markets, either because the lure of large profits causes a breakdown in social cohesion and management rules within the community or because of pressures by clandestine harvesters from outside the system. The latter is, in part, a consequence of governments not recognizing and enforcing traditional communal systems. This breakdown, and the resulting depletion of fish stocks, has been well documented in coastal marine systems (Hinton 1987; Ruddle, Hviding, and Johannes 1992; Weber 1994). I. Scoones, C. Toulmin, and C. Lane (1993) suggest that customary uses of savannas based on common-property management have been sustainable in the past, but that resource commercialization tends to undermine the structures that maintain community interdependence and reduces incentives for community-level management.

Despite these problems, the primacy of individual ownership as the best private ownership strategy for resource conservation has begun to crumble as evidence has accumulated regarding successes in communal ownership and management. Communal management systems have been shown to be effective for the ecological sustainability of CCU in management of tropical and temperate forests (Kiernan and Freese 1997; Simeone, Krones, and Nesper 1992; Singh et al. 1997) and coastal fisheries (Acheson 1987; Ruddle, Hviding, and Johannes 1992), and recent work on communal management of wildlife in southern Africa shows similar promise (CAMPFIRE n.d.; Steiner and Rihoy 1995). I. Scoones, C. Toulmin, and C. Lane (1993) suggest that communal property regimes are more efficient than any other form of tenure in ecosystems where productivity is low and the costs of establishing and enforcing property rights are high. Semiarid savanna ecosystems may be particularly suitable for communal management if, as indicated earlier, commercialization does not cause a breakdown of traditional systems. Both J. H. Holmes (1993) and C. Perrings and B. W. Walker (1995) conclude from their review of savanna ecosystems that common-property regimes are less likely to cause land degradation from overgrazing than are private-property regimes (*private* meaning "individual" in this case).

Communal ownership allows people to pool financial and natural resources as well as knowledge, which may be particularly advantageous under conditions of poverty. Whereas individuals may be financially incapable of making investments for the harvest and management of resources, together they can do so (Runge 1986). The communal pooling and sharing of revenues from CCU help buffer against the market and ecological vagaries individuals would otherwise encounter. Larger areas will reduce variations in numbers of harvestable individuals caused by patchiness in the distribution of populations, population fluctuations, and migration of populations. This should help avoid crisis conditions that may lead to overharvesting. E. B. Barbier, J. C. Burgess, and C. Folke (1995, p. 163), for example, suggest that common-property systems "may be well suited to the ecological characteristics of the rangeland, enabling flexible herd size and grazing patterns," and to the economic needs of local rangeland societies, "providing a risk-pooling system for risk-averse individuals in a highly unpredictable and fragile environment." J. M. Acheson (1987, p. 61) suggests that lobster fishers form territories held by sizable groups, rather than individual territories, because "concentrations of lobsters are found in different places over the annual cycle and from year to year. . . . A lobster-fishing territory, if it is to be viable, must contain several different microecological niches." Lobster fishers thus need access to large areas, and one is able to defend such a large area only as a member of a large group that shares in its defense.

The larger areas afforded by collective ownership should also be more amenable to ecosystem-level approaches to management than are the smaller management units of individual ownership. Viewed another way, larger management areas are better able to internalize the costs of resource use; negative externalities are more readily avoided. (Although, as noted previously, there may be a tendency to externalize costs to the community.) For example, under individual ownership, the owner of a forested hillside will have no incentive to consider the possible contribution of increased runoff from logging to the erosion of a farmer's field downhill. If, however, the two owners fuse their properties under communal ownership, those costs are internalized, with the result that logging on the hillside might be reduced. Thus, although enforcement of property rights and handling of externalities can be addressed when parcels of land are individually owned, communal ownership, through the development of collective management rules, may be an administratively more efficient and less costly way of handling these issues. In fact, as resource use intensifies, the owners of individual but contiguous parcels of land may have an increased desire for mutual regulation of land use to ensure compatible and complementary uses of their parcels. The result is often land-use zoning, which is in effect the creation of common-property rules that apply to individual owners (McKean and Ostrom 1995).

Management difficulties posed by individual ownership can be overcome by other forms of collective management among owners of adjacent land. Management of commercially important wildlife populations that regularly move across individual parcels of land may benefit from such cooperation. For example, in the United States, the Ranching for Wildlife Program in the state of Colorado allows extended hunting seasons and offtake of big game for ranchlands covering more than 2,000 hectares if state-established wildlife management guidelines are followed. Owners of ranches too small to qualify have pooled their lands into management units of greater than 2,000 hectares in order to participate in the program and thereby earn extra revenues from fee hunting (Rasker and Freese 1995). Ranchers in various regions of southern Africa have also pooled their lands to manage for big game. One of the most significant such programs is the Save Conservancy in Zimbabwe, which covers 300,000 hectares and involves seventeen properties (D. Cumming, pers. comm., 1997; du Toit 1992).

Finally, compared with individual and corporate ownership, communal ownership would be expected to exhibit greater site fidelity: individual members of a communal group may leave, but communal societies should be inherently less mobile than are individuals or corporations. Nomadic pastoralism is only a variation on this, since the communities return periodically to the same places. This should render wild species resources less vulnerable to the cut-and-run frontier mentality and to resource mining exacerbated by high discount rates. Assuming that communal groups are less mobile than individuals or corporations and that such groups are often characterized by long-established management systems and cultural links to the resource, one would expect them to take a long-term view of the resource base. O. J. Lynch and J. B. Alcorn (1994, p. 385) note that "Maintaining biodiversity reserves is one strategy that enables communities to maintain their identity and self-reliance; biological resources, as the ultimate safety net for the poor, also serve to secure survival." Thus, where poverty prevails, communal systems may be particularly important in maintaining biodiversity.

Corporate Ownership

Corporate ownership comes in many forms and sizes, but the principal concern in this discussion is how large corporations, including transnational corporations, manage wild species commodities. Compared with individual and communal ownership, corporate ownership surely places greater emphasis on maximization of profits. Unlike communities and individuals, corporations do not depend on diverse products and services from the ecosystem for their survival and therefore will focus more exclusively on marketable commodities from the ecosystem. A wild species commodity may be just one product among many investments, and one product or investment can be substituted for another, depending on what yields the highest return. For example, timber in a forest and the land on which the forest grows can be seen as simply two holdings in a large portfolio of investments. Thus, if an alternative investment, whether in bonds or bonbons, looks better, the corporation's logical economic choice (and its financial obligation to shareholders) may be to clear-cut the forest and reinvest the proceeds elsewhere. The combination of high or even moderate discount rates and the slow growth of most forests would seem to favor such cut-and-switch investment strategies (Dudley, Jeanrenaud, and Sullivan 1995). Such corporate policies may be facilitated by government policies that have been shaped by the political influence often wielded by large corporations. Thus, large corporations can be highly mobile in terms of investments, a feature that does not lend itself to ecological sustainability. C. D. Becker and E. Ostrom (1995, p. 126) caution that "Societies driven to the efficiency level that favors overcapitalization and large corporate exploitation have failed to use diverse biological resources sustainably, largely because resource developers have banked on substitutability for sustainability."

In fact, current trends in the structure of timber companies and globalization of the timber trade do not favor long-term investments in forest management. Timber trade is becoming increasingly concentrated among fewer, predominantly corporate interests. Mergers and acquisitions have resulted in the assimilation of many locally owned timber companies into larger international industrial concerns. Shareholders of publicly owned corporations are generally uninterested in and ignorant of the resource management practices of the corporations they own. About 80–90 percent of trade in forestry products is now controlled by transnational corporations, and such corporations are becoming increasingly vertically integrated so that they control all stages, from cutting in the forest to selling the end product. Consequently, companies tend to see the forestry sector as just one part of their overall operation and investment portfolio and are much more willing and able to move their investments around the world (Dudley, Jeanrenaud, and Sullivan 1995). The problem is most apparent in terms of a corporation's ability to switch its logging operations from one country to the next as supplies are depleted.

Large corporations, however, do offer potential advantages if there is a corporate philosophy or incentive for good resource stewardship. They can often own or acquire use rights to large enough areas so that ecosystem-level management is possible. Corporations can afford to invest in research and management, including the demands of a precautionary approach, that individual or communal owners may find financially impossible. FSC certification of several corporately owned timber operations (Good Wood Alliance 1996) testifies to the corporate potential for sound management. Moreover, compared with a highly fragmented ownership system, when major industries control large sectors of markets for wild species commodities the potential exists, if the leverage can be found, to convert significant portions of commodity markets to more sustainable practices. The recent agreement between WWF and Unilever, the world's largest fish marketer, to develop a sustainable fisheries program illustrates this potential (see chapter 3).

Poverty and Sustainability

The issue of poverty and sustainability is given separate treatment here because it presents a widespread but often insidious and unarticulated problem for wild species use, particularly in areas that harbor some of the earth's greatest biodiversity—tropical forests and tropical coastal communities. Three problems are closely associated with poverty-ridden peoples—the poorest of the poor—in terms of resource management: (1) general lack of rights and resources; (2) political marginalization and exploitation; and (3) socioeconomic marginalization and lack of safety nets.

Poor people in rural areas often do not own land. Many of the poor, particularly indigenous peoples, once owned land or had unfettered rights to its use but could not defend the land against those who would take it—often central governments or economically and politically powerful individuals. Under such conditions, many poor people are forced to be free riders, either legally or illegally, in the use of resources or to depend on employment, often at depressed wages, by those who own the resource. Rural poor often depend directly on the harvest, use, sale, and barter of natural resources for most or all of their subsistence needs. Resource-dependent communities are often economically impoverished and politically marginalized (Bailey and Pomeroy 1996). Indeed, because they depend so directly on natural ecosystems for goods and services, the poor often most heavily bear the costs and suffer the consequences of resource degradation and the negative externalities inflicted by bad management (Lonergan 1993).

Without resource ownership or secure use rights by the poor, long-term socioeconomic security and therefore incentives for sustainable harvest practices are absent. In addition, when resource owners or harvesters are poor and sociopolitically marginalized, they often lack the means to obtain a more judicious distribution of revenues. If they do not receive their fair share of the revenue stream, socioeconomic sustainability is jeopardized and incentives for sustainable resource use are lacking.

Poverty and lack of ownership also preclude many options for developing sound programs for the use of wild species because the funds for investment in land, capital, ecological and marketing research, and other necessities for many such programs are not available. Community-based or communal approaches can help rectify this situation for those who have resources to pool, but many of the poor do not. Even when support for such investments is available, the poor, unlike those who own land and are otherwise resource rich, must strike a bargain (with little leverage), usually with government, for the comanagement of resources and the sharing of revenues. In some respects, people living in indigenous reserves, though often financially poor, at least have a somewhat more even playing field than do the landless poor for negotiating resource-use management and rights with the government.

The dependence of the rural poor on randomly distributed and unpredictable natural resources, including low-scale agricultural output, creates a high degree of uncertainty regarding income streams. They do not have the cushion against periods of scarcity that accumulated wealth provides. In contrast, the technologies of highly developed economies, whether irrigation systems, feedlot livestock operations, or sophisticated food distribution systems, confer a hedge against natural ecological fluctuations. Further, individuals in wealthy societies have financial insurance and credit to carry them through hard times. Ecological fluctuations and uncertainty are an incentive for the poor not only to develop common-property regimes, but also to maintain a diverse biological base of resources (Runge 1986).

The poor are generally marginalized, if not totally disconnected, from the political process that shapes resource-use and tenure policies, particularly in developing countries (Durning 1994; Warford 1987). Worse, those belonging to certain ethnic and indigenous groups may be actively discriminated against. Such peoples are often exploited as pawns in the political process. Therefore, they frequently do not trust the political process and find little security in any gains they achieve in resource use. Further, political and socioeconomic marginalization occurs in many forms at various scales—North-South, haves–have nots, men-women, and other sociocultural biases. Regardless of why people are marginalized, the effect is generally to weaken incentives for good resource stewardship.

Conditions of poverty force people to think and act for the short term—the "have-to-eat-today" principle. Thus, when a resource is in decline, there is no option to forgo this week's or month's or, much less, year's harvest to allow for its recovery. Worse, poverty renders people susceptible to socioeconomic exploitation in ways that serve only to exacerbate overexploitation of natural resources. Although it operates at a different economic scale, the same perverse incentive—the overriding requirement to meet short-term needs—emerges when individuals or corporations overcapitalize, whether in forestry or fisheries operations.

Since many populations of wild species that inhabit ecosystems with the landless poor are already depleted from overuse and the people who depend on them have no alternatives, the conservation and development communities face a special problem in improving the conditions of both. One will probably not happen without the other.

International Accords

The increasingly global economy and the expanding international flow of CCU products have elevated the role of international accords in managing biodiversity. International accords that focus on CCU and biodiversity largely address two issues: (1) management of species that are part of the global commons, particularly open marine fisheries and migratory species, and (2) regulation of international trade in wild species and their products. In addition, a third category of international accords—free trade agreements—is, as noted in chapter 3, having a growing effect on wild species use and trade.

Although controlling open access is the major problem facing fisheries management within each nation's EEZ, the problem is magnified beyond the EEZ. International agreements such as the United Nations Convention on the Law of the Sea (UNCLOS), the Convention on the Conservation of Antarctic Marine Living Resources (CCAMLR), the International Convention for the Regulation of Whaling (ICRW), and the International Convention for the Conservation of Atlantic Tunas (ICCAT) exhibit varied degrees of success and failure. Multiple problems remain with implementation and compliance as most of the world's stocks continue to be depleted (Norse 1993; Safina 1993).

CITES, with 132 member nations, is the single most important international treaty affecting global commerce in wild species. CITES has been effective in regulating international trade of several species and their products, but illegal trade remains significant for many others (Hemley 1996). Ensuring that legal trade remains within sustainable levels remains the cornerstone and the major challenge of CITES. Many exporting countries lack the resources or the political will to undertake sufficient research and monitoring to demonstrate that they can prevent international trade from depleting wild populations of species listed in the CITES appendices. Importing countries often lack the regulatory means by which to restrict trade known to be detrimental. The CITES "significant trade process" (designed to identify and address potentially detrimental trade) and CITES legislative reviews have focused attention on these implementation issues. However, to date, the parties to CITES have been far more successful at identifying problematic trade than they have been at directing resources toward improved management of listed species (S. Broad, pers. comm., 1996). The latter requires progress in harnessing the economic value of traded species as a conservation tool.

To accomplish this, CITES must work toward more constructive trade control mechanisms that provide stronger economic incentives for those who sustainably manage populations. For CITES to improve its effectiveness, it must move beyond a single focus on penalizing unsustainable trade through sanctions and seek mechanisms to reward sustainability. CITES could provide an international structure for certification, tracking, and monitoring of sustainably produced wild species commodities, the cost of which should be borne by the importing countries as part of the cost of sustainability (Swanson 1994). This, however, would entail a major overhaul of CITES. It would have to broaden its scope to include trade in nonthreatened species and much more comprehensive assessments of ecological and genetic effects as criteria for judging sustainability. It would also need to consider, for any threatened species, whether cessation of trade or continuation of trade with restructured incentives would be the best approach to population recovery (Freese 1997b).

The Convention on Biological Diversity (CBD) provides a broad and ambitious global conservation agenda with potentially far-reaching effects on the linkage between CCU and biodiversity. Ratified by more than 170 countries, the CBD has three main objectives: (1) conservation of biodiversity, (2) sustainable use of biodiversity, and (3) fair and equitable sharing of the benefits arising from its use. Responsibility for implementation is placed at the national level; governments are to ensure that natural resources are not used in a manner that would jeopardize a country's own biodiversity or that of its neighbors. The CBD establishes mechanisms for parties to share information on the status of biodiversity and technologies for its conservation and sustainable use (Convention on Biological Diversity 1994). Work has begun on a protocol to cover the production, testing, use, and transfer of genetically modified organisms. The CBD may strengthen the workings of other conventions such as CITES and Ramsar (Convention on Wetlands of International Importance Especially as Waterfowl Habitat); an agreement has been signed between parties to both the CBD and Ramsar on implementation objectives.

Summary

A requisite for sustainability is that those who are ultimately responsible for the management of wild species populations and associated ecosystems must receive a major share of the economic benefits derived from their use. Unsustainable use often results from poorly defined property rights that result in open-access conditions. The legitimate role of government in most programs for use of wild species is to defend the property rights of resource owners and to ensure that broader public interests in biodiversity are accounted for in management.

Governments also have assumed ownership of a large portion of the world's land, water, and wild species resources, but government oversight and resource management have often been inept. Recent trends in privatization of natural resources are an attempt to correct both social injustices and failed government programs for resource management. Much of this trend involves not outright privatization but experiments in comanagement between the private sector and government agencies. What constitutes the best balance for ecological sustainability between local resource rights and government control depends on both social and economic factors as well as the type of use and the ecosystem in question. A central issue among the social factors concerns the differences that individual, communal, and corporate ownership bring to the management of CCU.

A diversity of other stakeholders, ranging from consumers to those concerned with biodiversity conservation and humanitarian issues, influence both resource owners and government policies regarding CCU. Stakeholders concerned with the ecological sustainability of CCU generally exert their influence through government policy instruments. Global markets and the international movements of many wild species populations, particularly marine species, have led to increasing reliance on international agreements to control wild species use and trade.

Poverty presents an acute problem for CCU and ecological sustainability in much of the world. Rural poverty, overexploited wild species populations, and high biodiversity are often sympatric. Poverty and the "have-to-eat-today" principle largely preclude the long-term investments required to restore depleted populations and degraded ecosystems. Enlightened support and investments by the conservation and development communities will often be necessary for any progress toward sustainability.

C H A P T E R 5

Ecological Issues

The adversary at the other side of the board is some complex combination of nature's genuine intractability and our hidebound social and mental habits.

—Stephen Jay Gould (1996)

Our understanding of ecosystems and population biology has changed considerably over the past decade or two, with wide-ranging implications for the ways in which we manage wild species. The most fundamental shift in knowledge concerns the stability, resilience, and predictability of ecosystems (Holling et al. 1995; Pimm 1991). The phrase *balance of nature* had a reassuring, comforting ring to it. It gave us the assurance that there was some balance to be found and understood by science and the comfort that with enough scientific knowledge and management, we could count on nature to predictably and steadily provide for us.

But the "balance of nature" has suffered the same fate as the "bounty of nature." Neither fits very well with what ecological research and our attempts to intensively manage wild species are revealing. Ecological systems and species populations, we now understand, are highly dynamic and change in significant and unpredictable ways with or without human intervention (Pimm 1991). Not only are they a moving target for the manager; at times they seem to present no clear target at all.

A second major shift in our perception of the way ecosystems work regards the role of biodiversity in maintaining ecosystem functions and in conferring stability (resistance to change) and the potential for recovery (resilience) when ecosystems are disturbed (Christensen et al. 1996; Holling et al. 1995). Part of this shift is a greater appreciation of the frequency and importance of disturbances (fire, hurricanes, droughts, pest outbreaks, changes in oceanic current, etc.) in structuring ecosystems.

Surely this "newfound" wisdom regarding the dynamic nature of ecosystems and populations, the limits of their productive capabilities, and the role of diversity and disturbance is nothing new to humankind. It might be better thought of as rediscovered wisdom. Human populations have always been subject to the erratic swings of nature, and the advent of agriculture was simply an adaptive response to both increase and even out what nature could provide. For societies that live close to nature and depend on it to meet their basic needs, living with wild ecosystems must be akin to sleeping with an elephant—you feel and respond to its every move. And although once in a while you may get crushed, if you understand the beast well enough it also offers considerable protection and security. Meanwhile, the sophisticated production and distribution systems for a handful of species that feed, clothe, and shelter industrialized societies buffer people from the vicissitudes of nature. The greater the demand for wild species products in the industrialized world, the more that society and markets expect the flow of those products to be steady, predictable, and bountiful. That is the way the industrialized world has attempted to manage CCU, and it is within that context that Western science's approach to natural resource management evolved. As M. Gadgil and F. Berkes (1991, p. 138) contend, scientific resource management "developed in the service of the utilitarian, exploitive, 'dominion over nature' world view of colonialists and developers. It is best geared to the efficient utilization of resources as if they were boundless. This is the legacy of the *laissez-faire* doctrine of Adam Smith."

The issues and trends examined in this chapter illustrate both the legacy to which Gadgil and Berkes refer and recent scientific efforts to move toward a more holistic, integrated approach to managing wild species commodities.

Uncertainty and Variability

Natural ecosystems and the species populations that inhabit them often are not very stable over either the short term (measured in years or decades) or the long term (measured in centuries or millennia). Further, we are not very good at predicting when, why, how, and at what rate changes in populations and ecosystems will occur. One year's, or decade's, abundant and harvestable population may, without warning, drop to low and unharvestable levels a year or decade later (Hilborn and Ludwig 1996; Pimm 1991). Moreover, what we perceive to be a static "natural" diversity and ecosystem structure may instead be one stage in a long transition or cycling from one ecosystem configuration to another (Botkin 1990; Sprugel 1991). This fundamental feature of ecosystems challenges the concept of sustainable offtake and the demands of commercial markets for a steady and predictable supply of wild species commodities.

A diversity of economically important species in a wide range of ecosystems display boom-and-bust cycles. Short-lived species with high reproductive rates, such as annual plants and many small animal species, and the products of mass reproduction, in particular seeds and fruits, show the greatest year-to-year variation. Significant variation in harvestable stocks of long-lived species with low reproductive rates, such as trees, elephants, and whales, may be noticeable only over decades or centuries. Size, however, may be deceptive. For example, ginseng and wild leeks (*Allium tricoccum*), both small, perennial herbaceous species of great commercial value, are long-lived and may show little year-to-year variation (Nantel, Gagnon, and Nault 1996)

Boreal, temperate, and tropical trees display considerable variation in the production of commercially important products, from fruits to timber. The seed crops of tropical forest trees, for example, are highly sporadic, with species of the primary forest canopy often pausing several years between crops. Intraspecific variation can be significant, and even among individual trees it is common for the seed crop to vary one-hundred-fold over years (Janzen and Vásquez-Yanes 1991). Although detailed information on production patterns of most nontimber forest products is lacking, L. H. Pendelton (1992, p. 256) suggests that "Left to the vagaries of nature, the variance of nontimber product yields is likely to be high." Brazil-nut production from individual trees, for example, is reported to vary greatly from year to year (Rosengarten 1984, cited in Clay 1997a), and illipe nuts (*Shorea* spp.) are produced sporadically every few years during intense masting periods (Salafsky, Dugelby, and Terborgh 1993).

Natural stands of trees in commercially important forests often exhibit long-term fluctuations in harvestable quantities of timber because of large-scale forest dynamics. For example, the most important commercial species of timber in Neotropical forests, mahogany, regenerates after large-scale episodic disturbances such as fires, hurricanes, and hydrologic changes. Because such disturbances occur infrequently, mahogany stands are composed of even-aged trees that have regenerated at the same time (Boot and Gullison 1995). Mahogany generally requires at least 75 years, and perhaps as long as 125 years, to reach harvestable size (Snook 1991). Thus, within any one stand, harvestable volumes of mahogany may be produced only once every few decades, with virtually no production in between. To date, plantation forestry is largely impractical for mahogany in the Neotropics because it leads to massive infestations of the shoot borer (*Hysipila grandela*), which causes high mortality and deformed growth in mahogany (Rodan 1992).

The balsam fir (*Abies balsamea*) and spruce (*Picea glauca, P. pubens*) forests of northeastern North America offer a similar picture of extreme long-term variation. In these forests, the spruce budworm (*Choristoneura fumiferana*), rather than an abiotic factor, is the primary agent driving natural forest development (Baskerville 1995). Like mahogany forests, these forests never develop into what would conventionally be called a "normal" forest with an uneven age structure because the periodicity of extensive destruction by the budworm results in the presence of only one or two large age classes at any one time (figure 5-1). Thus, G. L. Baskerville notes (1995, pp. 59–60), "The natural dynamics of the fir/spruce forest were shown to be biologically inconsistent with the even-flow harvest paradigm of forest management."

Savannas exhibit some of the greatest annual variation in productivity among all ecosystems because of the extreme periodicity of rainfall. Savannas, particularly dry savannas, are nonequilibrium systems that may oscillate either regularly or chaotically (Ellis 1992; Ellis and Swift 1988; Solbrig 1993). According to M. Westoby, B. H. Walker, and I. Noy-Meir (1989, cited in Solbrig 1993, p. 31), "Savanna dynamics can be described by a set of alternative stages of the vegetation and

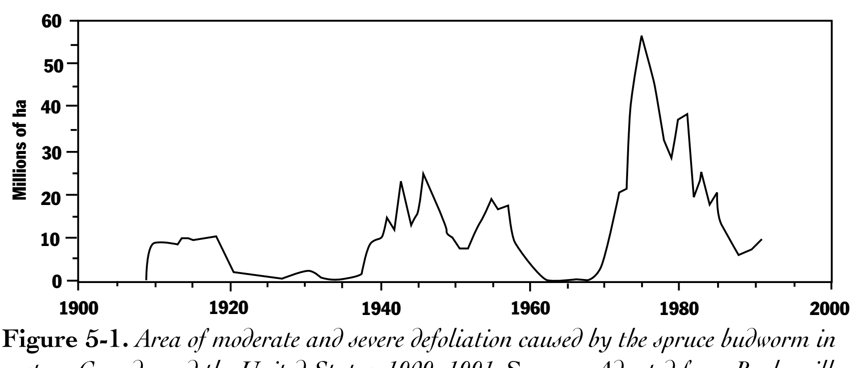

Figure 5-1. *Area of moderate and severe defoliation caused by the spruce budworm in eastern Canada and the United States, 1909–1991.* Source: *Adapted from Baskerville 1995.*

a set of discrete transitions between states," generally called the "state and transition" model. Episodic changes in vegetation, including species composition, often occur in response to extreme or rare events such as drought or alteration in fire regime. Thus, savannas are highly variable over time in terms of both productivity and product. Nomadic pastoralism, such as that which historically characterized African savannas, is adapted to the alternation of such excess and shortage of grass and water. Constant stocking rates, as often attempted by commercial enterprises, can lead to overgrazing and degradation of the range and may facilitate transitions to other states (Ellis 1992; Walker 1993).

Migratory waterfowl populations exhibit sizable population fluctuations that complicate the setting of annual hunting quotas in North America (Nichols, Johnson, and Williams 1995). Variability in the spring condition and extent of wetlands in waterfowl breeding grounds is the primary factor affecting year-to-year reproductive success. For example, populations of the ten principal duck species were high during the mid-1950s and then declined in the early 1980s because of drought conditions in the prairie-parkland region of North America. Populations of individual species, such as the mallard (*Anas platyrhynchos*), pintail (*Anas acuta*), redhead (*Aythya americana*), and canvasback (*Aythya valisineria*), all of great importance in waterfowl hunting, displayed roughly twofold decadal population shifts during this period, as well as significant year-to-year changes (U.S. Department of the Interior 1988).

The six- to ten-year cycle of lynx (*Lynx lynx*) and snowshoe hare (*Lepus americanus*) populations and other carnivores associated with this fluctuating system are well known (Keith 1963). Other populations of large mammals also exhibit sizable but less predictable population changes. T. P. Young (1994), for example, found large die-offs reported in the literature for seventy species of large mammals, many of importance for CCU and representing a diversity of ecosystems and mammalian orders. A disproportionately high number of cases involved die-offs of 70–90 percent of the population.

J. F. Caddy and J. A. Gulland (1983) identify four basic population patterns among stocks in marine fisheries: steady, cyclical, irregular, and spasmodic. Boundaries between these four patterns are not discrete; thus, some stocks may fall between two patterns. Steady populations show little fluctuation, with variations generally within 20–30 percent of the long-term average. Examples of such stocks include the Greenland halibut (*Reinhardtius hippoglossoides*) and the Georges Bank haddock (*Melanogrammus aeglefinus*) prior to 1965 (Caddy and Gulland 1983). Cyclical stocks show periods of high and low catches repeated at regular intervals, similar to the cycles of northern furbearers. Examples include the Bay of Fundy scallop (*Placopecten magellanicus*) and the saffron cod (*Eleginus gracilis*) in the Sea of Japan. Irregular stocks show wide fluctuations in numbers, often from year to year, without any clear pattern. Examples include the Norwegian juvenile herring (*Clupea harengus*) and the Georges Bank scallop (Caddy and Gulland 1983). Pacific salmon stocks illustrate a combination of cyclical and irregular fluctuations, as exemplified by the sockeye salmon (*Oncorhynchus nerka*). The sockeye displays both regular five-year population cycles caused by the return of different stocks each year, with one strong stock reappearing every five years, and longer-term, largely unpredictable fluctuations caused by climate-induced changes in the ocean ecosystem (figure 5-2) (Francis 1997).

Spasmodic stocks are characterized by periods of abundance alternating with collapse or rarity of the resource, the latter often lasting for years or decades. During periods of abundance, these stocks represent some of the largest individual wild species resources in the world. Examples include the Japanese sardine (*Sardinops melanosticta*), the Pacific sardine (*Sardinops sagax*), and the Peruvian anchoveta (*Engraulis ringens*). Rates of accumulation of anchoveta scales off the Peruvian coast reveal the dramatic fluctuations in populations over the past 18,000 years (figure 5-3). Recent fisheries statistics and paleoecological studies show that populations of the anchoveta, Japanese sar-

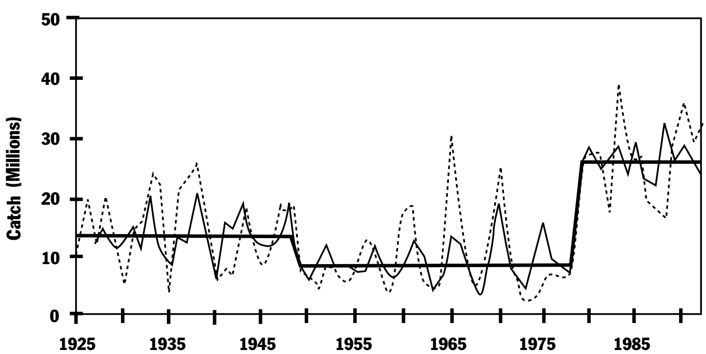

Figure 5-2. *Population levels of western Alaska sockeye salmon, 1925–1992, showing periodic five-year fluctuations (dashed line indicates catch; and thin solid line indicates an intervention model fit) and decadal-scale fluctuations (thick solid line).* Source: *Francis and Hare 1994.*

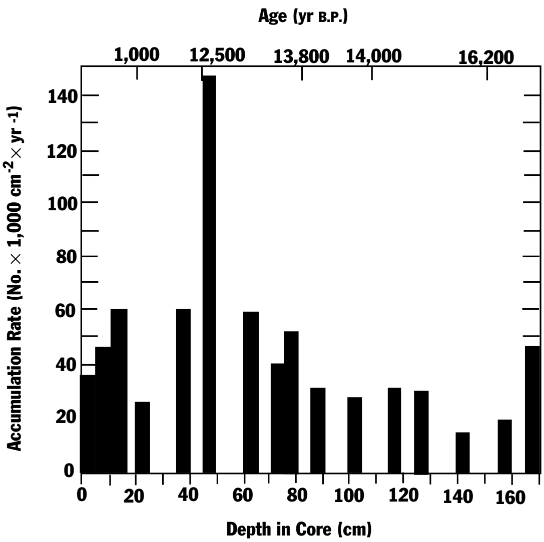

Figure 5-3. *Accumulation rates of anchoveta scales off the Peruvian coast as an example of extreme long-term population fluctuations in marine fish.* Source: *Adapted from Cury 1993.*

dine, and Pacific sardine, all species with short life cycles (life spans of four to eight years), may collapse to virtual extinction for decades or centuries without any apparent regularity (Cury 1993).

Although natural factors in the oceanic environment account for most stock fluctuations, fluctuations may be amplified by increased fishing effort. As the population of a fish stock rises, investments to expand fishing effort can respond quickly, but feedback of information is slow regarding the effect of the expanded effort on the stock. The result is continued overfishing of a declining stock. This is exacerbated when in response to a call for help by fishers facing declining stocks and income, governments provide subsidies to fishers, which only further delay any tendency for the fisher to leave the fishery or reduce investments. This can lead to cyclical patterns even in otherwise stable resources (Caddy and Gulland 1983).

Marine systems, particularly in high latitudes and in areas of great nutrient upwelling, may even exceed savannas and grasslands in their propensity to undergo major shifts that affect both the size and composition of important CCU species. For example, R. C. Francis and S. R. Hare (1994, p. 279) conclude from their examination of population patterns of salmon and zooplankton that "climate-driven regime shifts, such as those we have identified in the northeast Pacific, can cause major reorganizations of ecological relationships over vast oceanic realms." Such large-scale and abrupt changes in populations of commercially important fish stocks pose a major challenge for fishers and fisheries managers who want to satisfy the ever-growing demands of commercial markets.

The Role of Biodiversity in Ecosystem Function

The link between biodiversity and ecosystem function is a two-way street for CCU and biodiversity conservation. One way concerns the level of biodiversity required to maintain ecosystem functions and to produce desired yields of the target population. The question for the manager of a CCU product is, How much can the ecosystem be simplified to direct a greater share of photosynthetic production toward the wild species commodity without undermining ecosystem processes and long-term sustainability of the offtake? The other way concerns how different levels of offtake of a species or set of species in an ecosystem affect the biodiversity of that ecosystem. For the biodiversity conservationist, the question is, Under what circumstances and to what extent can sustainable offtake be ecologically sustainable?

Biodiversity, Disturbance, and Productivity

The most important ecosystem function for most forms of CCU is productivity, in particular the productivity per unit of land or water of the commercially important species. Where CCU dominates the management of an ecosystem and the incentives are right for avoiding overexploitation, management will attempt to increase the productivity of the commercial species and avoid loss of that productivity to natural mortality and higher levels of the food chain. Whether directly through management interventions (e.g., physical removal) or indirectly through changes in the the ecosystem (e.g., disruption of the food chain), the result is that populations of nontarget species decline or become extinct and ecosystem structure is generally simplified. The point at which these changes begin to compromise ecosystem function in a way that reduces productivity of the target population is a central management question. Time is an important dimension of this question because short-term productivity may be enhanced at the expense of long-term productivity.

The question facing the manager is somewhat different where the

CCU species is but one of several ecosystem values, such as biodiversity and recreational values, that must be maintained, as is the case on public forestlands in many countries. Here, the job of the natural ecosystem manager may be to find "a compromise between maximizing monetary profits and maximizing biological diversity" (Kuusipalo and Kangas 1994, p. 456). If the profits are derived from CCU species, this may be an impossible task. How does one determine what compromise to strike? As A. J. Hansen (1997, p. 220) states, "A challenge of sustainable forestry is to understand forest ecosystems well enough to maintain biological integrity even while extracting resources."

According to Hansen, the paradigm of maximum sustained-yield forestry, which guided foresters during most of the twentieth century, is outdated in North America for two reasons: (1) society has placed increased value on the various noncommodity values of forests; (2) research suggests that traditional sustained-yield management often did not sustain long-term timber production. He notes that "Intensive plantation forestry removes some elements of forest structure, composition, and function that are essential for long-term primary and secondary forest productivity." The primary advantage of greater species diversity in natural forests over less diversity in tropical plantation forests may be that it provides functional redundancy. Nutrient cycles, production, and turnover of organic matter are thus buffered from the variability and stress of abiotic factors, and the ecosystem is therefore more resilient to disturbance (Lugo 1995; Silver, Brown, and Lugo 1996).

One management technique used to maintain biodiversity in managed forests in the Pacific Northwest of the United States is "green-tree retention," whereby various densities of trees and shrubs are left intact during harvesting. The concept is that this may mimic the substantial levels of natural legacy that often remain after natural disturbances (such as forest fires) in this region. Green-tree retention and longer rotation periods in these forests appear to create levels of structural complexity and native species diversity closer to those of natural stands than does clear-cutting with shorter rotations. Although green-tree retention results in lowered net primary productivity than is found in stands clear-cut within the previous ten years, it remains possible that green-tree retention benefits productivity over the long term (Hansen 1997; Hansen et al. 1995). Despite the fact that the forests of the Pacific Northwest are among the most studied in the world, it remains unclear what components of biodiversity are important for long-term forest productivity.

A larger framework within which to view the biodiversity-productivity linkage is provided by the fact that change in many, if not most, ecosystems is not continuous and gradual but episodic, with rare but major disturbances such as hurricanes, fires, pest infestations, droughts, and human use playing crucial roles (Holling et al. 1995). Further, ecological theory suggests that maximum diversity develops at intermediate levels of disturbance (Roberts and Gilliam 1995). A. E. Lugo (1995), for example, notes that the belief that tropical forests are fragile ecosystems is based largely on the notion that they are ancient ecosystems, untouched by catastrophe. He argues, however, that "Wherever detailed analyses are made in the tropics, vegetation is found to be the product of past disturbance by people or by natural events including fires, hurricanes, windstorms, floods, or biotic outbreaks" (p. 957). More localized disturbances created by tree falls in tropical rain forests are also important in maintaining diversity (Denslow 1995).

The concept that catastrophic disturbances often play a major role in structuring ecosystems and the idea that ecosystems may have two or more functionally different states point out the need to better incorporate concepts of ecosystem resilience into management. C. S. Holling and co-workers (1995, p. 50) argue that the most useful definition of resilience in this case is "the amount of disturbance that can be sustained before a change in system control or structure occurs."

Semiarid grasslands illustrate this issue well. Under natural conditions (and including the presence of nomadic herders) in eastern and southern Africa, grasslands were periodically intensively grazed by large herbivores, with little grazing between such episodes. This grazing regime resulted in a dynamic balance between two groups of grasses, one able to withstand intense grazing pressure and droughts because of deep roots and the other more productive and palatable to grazers and less resistant to drought because more of the plant's biomass occurs above ground as foliage. Between bouts of intensive grazing, the drought-sensitive species have the advantage, but the advantage shifts to the drought-resistant species during the grazing periods. The result is that a balance between two diverse sets of grass species is maintained, one whose function is primarily productivity and the other, drought protection. When subjected to moderate grazing pressure from cattle, however, the productive, drought-sensitive species have a consistent advantage over the drought-resistant species and their soil- and water-holding capacity. Functional diversity of the ecosystem is reduced as the species assemblage narrows to one functional type—high productivity but low drought tolerance. As a consequence, a grass-dominated ecosystem previously resilient to drought is, when subjected to drought, suddenly transformed into a shrub-dominated ecosystem. Moderate grazing had the effect of making the system less resilient. Intensive grazing can lead to the same result by removing the suppressing effect of both grasses and the grassland fires on woody seedlings. In either case, even if cattle are removed, return to the former grass-dominated ecosystem state may not occur because the shrubs suppress grass regeneration and there are no periodic fires to control woody plant growth (Dublin, Sinclair, and McGlade 1990; Holling et al. 1995; Perrings and Walker 1995). In short, the system has flipped to a different steady state.

Such alternative ecosystem states also appear to be maintained by abiotic disturbance in some tropical forest regions. T. C. Whitmore (1991) reports that in a purportedly primeval dipterocarp (*Dipterocarpus* spp. and *Shorea* spp.) lowland rain forest in Sungai Menyala on the Malay Peninsula, shade-tolerant species, rather than the light-demanding, commercially valuable dipterocarps, were regenerating in the undercanopy. Records indicate the occurrence of three or four windstorms in the past century, each of which destroyed a patch of forest several kilometers long and wide. Canopy destruction by such windstorms opens the forest to colonization by dipterocarps, and the frequency and coverage of such storms in this region is sufficient to maintain extensive tracts of dipterocarp forest. Whitmore also provides evidence of cataclysmic destruction of rain forests by forest fires and changes in the courses of meandering rivers.

This disturbance-dominated feature of many ecosystems has led to the development of a model of ecosystem dynamics that is divided into four phases—exploitation, conservation, release, and reorganization (Holling et al. 1995). Phase 1, exploitation, begins soon after a site has been disturbed by a major event such as a fire, a hurricane, an intense pulse of grazing, or a pest infestation, leaving an abundance of nutrients to be exploited. This phase is dominated by opportunistic pioneer species that exhibit the characteristics of r-strategists—relatively short-lived species with rapid growth rates, high reproductive potential, and the ability to enter and colonize a site rapidly. There is little close interdependence among species, since many species are generalists.

During phase 2, conservation, the "capital" of nutrients and biomass in the ecosystem slowly builds up and interdependence and stability increase. K-strategists, species characterized by slow growth rates and long life spans, are more common in this phase. The ecosystem is relatively stable at this stage as long as disturbances are small, but its resilience to major disturbances is low. It is, as Holling and colleagues (1995, p. 65) suggest, "brittle," "an accident waiting to happen." For example, in the spruce and fir forests of eastern North America, periodic outbreaks of the spruce budworm are triggered when the amount of foliage reaches a threshold that supplies abundant food and habitat for the budworm and decreases the foraging efficiency of insect-eating birds (Holling 1988). Management aimed at maintaining an ecosystem indefinitely at this conservation phase only increases the brittleness of the ecosystem, so when an accident does occur it may be even more severe. The extensive fire of 1988 in Yellowstone National Park, after years of fire suppression, is an example (Costanza, Kemp, and Boynton 1995). Similarly, as described earlier, grazing management in semiarid rangelands that attempts to maintain a system of highly palatable and productive, but drought-intolerant, plants leads to high vulnerability to drought.

A major disturbance (e.g., a fire) triggers rapid transition into phase 3, release of stored nutrients, and phase 4, reorganization of the nutrients (e.g., through decomposition), which are then available again for phase 1, exploitation. How the ecosystem will now develop depends on various factors such as what seeds are present and ready to sprout and whether exotic invaders are ready to colonize the site. As in the previous rangeland example, in which woody shrubs take over a moderately grazed grassland system after a drought, a different ecosystem may replace the one that was present before the disturbance.

Not all ecosystems fit this model. Estuaries are one exception, of particular importance because they are among the world's most productive and commercially important ecosystems. Because of extreme fluctuations in temperature, salinity, and water movement over relatively short time periods, estuaries are in a constant state of disturbance and thus never make it to the conservation phase. Estuaries are also highly resilient because even though their species diversity is low, the species that are present tend to be generalists that confer high functional diversity to the system. If disturbed by, for example, a major flood, estuaries return rapidly to their former state because their high hydrologic flux and the mobility and generalist nature of their organisms ensure recolonization and because they have little structural complexity that must be rebuilt (in contrast to ecosystems such as forests and coral reefs) (Costanza, Kemp, and Boynton 1995).

As these examples suggest, and as examined in greater detail in chapter 6, an understanding of the large-scale dynamics of ecosystems is of paramount importance in managing them for commodity production. Attempts to maintain an ecosystem in a single, steady state to optimize and stabilize productivity may have the opposite result.

Ecological Redundancy and Keystone Species

Ecological redundancy and the role of keystone species in ecosystems are two other important factors in the biodiversity-productivity link. Theoretical and empirical studies suggest considerable ecological redundancy among species within ecosystems (Lawton and Brown 1993; Schulze and Mooney 1993; Vitousek and Hooper 1993; Walker 1995). J. H. Lawton and V. K. Brown (1993, p. 267) suggest that "The absolute minimum level of species richness necessary to maintain particular ecosystem functions . . . may be far below pristine levels," though both they and S. L. Pimm (1993) caution that such a conclusion is based on limited evidence.

The level of redundancy will vary among ecosystems and among functional categories within a given ecosystem. For example, functional categories for plants in rangelands in western New South Wales, Australia, include growth form (e.g., tree, shrub, forb), longevity (e.g., annual, biennial, fewer than twenty years, more than twenty years), resprouting ability, drought resistance, seed longevity (dormancy), susceptibility to fire, palatability, and the season during which most growth occurs (Walker 1995). In general, several species occupy each functional group. As B. H. Walker notes (1995, p. 748), "The existence of a number of species within a functional type is an important element in conserving biological diversity because, where one of the member species declines or disappears . . . ecological equivalence allows functional compensation by the other member species."

This is not the case where only one species, by definition a keystone species, represents a functional group. Because keystone species are crucial in maintaining important ecological processes (Paine 1969; Schulze and Mooney 1993; Teer 1997) and the species composition of a community (Paine 1969; W. J. Bond 1993), their use and management require special attention.

Many species of commercial importance are keystone species. Sea otters (*Enhydra lutris*), once heavily exploited for their fur, are key predators in nearshore marine systems and strongly affect biomass at lower trophic levels (Estes and Palmisano 1974). Some anadromous fish, such as salmon, are keystone species with wide-ranging ecosystem influences. Salmonids appear to be an important vehicle for moving marine nutrients into their natal watersheds (Francis 1997). Large numbers of salmon moving into freshwater streams for spawning may be important for maintaining production in the stream regardless of other nutrient sources (Bilby, Fransen, and Bisson 1996; Kline et al. 1990, 1993). Further, salmon are an important food source for many terrestrial vertebrates and thus may play a keystone role in the movement of nutrients from aquatic to terrestrial ecosystems (Francis 1997). Such keystone functions of salmon are strongly linked to their abundance and raise several issues regarding salmon management. One issue is the effect that hatchery-released salmon, by replacing natural spawning runs, have on these ecosystem linkages. Another regards how operators of salmon fisheries manage for what they often call "overescapement," a term meaning that too many adult salmon are returning to spawn, resulting in lower recruitment per spawner. The management implication is that fishing is good for the stock because it absorbs "surplus production" (Francis 1997).

Ungulates and other large herbivores, generally of great importance in CCU, are often keystone species that, by influencing succession, can affect the quality of habitat for other species. High population densities of cervids can cause major changes in ecosystem structure and species diversity (Teer 1997). Similarly, elephants can have widespread effects on the function and structure of savanna ecosystems in eastern Africa (Dublin 1991).

Waterfowl significantly influence nutrient dynamics and energy flow in wetlands (Callaghan, Kirby, and Hughes 1997). For example, waterfowl were found to be responsible for 6 percent of all carbon, 27 percent of all nitrogen, and 70 percent of all phosphorus entering Wintergreen Lake in the United States (Manny, Johnson, and Wetzel 1994). In Tipper Grund, Denmark, T. Kiørboe (1980) estimated that waterfowl consumed about 30 percent of the annual macrophyte production, and in the southern Baltic Sea, L. Nilsson (1980) found that diving ducks consumed about 26 percent of the annual production of food animals.

Antarctic krill (*Euphasia superba*) have become an important food for human populations during the past decade. Their potential as a food source is enormous; the annual production of Antarctic krill probably exceeds, perhaps by several times, the total annual harvest of nearly 90 billion metric tons of marine species from all oceans. Krill, which eat plankton, are also a major food source for large carnivores—seals, fish, squid, seabirds, and baleen whales. Indeed, because of krill's central role in the food web, Antarctica's marine ecosystem is often described as krill based (Nicol and de la Mare 1993).

Oligarchic ecosystems, in which one or a few commercially important species dominate, present special management opportunities and risks. Many pine forests and palm forests are dominated by one or two commercially important species that largely determine the structure of the ecosystem and much of its biodiversity. The marine ecosystem of Antarctica is arguably something of an oligarchic ecosystem dominated by the high productivity of krill populations, and rivers that host massive salmon runs may be viewed as seasonally oligarchic ecosystems. The opportunity with such abundant and dominant keystone species is that the ecosystem naturally approximates a cultivated monoculture. Thus, compared with biologically diverse ecosystems, in which commercially important species occur at low densities, oligarchic ecosystems should have less need for intensive management to increase population densities and productivity of the high-value species (Peters 1992). The risk is that any mistakes in management or harvest rate, to say nothing of blatant overexploitation, can readily lead to degradation of the entire ecosystem (Freese 1997a).

The Design of Harvest Strategies for Sustainable Offtake

P. A. Larkin (1977) concluded his keynote address to the 1976 meeting of the American Fisheries Society with the following epitaph for the concept of maximum sustained yield (MSY):

M.S.Y.
1930s–1970s
Here lies the concept, MSY.
It advocated yields too high,
And didn't spell out how to slice the pie.
We bury it with the best of wishes,
Especially on behalf of fishes.
We don't know yet what will take its place,
But hope it's as good for the human race.

Questionable, perhaps, as poetry, but sound as policy regarding MSY. Larkin argued that the concept of MSY is not sufficient from the biological perspective though it is valuable as an index of production potential and as a first rough cut at management policy for commercially important species. The primary problems with MSY, according to Larkin (1977, p. 9), are that it is "not attainable for single species and must be compromised: (1) to reduce the risk of catastrophic decline and reduction of genetic variability; and (2) to accommodate the interactions among the species of organisms that comprise aquatic communities." R. Hilborn, C. J. Walters, and D. Ludwig (1995, p. 46) suggest that "The simple view of MSY of the 1950s has gradually faded as we have come to recognize the complexity of society objectives, the difficulty in estimating the productive potential of natural populations, and the problems in regulating the exploiters of the resource."

Although the shortcomings of MSY are now widely recognized, Larkin believes that its development in the mid-1930s in fisheries management represented an important initial step in curbing many fisheries problems. Twenty years after Larkin's epitaph, however, we are still uncertain what should take MSY's place as a guide or strategy for designing sustainable harvest programs, whether in fisheries, forestry, or recreational hunting. Indeed, the epitaph for MSY appears to have been premature, as the goal of maximizing profit by maximizing yield is still the driving force behind many CCU programs. Rather than backing off from the goal of MSY, the major conceptual change seems to be that scientists and managers have become more attuned to the fact that *sustained* seldom means "invariable," and thus management seeks a *variable* MSY.

Why the continuing confusion? In theory, the design of sustainable harvest programs should be simple. Sustainable offtake from a wild species population requires the existence of a reproductive surplus. If an annual plant produces five seeds and one seed falls and sprouts to grow into another reproducing plant, the other four seeds theoretically represent a surplus that can be harvested (Hilborn, Walters, and Ludwig 1995). This, of course, is an oversimplified approach to setting harvest levels. It may work well where sustainable offtake is the only goal, but where ecological sustainability is the goal, we must ask what other animals eat those four seeds and what the effect on them and the ecosystem would be if we harvest one, two, three, or all of the seeds. These broader ecological effects are covered in chapter 6.

The Problem with Maximum Sustained Yield

Underlying the concept of maximum sustained yield was the belief that nature is constant and that if disturbed by humans, it returns to its former status. The initial development of MSY was, in fact, based on experimentation with animal populations under such conditions of constancy—a laboratory enclosure with constant environmental conditions and supply of food. Growth of populations under these conditions generally follows the classic S-shaped logistic growth curve, whereby population growth levels off and becomes zero at the carrying capacity, the number of organisms a habitat can support and the point at which the population remains constant unless disturbed. If disturbed by, for example, the harvest of some individuals, the population will recover to that carrying capacity. Under conditions of logistic growth, maximum population growth—that is, MSY—is achieved when the population is at exactly one-half the carrying capacity. Thus, harvesting for MSY entails harvesting the population down to one-half of its carrying capacity every year (Botkin 1990).

D. B. Botkin, however, suggests that several conditions must be in place for MSY to work. First, the population must have a single, constant carrying capacity and its growth must follow the logistic growth curve. However, constancy is the exception rather than the rule in ecosystems and wild species populations, and wild species populations often do not follow the logistic growth curve. D. R. McCullough (1992, p. 967), for example, warns that following the logistic model where MSY is obtained at one-half the carrying capacity "is a formula for overexploitation of large herbivores." His analysis suggests that this admonition may hold true for most slow-reproducing, K-strategist species.

Second, managing for MSY requires a clear understanding of the relation between harvest mortality and natural mortality. If harvest mortality is fully compensatory, humans can harvest the number of individuals that would have died naturally with no negative effect on the population. To return to the seed example, if, on average, two of every five seeds dropped by a plant die naturally, under fully compensatory mortality the harvest of two seeds by humans would reduce natural mortality to zero. That is, human seed predation would be perfectly compensated for by a lower natural loss. But if humans harvest two seeds and two additional seeds are still lost to natural causes, leaving just one seed, we have a case of additive mortality. The degree to which compensatory versus additive mortality is at work is still unclear for such commercially important organisms as waterfowl (Johnson and Owen 1992) and furbearers (Clark and Fritzell 1992). Among mallard ducks, for example, hunting mortality is compensated by reduced natural mortality up to some unknown threshold harvest rate (Johnson and Owen 1992). Yet despite considerable research, this relationship has not been clearly established for any other duck species (Johnson and Owen 1992), and hunting mortality is additive in many goose species (Ebbinge 1991). For short-lived, fast-reproducing (i.e., r-strategist) furbearer species, such as the muskrat (*Ondatra zibethicus*), harvest and natural mortality appear to be compensatory, whereas for longer-lived carnivores, such as raccoons (*Procyon lotor*) and foxes (*Vulpes* spp.), evidence suggests that increased harvest is additive (Clark and Fritzell 1992).

The third condition proposed by Botkin (1990) is that managers must know both the carrying capacity and the present population size. The complexity and variability over time of factors that affect how large a population an area can support make estimations of carrying capacity difficult or, often, impossible. In fact, where climatic variance is high and ecosystems exhibit large fluctuations, carrying capacity as a concept may be largely an abstraction of little usefulness in management (Caughley, Shepherd, and Short 1987). Moreover, except perhaps in well-funded forestry and wildlife management operations, accurate counts of individuals in a population can seldom be obtained. In most regions of the world, financial and technical constraints severely limit the ability to accurately assess population size and factors that affect it.

Finally, full cooperation, as well as considerable skill, from those harvesting the resource is required so that exactly the right number is harvested. Unless the incentives are just right for the harvesters, such cooperation is difficult to achieve. Moreover, even if harvesters wish to cooperate fully, they may lack the skill to carry out the harvest as prescribed by management—they may overshoot or undershoot their quota.

MSY is criticized as a management goal because usually few if any of these conditions are met in the real world. The decline in the catch of Atlantic menhaden (*Brevoortia tyrannus*) from 785,000 tons in 1956 to 178,000 tons in 1969 and that of North Atlantic haddock from 155,000 metric tons in 1965 (after years of averaging 50,000 tons) to 12,000 tons in the early 1970s are two well-known examples in which MSY-based estimates led to overexploitation (Botkin 1990).

Despite these problems, MSY still appears as either an explicit or an implicit goal in marine fisheries operations (Hersoug 1996; Sissenwine and Rosenberg 1993), forestry (Dudley, Jeanrenaud, and Sullivan 1995), and wildlife management (Clark and Fritzell 1992; McCullough 1992). J. G. Robinson and K. H. Redford (1991a, p. 415), in examining the management of Neotropical wildlife, state that a requirement for sustainable harvest of wildlife "is that the maximum production from the population for human use is achieved" without endangering the population or affecting ecosystem function. Indeed, where commercial interests prevail and profitability is based on quantity rather than quality of the commodity, it is difficult to imagine how anything less than offtake close to the MSY would be acceptable. The concept of MSY, however, is less of an issue in some forms of recreational hunting and fishing in which greatest profitability is achieved by managing for quality in part of the population. For example, a goal of producing male ungulates with large horns or antlers or of producing large fish that fight vigorously subjects the population to less risk of overharvesting than do goals of maximizing harvest from the total population. B. C. Lubow, G. C. White, and D. R. Anderson (1996, p. 795), in discussing harvest strategies for cervid populations, observe that "Male-only harvests are robust and have been conducted safely over extended periods with a minimum of information." They caution, however, that attempts to optimize harvests of females require much more information, which can be costly and difficult to obtain.

Life Histories and Harvest Strategies

As discussed earlier, the most common way to categorize a species according to its life history strategy is to describe it as either an r-strategist or a K-strategist. The *r* in *r-strategist* refers to the fact that these species' life history strategies assign priority to reproduction. They are primarily short-lived, fast-reproducing, fast-growing species that put a relatively large amount of energy into rapid growth and production of abundant offspring that can colonize new habitats. Large, periodic environmental fluctuations, as are found in estuaries, favor r-strategists (Costanza, Kemp, and Boynton 1995), and thus reproduction is generally density independent. Examples of r-strategists are sardines, anchovies, rabbits, deer (generally r-strategists relative to other large herbivores), peccaries, and most plant species that colonize newly disturbed sites.

The *K* in *K-strategist* is the symbol generally used for carrying capacity. Species with this life history strategy are slow-growing, slow-reproducing, long-lived species that generally live under conditions in which their population is at or near carrying capacity. Thus, their habitats are generally more stable than are habitats of r-strategists, and such species tend to put more energy into maintenance than into growth. All these characteristics represent a largely density-dependent reproductive strategy, such that the reproductive rate goes up when the population density goes down. Examples of K-strategists are whales, sharks, elephants, primates, ginseng, and most canopy species in old-growth forests.

The r/K distinction has important consequences for the design of harvest strategies. K-strategists, for example, will tend to have low levels of sustainable harvest despite large standing crops, and thus they are more vulnerable to overharvesting (Mangel et al. 1996; McCullough 1992). In an analysis of population growth among large herbivores, D. R. McCullough (1992, p. 980) found that achieving maximum sustained yield in K-strategists required keeping the population much closer to carrying capacity than the one-half K predicted by the logistic growth equation. He concludes that "Exploitation of the most K-selected species . . . is akin to harvesting of old growth forests."

However, to the extent that K-strategists exhibit density-dependent reproductive strategies, managers should be better able to predict population responses to offtake, and thus harvest protocols should be easier to design than for species exhibiting density-independent growth patterns. The difficulty lies in the implementation—the maintenance of low harvest levels (Freese 1997a). G. P. Kirkwood, J. R. Beddington, and J. A. Rossouw (1994, p. 226), in examining reproduction in fish stocks, suggest that longer-lived species "retain a resilience to catastrophes that is not available to the short-lived species." They caution that though short-lived species are capable of producing high sustained yields, they "can be particularly vulnerable to a combination of high exploitation and occasional environmental events that devastate the spawning stock." This seems to describe the situation in two well-known collapses of r-strategist populations—the Peruvian anchoveta and the passenger pigeon. The problem with r-strategists, then, is that because of their density-independent reproductive strategy, once a population is knocked to low levels it may not bounce back, and in fact, if it is truly density independent, its next movement may be further downward.

In most cases, however, r-strategists appear to fare better than K-strategists under intensive harvesting, as demonstrated in both African and Neotropical forest wildlife. In the Tamshiyacu-Tahuayo Communal Reserve in Amazonian Peru, several primate species and the tapir (*Tapirus terrestris*), all with low reproductive rates, exhibited depressed densities due to hunting. In contrast, hunting of species with higher reproductive rates, such as peccaries (*Tayassu* spp.), deer (*Mazama* spp.), and rodents, was apparently sustainable, even though they constituted most of the harvest (Bodmer 1995; Bodmer et al. 1997). Similarly, market hunting in the Bioko and Río Muni regions of Equatorial Guinea appeared to be unsustainable for slowly reproducing primates and for one ungulate with a relatively low reproductive rate, the bay duiker (*Cephalophus dorsalis*) (Fa et al. 1995). And in Kenya's Arabuko-Sokoke Forest, hunting was unsustainable for populations of large ungulates and two primate species, whereas offtake rates for the fast-reproducing elephant shrews (*Rhynchocyon* spp. and *Petrodomus* spp.), squirrels (*Heliosciurus* spp. and *Funisciurus* spp.), and duikers (*Cephalophus* spp.) were sustainable (Fitzgibbon, Mogaka, and Fanshawe 1995).

The determination of sustainable harvest levels presents special problems where some part of the organism, rather than the organism itself, is harvested. Because the effects on survivorship and reproduction in the population are less direct, changes are difficult to detect and monitor. Nontimber forest products, such as fruits and latex, are most prominent in this category (e.g., see Bodmer et al. 1997; Boot and Gullison 1995; Clay 1997a, b; Singh et al. 1997). C. M. Peters (1994), for example, cautions that the collection of commercial quantities of fruits and seeds can cause changes in the structure and dynamics of a tree population and, if uncontrolled, can result in its gradual extinction.

One logical response to resource fluctuations, both natural and human induced, is opportunistic or pulsed harvesting. A pattern of overexploitation, population recovery, and repeated overexploitation may result. This has arguably been an effective strategy under conditions of low human population density for migratory systems of swidden agriculture and hunting in the lowland tropics—deplete the local resources, move on to more productive lands while the exhausted ones recover, and so on (Hart 1978; Hart and Hart 1986). This is also the traditional strategy for African pastoralists in the semiarid savannas of Africa discussed previously. Such a strategy may be viable where human population density and consumption are low relative to the geographic scale of the resource. Unfortunately, few if any such frontier conditions now remain anywhere in the world for wild species of significant value for CCU (Postel 1994), and thus this strategy rarely seems feasible.

Population Size and Reproductive Surplus

What population level results in the greatest reproductive surplus? For the manager of wild species, the more precise question is, How far should the population be reduced to maximize total reproductive output? According to R. Hilborn, C. J. Walters, and D. Ludwig (1995, p. 49), "This question plagues everyone trying to determine the sustainable yield and optimal harvesting strategy for natural populations."

The relation between population size and reproductive surplus is a major point of controversy in fisheries management (Hilborn, Walters, and Ludwig 1995). T. H. Huxley (1881, 1884, cited in Hilborn, Walters, and Ludwig 1995) was the first to argue that since fish generally produce enormous numbers of eggs, the size of the adult (breeding) population, and therefore the number of adults removed by fishing, should have no effect on the number of fish surviving to reproductive age. More recently, J. A. Gulland (1983) reached a similar conclusion regarding the relation between adult population size and reproductive surplus, whereas R. A. Myers and colleagues (1994) concluded that in general, reproductive surplus declines with a decline in the adult population.

The relation between adult population size and reproductive surplus is less controversial in wildlife management because birds and mammals do not produce such large numbers of offspring. The need to maintain a sizable adult population is therefore easily recognized (Hilborn, Walters, and Ludwig 1995). The issue in traditional wildlife management is directed more at how large the adult population must be relative to carrying capacity in order to maximize the reproductive surplus. That proportion increases to as high as 0.8 or more as one moves across the spectrum from r-selected to K-selected species (McCullough 1992).

Forest ecologists have circumvented the issue of reproductive surplus by managing forests so that seed sources are near enough to ensure adequate regeneration or by directly planting seedlings, combined with practicing silvicultural techniques that ensure adequate habitat for recruitment (Hilborn, Walters, and Ludwig 1995). This may generally be an attainable management goal for oligarchic temperate forests, where large areas can be opened in the process of logging and thus made available for regeneration. In some species-rich tropical forests, however, ensuring adequate recruitment of commercially valuable species is often much more difficult. In some Neotropical forests, the density of marketable species such as mahogany is so low that the gaps opened during logging may be inadequate for regeneration (Boot and Gullison 1995; Kiernan and Freese 1997), both because mahogany requires ample light for regeneration and because of the high probability that other species of lower commercial value may colonize the site first (Hartshorn 1994). In contrast, the forests of western Africa and, particularly, Southeast Asia have numerous marketable species that readily colonize disturbed sites, and thus managing recruitment in logged sites poses less of a problem in commercial forestry (Whitmore 1991).

The "Fall-Down" Problem

Fall-down in forestry refers to a rapid decline in yield after initial harvest of a large, often old-growth stand of trees. This decline in yield is common to other sectors of CCU as well. Previously unharvested populations may contain numerous old individuals of large size, and there may be nonreproductive subpopulations. The harvest of these individuals and subpopulations often creates a windfall yield in the first harvest cycle (Hilborn, Walters, and Ludwig 1995).

This problem is acute among long-lived species of fish, in which the initial yields of a previously unharvested population may exceed the sustainable yield by several orders of magnitude. For example, if reproduction in a fish stock fits the logistic growth curve, the optimum stock size for producing a sustainable yield is one-half the size of the unfished stock. Thus, in the fishing-down phase, 50 percent of the stock will have to be taken during the initial harvest. If the annual mortality rate of adult fish is 20 percent, the yield while fishing down will be sixteen times the annual sustainable yield (Hilborn, Walters, and Ludwig 1995).

The unsustainable high yields obtained during the first harvests of virgin stocks in fisheries and forests create problems at the outset for long-term sustainability. Managers of the stocks develop expectations for higher yields than are sustainable and thus make investments in infrastructure that exceed those required for a sustainable level of production. When the inevitable fall-down occurs, it is difficult to curtail harvest effort because of the large investments made. Government subsidies and excessive quotas may be sought. The result is that populations are driven below the most productive sustainable levels and the fall-down is even greater than necessary (Hilborn, Walters, and Ludwig 1995).

Harvest Refugia

In contrast to parks and protected areas established to protect species within their boundaries, harvest refugia are designed to further the goal of harvesting wild populations outside the refugia boundaries. The concept of protected areas as a harvest management tool is not entirely new. For example, the marine reserve concept has probably been employed both incidentally and purposefully for some time. J. A. Bohnsack (1993) suggests that until recently, most reef fisheries were probably maintained by natural refuges, areas too deep or remote to be readily fished. Recent advances in fishing technology, however, have made these places increasingly less secure. R. Johannes (1978) identifies several traditional fishing societies in Oceania that maintained areas closed to fishing as a management tool. He cites the example of a chief at Satawal closing part of a reef to fishing in order to maintain spawning stocks to replenish surrounding reefs. N. V. Joshi and M. Gadgil (1991) cite the creation of sacred groves and sacred ponds by traditional societies in India as possible examples of resource management through the use of harvest refugia. Aldo Leopold (1936, p. 195) outlined this same concept for the developing field of wildlife management early in the twentieth century when he defined *game refuge* as "an area closed to hunting in order that its excess population may flow out and restock surrounding areas." Game refuges declined in importance as a management tool in North America when game populations recovered and harvest quotas became widely used (McCullough 1996). The development of waterfowl refuges in North America is based on this concept in that their primary function is to maintain habitat critical for migratory waterfowl and the large hunting enterprise they support (Greenwalt 1978).

Modeling in fisheries management indicates that the major benefit of harvest refugia is the increase in spawning stock biomass (Polacheck 1990). Various marine reserves representing a broad geographic range show higher population densities and greater average fish sizes than do surrounding fished areas (Roberts and Polunin 1993). This effect enhances overall recruitment in two ways: (1) more fish are able to spawn; and (2) larger fish produce many more eggs than do smaller fish. For example, one 61-centimeter red snapper (*Lutjanus* spp.) can produce as many eggs as do 212 fish that are 42 centimeters in length (Bohnsack 1993). Fishing reserves are also important in protecting fisheries from stock collapse (Bohnsack 1993). In particular, they may be crucial in protecting the spawning stock of large, long-lived reef species, such as sharks and the Caribbean grouper (*Epinephelus itajara*), that are especially vulnerable to overfishing because they become sexually mature only after reaching harvestable size (Roberts and Polunin 1993). Protection of spawning stock within a reserve requires significant dispersal of eggs, seed, or young into the harvest area if harvest refugia are to be a tool for sustainable harvest programs. C. Roberts and N. Polunin (1993, p. 367) caution, however, that despite intensive research "the larval dispersal stage remains a virtual black box," and they conclude that "The jury is still out on whether marine reserves provide the hoped for answer to managing the fisheries of coral reefs" (p. 368).

In coral reef fisheries, in addition to protecting spawning stock, harvest refugia may be more effective than conventional methods of fisheries management because they require less detailed information on species' life histories, are more suitable for the multistock fisheries of reefs, and pose fewer enforcement problems (Roberts and Polunin 1993).

Species mobility, at both the seed, egg, and larval stage and the adult stage, is a major determinant of how large harvest refugia should be and where they should be located. All else being equal, species with highly mobile adults will require larger areas than will those with less mobile adults in order to ensure that a significant portion of the population never moves outside the reserve into harvest areas (McCullough 1996; Polacheck 1990). At one extreme, a harvest refugia may be no more than a single seed tree left intact in a stand of harvested timber, although ecologically sustainable forestry requires much larger old-growth stands to serve as a source for reestablishment of species and ecological processes in logged stands. Mobile species, such as many reef fish and upland game birds, require several to hundreds of hectares to ensure that a significant portion of the population is protected from harvesting. For highly migratory species, such as many waterfowl, large ungulates, and pelagic fishes, harvest refugia will not work as the principal regulator of harvest because all or a large portion of the population would be exposed to harvest during part of the year. Finally, exact placement of the harvest refugia is crucial for fish species that form large spawning aggregations (Polacheck 1990).

How extensive should marine reserves be to serve as effective tools for fisheries management? J. A. Bohnsack (1993) suggests that 10–20 percent of the continental shelf should be in nonfishing reserves, whereas C. W. Clark (1996) cites recent work suggesting that marine reserves may need to cover 50 percent or more of the area occupied by a stock as a hedge against overexploitation.

D. R. McCullough (1996), concluding that harvest programs based on quotas alone are highly subject to overexploitation, has developed a simple model for developing MSY for wildlife based on a harvest refugia system. The system consists of a grid of squares within a large block of landscape, each grid large enough to support a viable population of the target species. The process begins with a large proportion of the grids protected from hunting. Then, over a period of harvest seasons, more grids are progressively opened to hunting until, based solely on careful monitoring of the number (or some numeric index) of animals harvested, the optimal ratio of protected to hunted grids is attained. The manager knows that too many grids have been opened to hunting when the number of harvested animals begins to fall off (figure 5-4). At this point, the removal of animals has exceeded replacement over the entire population within the set of grids. However, unlike a quota system, in which an overharvested population can quickly be driven to near extinction levels, in a hunting refugia system, a significant portion of the population is protected by the unhunted squares. In such a system, no quotas are required within the hunted areas; indeed, all animals could be removed every year from them. Recruitment into the population may come entirely through dispersal from the protected areas. MSY is determined by finding the number of squares that yields the greatest harvest. Grid size is a compromise between two needs—large enough to maintain a viable population size and small enough to enable rapid dispersal of animals into the adjacent hunting areas. Using a somewhat different approach, N. V. Joshi and M. Gadgil (1991) also concluded that harvest refugia may provide a less costly approach, in terms of both research needs and management costs, to achieving MSY.

A major advantage of the harvest refugia system is that no sophisticated knowledge of the population biology or even population level of a species is required. Only careful monitoring of the trends in animals harvested is needed, a much more viable alternative for regions where resources for research are scarce or populations and ecosystems simply

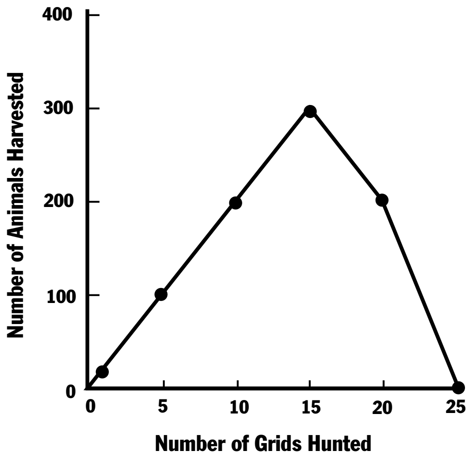

Figure 5-4. *Expected relation between number of grids hunted and total harvest in a hypothetical area divided into twenty-five grids, based on a model for developing MSY of wildlife in a harvest refugia system.* Source: *McCullough 1996.*

defy the acquisition of sufficiently detailed population information. Although confining hunters to hunting areas and monitoring harvest levels can be difficult, it is a less formidable task than regulating of hunters' behavior and harvest numbers as required in quota systems (McCullough 1996).

A version of the concept of harvest refugia has been suggested as a management tool for lynx in areas of high accessibility and unlimited entry for trappers. B. G. Slough and G. Mowat (1996) recommend a minimum refugia size of 500 square kilometers, spaced 50 kilometers apart in continuous habitat, thus protecting 15 percent of the habitat. The benefits of such a network would be facilitation of normal lynx population responses to changing densities in snowshoe hare populations, prevention of local extinctions, and maximization of lynx harvests over a complete population cycle. A variation of this model, closer to that proposed by Joshi and Gadgil in which a single large protected area and hunting area are paired, is being tested in the Tamshiyacu-Tahuayo Communal Reserve in Amazonian Peru. The reserve includes a fully protected core area, a principal function of which is to replenish wildlife populations that may become depleted in the buffer zone of the reserve and beyond (R. Bodmer, pers. comm., 1996; Bodmer et al. 1997).

Because the McCullough-Joshi-Gadgil model is a single-species approach, it requires that attention be given to broader ecosystem effects wherever it is applied. For example, animals may move out of hunted grids and into unhunted grids (see the waterfowl example that follows), where their high numbers could cause habitat degradation.

Besides protecting breeding populations and serving as sources of recruits, harvest refugia protect key ecological processes and function as a control for monitoring natural ecosystem change and studying the effects of harvest on natural populations and systems (Agardy 1994; Bohnsack 1993; Lindeboom 1995; Roberts and Polunin 1993). In forestry, reserves within logged areas help maintain key ecosystem processes, act as refugia for species that require forest interior habitats, and facilitate seed recruitment into logged areas (DellaSala et al. 1996; Hansen 1997). As in fisheries and wildlife reserve models, patch size and location are key questions facing forest managers.

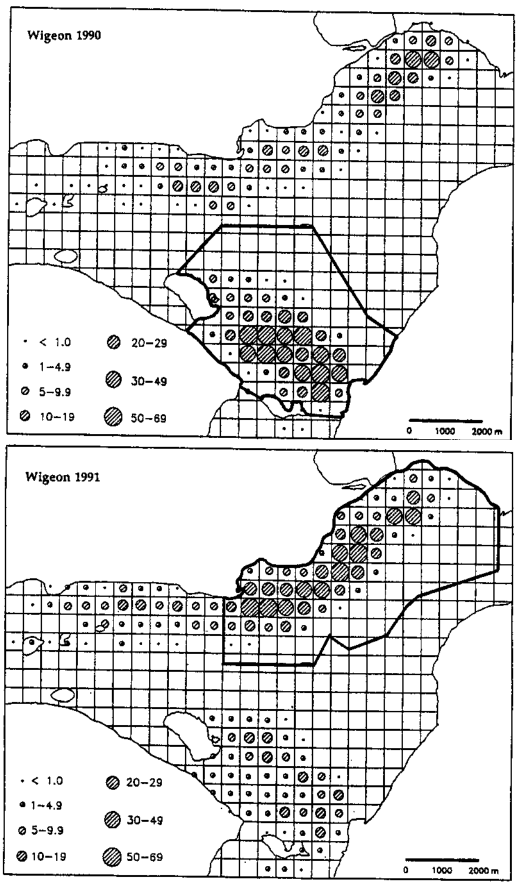

Figure 5-5. *Distribution of wigeon-days during autumn at Nibe-Bredning, Denmark, in 1990 and 1991. In 1990, shooting was banned in the southern part of the area (within bold line); in 1991, shooting was banned in the northern part.* Source: *Madsen 1995.*

Another potential use of harvest refugia is for managing the effects of hunting disturbance on wildlife movement and, by implication, on a variety of ecosystem features. Shooting disturbance and the location of nonshooting areas, for example, strongly influence the movement and distribution of waterfowl (figure 5-5) (Callaghan, Kirby, and Hughes 1997). Given these effects, and the broader ecosystem effects resulting from the major role waterfowl play in nutrient flow in wetlands, D. A. Callaghan and co-workers recommend the creation of refuge areas within hunting zones, with research focused on their optimal configuration.

Ecological Knowledge: Merging Western Science with User Experience

Fishers, hunters, forest extractivists, and other primary users of wild species commodities often acquire detailed knowledge about the species they harvest and the ecosystems within which they operate. In traditional societies where there is a long history of resource use, this knowledge represents the collective experience and wisdom of generations of users. Such knowledge is highly contextual, conforming to both specific resources and areas used and the immediate social environment of the user. The maintenance or enhancement of biodiversity is often a central feature of these traditional societies. This collective knowledge, highly practical and attuned to the needs of the user, is tested daily, monthly, and yearly through direct feedback in terms of success or failure of the user's efforts. The depth and application of this knowledge are widely evident in cultures that have a long history of dependence on resources from natural ecosystems, from forest dwellers to coastal fishing communities, from the Arctic to the tropics. Although it may not always be applied in a sustainable manner, the knowledge is substantial (Gadgil and Berkes 1991; Gadgil, Berkes, and Folke 1993).

Western science, meanwhile, gathers and analyzes information on species and ecosystems and their interactions much more systematically and at generally larger scales than those of individual users. Thus, a greater understanding is obtained regarding large-scale trends and interactions, though the social and cultural context of this knowledge is often lost. The goal in terms of managing species is to be able to predict responses of populations and ecosystems to different harvest and management interventions. Feedback is less direct, as knowledge and management recommendations are filtered first through policy mechanisms and then through their application by user groups. Again, the knowledge and predictive powers of Western science are only as good as the policy maker's and resource user's willingness to apply them.

There has yet to be an effective merger of these two fundamental sources of knowledge for managing wild species and ecosystems. This problem is in part a result of the comanagement issues addressed in chapter 4—that is, the difficulty of finding the right balance between local, customary rules of resource management and government control and oversight. Local knowledge serves the former, and Western science generally serves the latter. As governments assumed ever greater control of forest, fisheries, and other natural resources over the past three centuries, Western science increasingly displaced, rather than complemented, local sources of knowledge. This problem is acute in marine fisheries, in which, according to D. Symes (1996, p. 9), "The contribution of generationally transmitted knowledge and experiential understanding has been replaced by experimental research, sample surveys and the linear programming of the results to estimate future fish stocks. In policy terms, this technocratic approach to the understanding of the resource has led to the substitution of flexible strategies, developed in the context of particular local fisheries, by rigid regulatory frameworks applied over much larger territories."

In the early development of fisheries science, confidence in the ability of science to accurately model and predict the behavior of fish stocks in response to different levels of exploitation led scientists and managers to ignore or, worse, disdain locally acquired knowledge. The science of fisheries management seemed like a relatively straightforward exercise, so why bother with the practical knowledge of local fishers? (Symes 1996). That attitude has slowly begun to change in recent years, for two interrelated reasons: (1) we have come to appreciate the unpredictable, seemingly chaotic behavior of fish stocks and the marine environment; (2) science has frequently failed to provide useful knowledge and models for fisheries management. If fisheries systems are chaotic, then science's numerical approach to predicting and controlling through harvest the abundance of fish stocks is largely unfeasible. Quota systems have not worked and will not work (Beverton 1994; Wilson et al. 1994). Under such conditions, it is likely that "a system based on the application of predictive science is likely to get it wrong almost as often as it gets it right" (Symes 1996, p. 7).

A management system adapted to the unpredictable behavior of populations and ecosystems will have to involve less regulation based on the predictive models of science and allow much greater flexibility to respond to changing local conditions. This will require much greater input of practical knowledge and experience from traditional resource-use communities, where flexibility is a fundamental management strategy. A more decentralized management structure that places greater responsibility on local communities will be needed to accomplish this (Gadgil and Berkes 1991; Ruddle, Hviding, and Johannes 1992; Symes 1996; Wilson et al. 1994; chapter 4). Such an integration of local experience with scientific approaches may have, for example, better anticipated the collapse of cod (*Gadus morhua*) stocks in the northern Atlantic at the end of the 1980s. In this case, while predicting an increasing stock abundance, scientists ignored warnings by inshore fishers that something was wrong as they saw catches declining and a higher percentage of small fish in their nets (Finlayson 1994). A. C. Finlayson's skillful examination of scientific management in the cod fisheries shows that its failure in this case is not due solely to the complexity and uncertainty of the marine ecosystem. It has as much to do with the way scientific objectivity is skewed by competition for power and authority among scientists and policy makers. Scientific objectivity can be most easily, and often unwittingly, compromised when high degrees of uncertainty prevail in the data and models, as often occurs in marine fisheries. Then "interpretive flexibility" allows a priori equally plausible alternative conclusions to be reached from the same data.

The need to integrate the knowledge systems of science and local users seems greatest and most productive where (1) the predictive abilities of science are limited either because there is inherent complexity or unpredictable behavior in the population and ecosystem (as in migratory fish stocks in marine ecosystems) or because relative ignorance of the ecosystem will remain significant for the foreseeable future (as in many tropical forests) and (2) practical knowledge can be obtained from communities with long (i.e., several generations and therefore, arguably, sustainable to some degree) traditions of natural resource use. Such an integration will require that the adaptive management approach to science take the management-science partnership (Walters and Holling 1990) one big step further to incorporate the knowledge and monitoring skills of local people into resource management. The goal is not only better science and more useful knowledge, but also a greater understanding and more participation by users in developing the knowledge and a greater acceptance by them in applying it.

Summary

Natural ecosystems and commercial markets are uneasy bedfellows. Whereas markets prefer steady and predictable supplies, natural ecosystems produce highly variable and unpredictable flows of CCU products. Further, the variable and unpredictable nature of wild species populations, and general ignorance about how they respond to harvest and management, often lead to overexploitation when commercial goals of profitability drive managers to strive for maximum sustained yield from populations. Although maximum sustained yield has been widely criticized as a management strategy, it continues to guide the management of many CCU products, from timber to fish.

Differences in the life histories of wild species and in the ecosystems they inhabit result in a diversity of approaches to developing sustainable harvest strategies. For some types of CCU, such as timber production in coniferous forests and recreational hunting of big game, managers can determine with some confidence sustainable levels of offtake. In contrast, in marine fisheries, the ability of scientists and managers to predict sustainable harvest levels is weak at best. Adaptive management, harvest refugia, and tapping of the experience and customs of traditional local resource users are being promoted as supplements to, and at times as replacements for, the predictive approach of Western science.

The link between sustainable offtake and ecological sustainability presents two pivotal ecological questions in CCU management, regardless of the species and ecosystem. One question regards how management can direct more energy and nutrients toward increasing the yield of CCU products without unduly simplifying ecosystems and sacrificing biodiversity, and without undermining ecosystem functions that are important for long-term productivity. The other concerns the effects of different types and intensities of offtake from target populations, particularly where keystone species are involved, on ecosystem integrity and biodiversity. In both cases, there are clearly thresholds beyond which we can expect ecosystem functions, long-term productivity, and biodiversity to be compromised.

CHAPTER 6

Biodiversity Consequences of Production Specialization

Naturalism . . . an effort to avoid artificiality in the manipulation of natural processes.

Game management and forestry grow natural species in an environment not greatly altered for the purpose in hand, relying on partial control of a few factors to enhance the yield above what unguided nature would produce.

—Aldo Leopold (1936)

A basic economic principle is that commercialization favors economic efficiency, which leads to specialization in production (Randall 1981; Swanson 1992b). Within any biologically diverse landscape, some genes, species, and ecosystem functions will have greater economic value than others, leading people to simplify and homogenize the landscape to increase production of those more highly valued resources. High-value wild species commodities, which often represent one or only a few species within an ecosystem, are prime candidates for economic specialization. Yet specialization in the production and use of such species is grease for the slippery slope between a biologically diverse natural ecosystem and a homogenized agroscape. This continues to be a major challenge in the new paradigm of ecosystem management (Salwasser 1994). Specialization in the production of commodities from natural ecosystems is an arena in which market demands and ecological constraints come face to face, often with pernicious consequences for biodiversity.

Motives for Specialization

The incentives that lead to specialization in production of a wild species commodity are the antithesis of those that lead to overexploitation.

Whereas overexploitation arises from poorly defined property rights, open-access conditions, high discount rates, and other disincentives for long-term stewardship, investment in production specialization occurs only when the resource owner perceives that his or her investment in and rights to the resource are secure and that long-term stewardship will reap greater socioeconomic benefits than will mining of the resource for immediate gain. No individual would rationally decide to plant high-value timber trees in a forest without the certainty that he or she or an heir would be able to harvest and sell those trees when they reach marketable size fifty years later.

Investments in commodity specialization are favored when the resource owner has a strong desire to generate profits from the natural ecosystem over the long term, when one or a few species have much greater commodity value than do other species in the ecosystem, and when the owner is not receiving compensation for biodiversity values that may be sacrificed by production specialization. Thus, the owner of a mangrove estuary may replace mangroves with artificial ponds to increase the production of shrimp because he or she receives no compensation for maintaining the natural ecosystem and biodiversity from, for example, bird-watchers, biodiversity conservation organizations, or offshore fishers who depend on the mangrove as a nursery ground.

Motives for specialization are not limited to stakeholders who profit directly from the commodity. Biodiversity conservation stakeholders may support specialization in order to generate sufficient revenues to offset the opportunity cost of an alternative land or water use. This may be particularly so in biologically diverse landscapes, where economically important species often occur at low densities (Freese 1997a). Attempts by conservationists to increase the production of mahogany in Neotropical forests are an example (Highsmith 1996; Kiernan and Freese 1997).

Production specialization is common because high-value species often grow slowly, grow at low densities, grow erratically, experience large or unpredictable fluctuations in population level, grow not at all where they are wanted, grow with less than optimal qualities, or are too mobile to be reliably available (table 6-1) (Freese n.d.). Add to these problems the difficulty of capturing or harvesting remote or elusive species and it becomes obvious why on a global scale only a few dozen plant and animal species have been domesticated for production under highly controlled conditions.

Table 6-1. *Production Problems Posed by Species of High Commercial Value for Consumptive Use*

TYPE OF PROBLEM	EXAMPLES
Grow or reproduce slowly	• Orange roughy, sturgeon, and many species of skate, shark, and tuna—slow maturation and reproduction[a] • Large whales and elephants—one offspring, slow maturation[b] • Many commercial tree species, especially hardwoods—require several decades to more than a century to reach harvestable size • Cycads, aloe, and wild leeks—slow growth and reproduction[c]
Grow at low densities	• Commercial tree species in species-rich Neotropical forests[d] • Many nontimber plant products in tropical forests[e] • Moose in areas of heavy wolf predation[f] • Most large predators
Intermingle with low-value species that interfere with harvesting	• Shrimp—catch of nontarget species exceeds the catch of shrimp[g] • Commercial tree species in species-rich tropical forests
Experience large and often unpredictable population fluctuations	• Anchovies, sardines—several orders of magnitude of change in populations[h] • Spruce—large fluctuations due to outbreaks of spruce budworm[i] • Lynx—periodic tenfold or greater change in population density[j]
Do not grow where wanted	• Eucalyptus and lodgepole pine—widely planted outside native range • Nile tilapia and rainbow trout—widely introduced outside native range • White-tailed deer and red deer—widely introduced outside native range • Mallard and ring-necked pheasant—widely introduced outside native range
Grow with less than optimal quality	• Many hardwood tree species—straighter boles preferred • Wild sheep—larger horns in bigger demand than smaller horns • Clams—undesirable milky appearance during egg production
Are highly mobile	• Most large herbivores and large predators • Most open-ocean fish and marine mammals • Migratory waterfowl

[a] Norse 1993; Weber 1994.
[b] Nowak and Paradiso 1983.
[c] Cunningham 1994; Nantel, Gagnon, and Nault 1996.
[d] Whitmore 1991.
[e] Lawrence, Leighton, and Peart 1995.
[f] Gasaway et al. 1992.
[g] Alverson et al. 1994.
[h] Cury 1993.
[i] Baskerville 1995.
[j] Keith 1963; Slough and Mowat 1996.

Methods of Specialization

Specialization is generally aimed at meeting one or more of four basic production goals:

1. Maximize or increase the number of individual organisms or the biomass harvestable from the population or some segment of it (e.g., trophy bucks and trophy fish) and reduce any fluctuation in those numbers.

2. Improve and standardize the quality of the product.

3. Enhance the harvestability of the wild species commodity and the efficiency (technology and skill) of the harvester.

4. Manipulate the wild species commodity and its environment to better secure property rights to it.

Here, rather than overexploitation and depleted target populations being the concern, questions emerge about the effects of specialization and augmented target populations on biodiversity.

The starting point for production specialization in a wild species commodity is development of a program of sustainable offtake for the target population or the most commercially valuable members of the population. MSY is often the goal. Two issues are of concern in terms of the ecological sustainability of offtake:

1. How the sustainable removal of individual organisms or their parts from the population affects the ecosystem.

2. How the harvest technique affects other species populations and the ecosystem.

Effects of Sustainable Offtake on Ecological Sustainability

What factors can help predict when sustainable offtake will affect biodiversity? Offtake of keystone species should have greater repercussions in the ecosystem than offtake of nonkeystone species. Taking the MSY of an uncommon species of tree in a species-rich rain forest will probably have less effect on biodiversity and ecosystem function than would taking the MSY of the dominant species in an oligarchic Scandinavian fir forest or a Neotropical mangrove forest. Offtake that does not attempt to reach MSY will on average have less effect on an ecosystem than will harvest at MSY. Harvesting that concentrates on a distinct segment or growth form of a population will be more likely to cause genetic change in the population than will less selective harvest regimes.

Genetic Effects of Sustainable Offtake

Although genetic change caused by selective harvesting can be subtle and hard to detect, two main effects can be predicted: (1) disproportionate reduction or elimination of locally adapted populations or stocks and a resulting decrease in overall genetic diversity within a species; (2) a directional change in gene frequencies through harvesting that favors some genetic characteristics over others. To the extent that such changes reduce the health and productivity of a population, the long-term sustainability of the offtake is compromised. The following is a review of these effects in timber harvesting, fisheries, and recreational hunting.

Timber Harvesting

F. T. Ledig (1992, p. 90) characterizes adverse genetic change caused by many timber harvest regimes, such as "creaming" and "high grading," as "ranging from selection against the most valuable forms to selection for the poorest." Extensive deforestation has undoubtedly caused major alterations in gene pools of trees, including the extinction of locally adapted populations. Few studies, however, have investigated the effects of more sustainable timber harvest practices on the genetic diversity of harvested species. Such practices may alter mating systems and thus the genetic structure of populations and may alter or directly eliminate locally adapted populations. Tree populations often are genetically adapted to microenvironmental conditions, such as elevation and slope-aspect, which can vary significantly over short distances (Ledig 1992; Millar, Ledig, and Riggs 1990). Thus, poorly planned clear-cuts with natural reseeding from nearby but genetically different stands can alter genetic diversity of the harvested stand.

A possible example of dysgenic, directional selection is found in the New Jersey pine barrens of the eastern United States, where pitch pine (*Pinus rigida*) was logged for three centuries at roughly twenty-year intervals to provide fuelwood. This population of pine now exhibits poor growth form, which may have been caused by a cutting regime that selected against the fastest-growing trees and favored slow-growing trees because the latter would not reach cutting size within each cutting cycle. Further, trees with single, straight boles, which are easier to trim, were probably selectively cut rather than scraggly or limby trees (Ledig 1992).

Another forest management practice that can cause genetic change is the retention of seed trees to reseed surrounding logged areas, resulting in increased inbreeding. Although the long-term effect on adaptive genetic diversity may not be serious, reproduction and viability may be affected for a generation or two. In most species of pine, for example, selfing (an individual tree fertilizing itself) reduces seed yields by 40–90 percent. Inbreeding also exposes recessive alleles, thereby increasing phenotypic variance and setting the stage for rapid evolutionary change (Ledig 1992). Evidence for inbreeding depression is reported for a selectively logged population of the dipterocarp *Shorea megistophylla* in Sri Lanka. D. A. Murawski, I. A. U. Nimal Gunatilleke, and K. S. Bawa (1994) postulated that because of the tenfold decrease in density of mature trees in the logged dipterocarp forest, inbreeding may be caused by pollinators spending more time foraging within crowns of single trees than in moving among crowns.

Many diseases, such as fusiform rust of pines and white-pine blister rust, preferentially attack young trees during forest regeneration. Silvicultural practices that maintain younger age classes, as opposed to mature forest, can thus create disease epidemics, making disease a much stronger selective force (Ledig 1992). For example, at the beginning of the twentieth century when blister rust was first discovered in western white pine (*Pinus monticola*) in the United States, only 1–2 percent of the seedlings were resistant. By 1964, 20 percent of the progeny from stands decimated by the disease were resistant. Harvesting and management had increased the number of young white pines in the population, with the result that blister rust became much more prevalent and thus a stronger selective force (Hoff, McDonald, and Bingham 1976, cited in Ledig 1992).

Fisheries

Genetic change from sustainable offtake in fisheries operates at both the multistock level and within individual stocks. Some stocks in multistock species may be more vulnerable to capture than others, resulting in decline and eventual loss of the more vulnerable stocks. Thus, the first stocks to disappear are those that grow rapidly and are easily caught, precisely the properties most desirable to the fishery (Nelson and Soulé 1987). The effect is similar for both genetically distinct stocks and genetically distinct individuals within a species. This problem may be particularly acute in anadromous fish species, which often have many genetically distinct stocks that correspond to different spawning streams.

Salmon fisheries often present major management problems when multiple stocks with various rates of production or harvest mortalities are indiscriminately harvested in a single fishery (Francis 1997). M. A. Henderson and M. C. Healey (1993, p. 26) raise these concerns regarding a proposal to double the production of sockeye salmon in the Fraser River in Canada: "As the abundance and productivity of the larger stocks is increased, more fishing effort will be required to harvest the fish. This will place the small, less productive stocks at an increasing risk of accidental extinction . . . and greatly increases the risk that potentially unique gene pools associated with local populations will be exterminated." Among the stocks most vulnerable to overfishing are small marginal populations. Because of their exposure to extreme environmental conditions, such populations often possess unique genotypes and adaptive traits of both inherent value and potentially great value to the long-term adaptive evolution of the species (Scudder 1989). Unless the stocks are easily distinguished by harvesters, such losses may go unnoticed. Regulations that permit fishing offshore, where individual stocks are mixed, rather than from individual streams, where the take of individual stocks can be controlled, may exacerbate this problem (Nelson and Soulé 1987).

A growing body of theoretical (Brown and Parman 1993; Policansky 1993) and empirical work (Rowell 1993) indicates that selective harvesting may lead to significant directional genetic changes in fish stocks. D. Policansky (1993, p. 2) notes that "Fishing mortality is often very high and nonrandom with respect to several life-history traits that are at least partly heritable. Therefore, it seems likely that fishing causes evolution in fishes." The most commonly cited mechanism is gear selectivity such as the mesh size of nets, which often means the selective removal of larger fish from the stock. In general, populations subject to high adult mortality from fishing should evolve toward early maturity and higher fecundity compared with less intensively fished populations (Policansky 1993). Although cause-and-effect relationships are generally not yet established, changes in age or size at maturity have been documented for several exploited stocks of marine fish. The heavily fished North Sea cod shows a clear trend over the past century toward smaller size and earlier age at maturity (Brown and Parman 1993; Rowell 1993). Genetic change caused by commercial fisheries taking fish of larger than average size may have been the main reason for a decline in size of some or all of five species of salmon caught in Canadian fisheries between 1950 and 1975 (Ricker 1981). A population of lake whitefish (*Coregonus clupeaformis*) subject to gillnetting in Great Slave Lake, Alberta, Canada, exhibited a decline in growth rate apparently caused by selective removal of faster-growing fish that mature at a larger size (Handford, Bell, and Reimchen 1977).

More notable changes in life history patterns also result from selective fishing pressure. In pandalid shrimp, which mature first as males and then change into females, the sex change occurs earlier in heavily fished populations than in unexploited ones (Charnov 1981, cited in Policansky 1993). Some species of salmon display two distinct, genetically determined reproductive strategies among males within the same population. There are “normal” hooknose males, which grow to a large size and rely on fighting to gain access to females, and there are “jacks,” males that mature early and at a small size and rely on sneaky approaches to females. Selective fishing for larger salmon would favor jacks, and indeed the proportion of sockeye salmon jacks has increased in the heavily fished Columbia River in the northwestern United States (Ricker 1972). Genetic change appears to be the best explanation for the altered reproductive strategies in both the pandalid shrimp and the salmon.

Two other common practices in sustainable fisheries that may lead to directional genetic change are the imposition of minimum size limits and intensive fishing for short periods of the year. Minimum size limits that are around the age or size of first breeding may cause genetic changes in the age or size at which individuals in the population breed. Short but intensive fishing periods that target spawning fish or fish migrating to spawning areas may artificially select for an earlier or later breeding season (Nelson and Soulé 1987).

Recreational Hunting

The selective harvest of particular cohorts from a population through recreational hunting can skew sex and age ratios and reduce genetic variability (Teer 1997). Large horns and antlers are preferred among trophy hunters, and hunting regulations often set minimum size limits on horns and antlers. Trophy hunting of moose in Prussia over centuries is reported to have resulted in unpalmated antlers by World War I. Eugenic culling, through the removal of moose with atypical and unpalmated antlers, restored moose with large, palmated antlers to the population (Kramer 1963, cited in Teer 1997). Most jurisdictions in which bighorn sheep (*Ovis canadensis*) are hunted in western North America permit only rams, often with horns that have a minimum of a three-fourths curl, to be killed (Jorgenson, Festa-Bianchet, and Wishart 1993). N. N. Fitzsimmons, S. W. Buskirk, and M. H. Smith (1995) found that bighorn sheep rams with large horns were usually more heterozygous than were rams with small horns. Because large-horned rams are more successful at mating than are small-horned rams, and because the heterozygosity of large-horned rams is important in maintaining genetic variability in the population, Fitzsimmons and coworkers hypothesize that selective killing of large-horned rams may reduce the genetic variability and the viability of populations. Similarly, simulation studies indicate that hunting can reduce genetic variability in moose and white-tailed deer (*Odocoileus virginianus*) (Ryman et al. 1981). The correlation of overall genetic variability with number of antler points in white-tailed deer in the southeastern United States (Scribner, Smith, and Johns 1989) raises the possibility that trophy hunting in deer may reduce population fitness.

Although the offtake in this instance was not recreational or sustainable, selective harvesting of tusked female African elephants suggests how heritable traits may be affected by trophy hunting. Due to illegal hunting of elephants for ivory in South Luangwa National Park and the Lupande Game Management Area in eastern Zambia, the number of tuskless females increased from 10.5 percent in 1969 to 38.2 percent in 1989. This change was apparently a direct effect of selective killing of animals with tusks to supply the ivory trade. The increase in tusklessness stopped when poaching dramatically declined beginning in 1988 because of a new law enforcement program. A rough approximation based on more limited data is that the proportion of tuskless males in the same population increased from 1 percent in the early 1970s to 10 percent in 1993. Although the authors do not differentiate what part of the decline in tusked individuals was due to their direct removal from the population and what part was due to a decrease in births of tusked females because of genetic changes, they suggest that the latter mechanism was the cause (Jachmann, Berry, and Imae 1995).

Where recreational hunting exerts a less skewed preference for certain traits, genetic effects are probably less important. For instance, although considerable concern has been expressed about the genetic effects of waterfowl hunting, research to date indicates no significant alteration or erosion of genetic diversity within species (Callaghan, Kirby, and Hughes 1997).

Ecosystem Effects of Sustainable Offtake

Significant sustainable removal of any species that is an important producer, consumer, decomposer, or determinant of structure in an ecosystem—a keystone species—has ramifications for other species in the ecosystem and for ecosystem functions. Most comparative research on the biodiversity effects of harvest regimes has been conducted in forestry, particularly temperate forestry. Given the diversity of silvicultural interventions and the fact that logging itself is often a silvicultural tool, it is difficult, if not impractical, to isolate the effects of sustainable offtake of trees from the effects of silvicultural interventions. In addition, it is often difficult to distinguish the effects of overexploitation of forests from the effects of sustainable harvest. For example, much has been written on the effects of clear-cuts and forest fragmentation on biodiversity (e.g., Frumhoff 1995; Noss and Cooperrider 1994), but clear-cuts are employed and fragmentation occurs in both unsustainable and sustainable logging practices. Further, this assessment must be largely speculative regarding tropical forests, given that only a handful of tropical forest management operations are considered sustainable. Thus, one must make inferences from the biodiversity effects of unsustainable practices.

Other uses of wild species—harvest of nontimber plant products from forests, fisheries, recreational hunting of large herbivores—have been even less researched regarding the ecosystem effects of sustainable offtake. Marine fisheries are like tropical forestry in that so few fisheries have a history of sustainable offtake that there are limited possibilities for examining the effects of sustainable offtake on biodiversity and ecosystem functions. Whereas the biodiversity effects of overfishing have been well reviewed (Dayton et al. 1995), the biodiversity effects of sustainable harvesting, even in some extensively researched fisheries, are largely unknown (e.g., Francis 1997; Upton 1997). Further, P. K. Dayton and colleagues (1995, p. 206) lament that "*all* (italics in original) efforts to evaluate bycatch and environmental effects of heavy fishing on natural systems are too late because most sensitive species have long been impacted, leaving no concept of natural relationships or patterns."

Timber Harvesting

Since its initial development in Germany and Scandinavia in the early 1900s, the sustained-yield model of forestry, in which timber production is the primary goal, has led to ever-increasing intensities of harvest and silvicultural interventions (Dudley, Jeanrenaud, and Sullivan 1995). Recent research in management of both temperate and tropical forests shows that species diversity may respond negatively, positively, or with no discernible effect to generally sustainable offtake (Frumhoff 1995; Halpern and Spies 1995; Hansen 1997; Hansen et al. 1995; Kiernan and Freese 1997; Roberts and Gilliam 1995; Salick, Mejia, and Anderson 1995). How effects on species diversity are judged depends in large part on the temporal and geographic scale considered as well as on the measure of species diversity used. For example, a clear-cut may increase species diversity at the local level because of overlap of forest and open-area species but may decrease species diversity at the landscape level because of the disappearance of species that are intolerant of disturbed forest. Further, though the number of species may be unaffected or may even increase after logging, the relative abundance of species may change substantially (DellaSala, Olsen, and Crane 1995).

The effects of sustainable offtake, as distinguished from intensive silvicultural interventions, revolve around similar issues in the management of both tropical and temperate forests. Thus, where commercially valuable species occur in high densities, sustainable levels of selective cutting will affect more of the forest and thus, one would predict, have greater effects on biodiversity than will selective logging where commercially valuable species occur at lower densities. J. Palmer and T. J. Synott (1992, p. 352) caution that "Maximum biological diversity may be conserved by very light and occasional harvests of genetic material, but attempts to maximize sustained yields of sawlogs may reduce the genetic resources characteristic of mature-phase primary forest."

Low-intensity selective cutting of individual trees at frequent intervals (e.g., twenty to thirty years), broadly known as the "polycyclic system" (Whitmore 1991) or "natural forest management" (Putz 1993), is more common in tropical forests than in boreal or temperate forests because the former generally have lower densities of commercially important trees. A more intensive version of this, developed in Southeast Asia, is the monocyclic system (commonly known as the Malayan Uniform System), in which all marketable trees are extracted in a single felling cycle, with consequently longer felling intervals (e.g., seventy years). Attempts at artificial regeneration have largely failed, and thus these systems now rely primarily on natural regeneration to ensure that native species of commercial importance colonize the gaps created by logging (Putz 1993; Whitmore 1991). Polycyclic and monocyclic systems in Southeast Asia involve more intensive logging than generally occurs in Africa and Latin America because commercially valuable species (the dipterocarps in Asia) occur at much higher densities in Southeast Asia than in the other two regions (Whitmore 1991).

The argument often made regarding the ecological sustainability of these various systems is that selective felling mimics natural tree falls (Dykstra and Heinrich 1992). F. E. Putz (1993) suggests that natural forest management systems are unlikely to have a substantial effect on biodiversity. According to R. J. Buschbacher (1990), if performed carefully, selective logging of tree genera such as *Swietenia* in Latin America, *Khaya* in Africa, and *Shorea* in Asia can have minimal effects on ecosystem structure and function and on biodiversity. The strip–clear-cut system currently being tested in Amazonian Peru (Hartshorn 1995) is also designed to mimic natural forest disturbance, such as tree-fall gaps.

P. C. Frumhoff (1995) reviewed the effects on animal diversity of logging in tropical forests in Africa, Southeast Asia, and Latin America. Although the sites he reviewed varied in sustainability of offtake, they indicate how biodiversity is affected under sustainable harvest regimes. In general, selectively logged sites had equal or greater numbers of bird and mammal species compared with unlogged sites. However, species composition changed, with an increase in species that inhabit disturbed habitats and a decline in those that require largely undisturbed forest habitat. For example, in French Guiana, 42 percent of the bird species found in a primary forest declined in number or disappeared eight to twelve years after a single logging of three trees per hectare, whereas species associated with edge and secondary growth greatly increased in diversity (Thiollay 1992). Logging of species that provide food for wildlife is an obvious mechanism by which selective logging may affect biodiversity. Logged areas in Southeast Asia showed marked declines in populations of animal species that feed on the fruits and seeds of the commercially valuable dipterocarp trees (Cannon et al. 1994; Johns 1992).

Species that inhabit forests subject to frequent and severe natural disturbances may be less affected by logging than species found in more stable forest habitats because the former species are preadapted to large-scale disturbance (Frumhoff 1995). One of the few relatively sustainable tropical forest management sites where biodiversity effects have been rigorously addressed is the Plan Piloto Forestal on Mexico's Yucatán Peninsula, where forests are frequently damaged by hurricanes. Selective logging here focuses on mahogany, Spanish cedar *(Cedrela odorata)*, and a few lesser-known species (Kiernan and Freese 1997). J. F. Lynch and D. F. Whigham (1995) found no difference in the numbers or species composition of migratory birds (which seasonally constitute a large proportion of the bird community) between selectively logged sites and unlogged sites; nor was any difference apparent in the composition of resident species. At the same site, however, M. Dickinson (1993) found more pioneer plant species regenerating in logging gaps than in natural forest gaps, though how this affects diversity at later stages of succession in the forest gap is unknown. In nearby Belize, A. A. Whitman, J. M. Hagan III, and N. V. L. Brokaw (1994) also reported no significant effects on the bird community in a forest being selectively logged for mahogany and Spanish cedar. P. C. Frumhoff concluded from his review that expansion of current forms of selective commercial logging in tropical forests will result in expansion of ranges of those species adapted to disturbed sites (as in the hurricane forests of the Yucatán Peninsula) and reduction of ranges of many primary forest specialists, with the overall effect being a regional loss of biodiversity.

Among temperate forests, selective removal of trees in commercial forestry is more common in hardwood forests than in coniferous forests because the former tend to be less homogeneous in commercially valuable species. Selective logging in a primary forest in the southern Appalachian Mountains of the United States reduced the diversity of vernal herbs (herbaceous forest floor species that appear in spring before canopy closure), though less so than did clear-cutting (Meier, Bratton, and Duffy 1995). Where clear-cutting is practiced, as in many managed coniferous forests of the world, the range of effects on forest diversity is much greater. Compared with selective cutting, clear-cutting represents a much more novel disturbance and, when combined with suppression of natural disturbances such as fires, commonly reduces both structural complexity and species diversity of the forest (DellaSala et al. 1995; Hansen 1997; McShane and McShane-Caluzi 1997). However, where the scale of clear-cutting is small relative to the total area of forest, diversity of some organisms may increase because of the increased diversity of habitats created. For example, in a hardwood forest of the northeastern United States, C. J. E. Welsh and W. M. Healy (1993) compared bird diversity in reserved forests in which no cutting occurred with that in managed forests in which clear-cuts were no larger than fifteen hectares and less than 20 percent of the forest was under thirty years of age. Bird diversity was more than 50 percent greater in the managed forests than in the reserve forests, and all species found in the reserves were also present in the managed forests.

Increasing intensity of harvest (and of silvicultural interventions, discussed later), however, eventually begins to erode a forest's structural complexity, which in turn leads to decreased species diversity. Thus, increasing harvest levels elevate the need to maintain mature and old-growth stands within a managed forest because such stands generally provide great structural complexity and harbor species not found in earlier successional stages (DellaSala et al. 1995; DellaSala, Olsen, and Crane 1995; Franklin 1995; Hansen 1997; McShane and McShane-Caluzi 1997; Payne and Bryant 1994). The size, shape, and placement of these stands are crucial. For example, a small patch of mature forest surrounded by clear-cuts is strongly influenced by edge effects; the microclimate of such a forest patch may be modified more than 200 meters from the forest's edge. Further, harvest methods that retain green trees, snags, and downed logs are important in maintaining structural complexity and biodiversity (Franklin 1995). Although these harvest methods and the maintenance of mature and old-growth stands are clearly crucial for biodiversity, such practices may be uneconomical from a commodity production perspective because they may result in lower wood production and revenues (Hansen et al. 1995).

Harvest of Nontimber Plant Products

The sustainable harvest of nontimber plant products from forests may have significant consequences for biodiversity, though such effects have been little researched. P. Hall and K. Bawa (1993) warn that harvesting of seeds and fruits decreases food for fruit-eating populations, limiting the number of organisms and perhaps decreasing the diversity of fruit eaters in a given community. These changes in turn may alter other food web relationships and thus negatively affect other species in the community. Even something as simple as removal of dead wood and leaves may devastate decomposer communities (e.g., microbes, fungi) critical for nutrient cycling. R. E. Bodmer and co-workers (1997) aver that the harvesting of palm fruits in Amazonian Peru has probably reduced populations of forest animals that eat them.

Fisheries

The most obvious and direct effects of fish offtake on biodiversity are mediated through the food chain, either because of release of prey through removal of predators or because of suppression of predators through removal of their prey. Benthic marine systems reveal cascading effects through the food chain caused by the removal of predators, but pelagic communities have relatively unstructured food webs and thus display few examples of such cascading effects (Dayton et al. 1995). Although most benthic marine systems have been intensively fished for so long that few if any natural systems remain against which to evaluate change, considerable evidence indicates that the removal of predators, even if sustainable, has community-wide effects. For example, removal of large predators from the northern Atlantic coastal fisheries of North America appears to have released a new suite of benthic predators such as echinoderms and crabs (Witman and Sebbins 1992). In Chile, fishing of predators has resulted in major changes in the intertidal community (Castilla and Duran 1985; Moreno, Sutherland, and Jara 1984), and intensive fishing of southern king crabs (*Lithodes antarctica*) may have released populations of sea urchins (*Loxechinus albus*), one of their major prey species (Dayton 1985). Other examples of cascading effects include the release of krill following overharvesting of southern ocean baleen whales, the apparent increase in squid populations following removal of fish that prey on their eggs and very young in tropical fisheries (Dayton et al. 1995), and the effects on kelp forests following a reduction in sea otter populations (Simenstad, Estes, and Kenyon 1978).

Just as old-growth stands may be eliminated in sustained-yield forestry operations, large, old fish may be eliminated in the process of maximizing fisheries yields. Groupers and basses weighing hundreds of kilograms were once common in coral reef systems, but such individuals have been largely eliminated by fishing (Dayton et al. 1995). Compared with these "old-growth" predators, current populations of smaller groupers and bass must have significantly different effects on herbivorous reef fish, with cascading effects throughout the coral reef system. The depression of predator populations by the harvest of prey is less well documented. However, S. Nicol and W. de la Mare (1993) suggest that the harvest of Antarctic krill may have to be considerably lower than the potential maximum sustainable harvest to ensure that populations of krill predators are maintained.

The effects of sustainable offtake in fisheries may operate via less direct food web mechanisms. For example, although salmon are a keystone food resource for many vertebrate predators and scavengers, perhaps more significantly, "salmon may themselves be an important 'glue' that holds their ecosystem together in that they serve as a nutrient pump from the marine to the freshwater parts of the system" (Francis 1997, p. 634). Thus, the sustainable harvest of salmon may affect riparian ecosystems through various mechanisms.

Hunting of Large Herbivores

Apart from being important prey for large predators, large herbivores such as ungulates modify ecosystems by creating spatial heterogeneity, influencing succession, and controlling the switching of ecosystems between alternative states (Hobbs 1996). Significant offtake from ungulate populations, whether by recreational hunting or culling, is therefore likely to have significant effects on biodiversity. A major factor in understanding the effects of large herbivore management on grassland and forest ecosystems is that intermediate intensities of grazing or browsing often result in greater biodiversity and productivity than do either low or high grazing intensities (Hobbs 1996; McNaughton 1993; West 1996).

Analysis of the biodiversity effects of offtake of large herbivores is complicated by the fact that in many ecosystems, populations of large predators have been reduced or eliminated, thus eliminating a major factor in the natural regulation of ungulate numbers. Further, seasonal movements of many ungulate populations have been inhibited or precluded by habitat alteration and by human-constructed barriers, with potentially important consequences for vegetative communities. J. G. Teer (1997) reviews evidence indicating that because of such factors, populations of large mammals such as white-tailed deer, moose, elk, and African elephants have expanded beyond natural levels, with negative consequences for the diversity of plants, birds, and small mammals. Thus, elephants and African buffalo (Cape buffalo) (*Syncerus caffer*) have been routinely culled in Kruger National Park, South Africa, to protect habitats (Joubert 1991). To the extent that culling and recreational hunting reduce populations of large herbivores in these ecosystems, the result may be an increase in species diversity. However, recreational hunting is often an ineffective means of controlling ungulate numbers (Teer 1997), and thus in many cases the ecosystem effects of such offtake are probably minor.

Biodiversity Effects of Incidental Harvest and Disturbance

Harvest techniques, regardless of whether the offtake is sustainable or not, tend to be sloppy in many forms of CCU, with significant effects on other species and the associated ecosystem. Soil and nearby vegetation are damaged during logging, and nontarget species are taken in fishing and recreational hunting. Such incidental disturbance and harvest are additional constraints that natural ecosystems impose on commercial harvesting of wild species. Wild species tend not to grow in straight-rowed monocultures for ease of harvest, and other species get in the way. These incidental effects are often some of the most difficult to manage; their improvement depends largely on enhancing the skill and attitude of the harvester and the selectiveness of the harvest technology.

Timber Harvesting

Incidental damage to soils and surrounding vegetation in forestry operations may be direct or indirect. Direct damage is caused by felled trees striking or pulling down nearby trees and other plants, a common problem in selective tree harvesting in the tropics. For example, in a lowland forest in Brazil, logging of less than 2 percent of the trees with diameter at breast height greater than ten centimeters resulted in 26 percent of trees of equivalent size being destroyed or seriously damaged and a 50 percent reduction in canopy. Further, logging roads scarred 8 percent of the forest floor cover (Uhl and Vieira 1989). Indirect effects occur when felled trees expose adjacent uncut forests to microclimatic changes. Exposure of the forest understory to the effects of direct sunlight and wind will lower humidity, create larger daily swings in temperature, and increase the frequency of forest fires, with modest to far-reaching effects on biodiversity (Frumhoff 1995; Mader 1979, 1981; Uhl and Buschbacher 1985). Forest edges may also be "ecological traps" because birds concentrate their nests in edges, where they are exposed to increased predation, and nest parasitism is often more common near forest edges (DellaSala, Olsen, and Crane 1995; Gates and Gysel 1978; Noss and Cooperrider 1994).

The negative effects of logging roads are not limited to edge effects. Throughout the tropics, logging roads have provided easy access for market hunters to new forest areas and game populations, often with devastating effects on game populations (Frumhoff 1995; Putz 1993). Even where hunting is not a factor, some species, such as the grizzly bear (*Ursus arctos*), avoid areas near logging roads, a factor that can lead to reduced bear populations (Paquet and Hackman 1995). Roads and logging trucks also facilitate the invasion of exotic plants and new pathogens into forest ecosystems, sometimes with major effects on biodiversity (Noss and Cooperrider 1994).

Fisheries

The incidental harvest, or bycatch, of nontarget species is massive in some marine fisheries, with an estimated 27 million metric tons of marine species discarded annually during 1988–1990 (table 6-2). Because discard figures do not include nontarget species that are landed, retained, and reported, the bycatch total (all nontarget species caught) is even larger. Most discards die from the physiological stress of being brought to the surface. Shrimp fisheries are the most problematic, with 5.2 metric tons of discard for every 1 metric ton of shrimp caught, accounting for 35 percent of the global marine fisheries discards. The scale and complexity of the problem become even more apparent when discards are examined in more detail. For example, the annual discard of 30,000 metric tons by shrimp trawlers in the northern Australian prawn fishery includes more than 240 species, including seventy-five families of fish, eleven of sharks, and several of crustaceans and mollusks (cited in Alverson et al. 1994). Shrimp fisheries in the Gulf of Mexico discarded an estimated 5 billion croakers (*Micropogonias undulatus*), 19 million red snappers, and 3 million mackerels

(*Scomberomorus* spp.) in 1989 (Murray, Bahen, and Rulifson 1992, cited in Alverson et al. 1994). Discards from bottom fisheries in the Bering Sea and the Gulf of Alaska total nearly 1 billion animals annually; this figure does not include discards from inshore salmon and herring fisheries and offshore crab fisheries (Alverson et al. 1994).

Although these numbers seem high, their ecological effects remain

Table 6-2. *Worldwide Annual Discards of Bycatch from Marine Fisheries (Based on 1988–1990 Harvest Levels)*

Species Group[a]	Mean Discard Weight (MT)	Landed Catch Weight (MT)	Ratio of Discard to Landed Weight
Shrimps, prawns	9,511,973	1,827,568	5.20
Redfishes, basses, congers	3,631,057	5,739,743	0.63
Herrings, sardines, anchovies	2,789,201	23,792,608	0.12
Crabs	2,777,848	1,117,061	2.49
Jacks, mullets, sauries	2,607,748	9,349,055	0.28
Cods, hakes, haddocks	2,539,068	12,808,658	0.20
Miscellaneous marine fishes	992,356	9,923,560	0.10
Flounders, halibuts, soles	946,436	1,257,858	0.75
Tunas, bonitos, billfishes	739,580	4,177,653	0.18
Squids, cuttlefishes, octopuses	191,801	2,073,523	0.09
Lobsters, spiny rock lobsters	113,216	205,851	0.55
Mackerels, snooks, cutlassfishes	102,377	3,722,818	0.03
Salmons, trouts, smelt	38,323	766,462	0.05
Shads	22,755	227,549	0.10
Eels	8,359	9,975	0.84
Total	27,012,098	76,999,942	0.35

Source: Alverson et al. 1994.

[a] Based on the International Standard Statistical Classification of Aquatic Animals and Plants of the Food and Agriculture Organization of the United Nations.

uncertain. For example, more than 300 million pollocks (*Theragra chalcogramma*) were discarded in the Bering Sea fishery in 1992, but this number represented only 1.6 percent of the estimated harvestable number. Because pollock have a high reproductive rate, the effects of such removal rates may be minor compared with the effects bycatch may have on slow-reproducing species such as sharks, marine turtles, skates, and marine mammals. Bycatch and discards can alter species assemblages, with potential changes in predator-prey relationships, increased food for scavengers, and modification of the structure and function of benthic communities as a result of oxygen depletion caused by decomposition of discards. Declines in stocks of nontarget species in which bycatch is suspected as the cause have been reported, but effects on individual species are difficult to determine (Alverson et al. 1994). The decline of the once-abundant common skate (*Raja batis*) in the Irish Sea appears attributable to its incidental catch in groundfish fisheries (Brander 1981, cited in Alverson et al. 1994). Discards also provide more abundant food for many species, such as birds, sharks, dolphins, and other marine mammals that commonly scavenge discards from fisheries. Thus, populations of some species increase when fishing and resulting discards increase (Alverson et al. 1994).

Bottom-fishing equipment can greatly alter benthic habitats and disturb benthic species. Trawl ground gears can penetrate as far as six centimeters and otter boards as far as thirty centimeters into bottom sediments (Arntz and Weber 1970; Caddy and Iles 1972; Krost et al. 1990, all cited in Alverson et al. 1994). Every square meter of the Dutch continental shelf is dragged by commercial beam trawls, which penetrate to a depth of 4–8 centimeters, an average of once or twice per year. As a result, the ecosystem has changed "from a diverse system to one where only fast-growing, good reproducing, smaller organisms can survive. The large, slow-growing bivalves are disappearing and being replaced by worms" (Lindeboom 1995, p. 595). Trawls used in a new scallop fishery in the Bass Strait of Australia were suspected of crushing or damaging four to five times as many scallops as were caught, and surviving scallops were decimated by infection caused by decomposing remains (McLoughlin et al. 1991, cited in Alverson et al. 1994).

Global figures for bycatch in recreational fisheries are unavailable, though regional analyses indicate it may also be substantial. In 1989, the discard from recreational fisheries in the United States totaled 1.035 billion fish, whereas the landed catch was 651.8 million fish, for a ratio of discards to retained fish of 1.5 (Alverson et al. 1994).

D. L. Alverson and coworkers (1994, p. ix) conclude that "Quick solutions to the bycatch problems are unlikely."

Recreational Hunting

Of the various types of recreational hunting, waterfowl hunting is among the most problematic in terms of incidental mortality and disturbance. With several species of waterfowl often occupying the same area and hunters trying to identify species on the wing, it is not surprising that the killing of nontarget species in waterfowl hunting approaches that in blind fishing with nets (table 6-3). Correct identification of the quarry is probably easier in most other forms of recreational hunting, either because the quarry are larger and slower (e.g.,

Table 6-3. *Hunting-Induced Mortality and Injury of Protected Waterfowl*

Species	Location	Dates	Results
Freckled duck	Australia	1979–1982	Estimated 13–25% of population was incidentally killed[a]
Canvasback	United States	1991–1992	27% of birds encountered within shooting range of hunters were shot at; 20% of shots resulted in killed or crippled birds[b]
Canvasback	United States	1975–1976	26% of live birds had lead shot in tissue[c]
Barnacle goose	United Kingdom	Not given	21% of live birds had lead shot in tissue[a]
Bewick's swan	England	1970–1973, 1989–1992, 1995	34% of live birds had lead shot in tissue[a]
Whooper swan	Scotland	1988–1989	12% of live birds had lead shot in tissue; 10.5% of those found dead had died from gunshots[a]
Trumpeter swan	North America	1976–1987	15% of live birds had lead shot in tissue; 12% of those found dead had died from gunshots[a]

[a] Callaghan, Kirby, and Hughes 1997 and references therein.
[b] Korschgen et al. 1996.
[c] Perry and Geissler 1980.

ungulates) or because there are fewer similar-looking species with which the target species can be confused within a given area (e.g., ungulates, upland game birds). Although the hunter's inability to distinguish species may often be at fault in waterfowl hunting, the high incidence of shot found in protected species of swans in areas where no similar-looking huntable species exists suggests that many hunters knowingly shoot protected species. The problem of selectivity in waterfowl hunting is corroborated by recent findings that despite dramatic species-specific changes in waterfowl hunting regulations in the United States, changes in species-specific harvest rates were largely undetectable (Johnson and Moore 1996). The implication is that there is a severe limitation in our ability to reduce harvest rates of waterfowl species that are vulnerable or in decline by reducing quotas for them.

Incidental mortality and potentially broader ecological effects result from two other aspects of recreational hunting of waterfowl—the use of lead shot and disturbance of birds due to shooting. Lead poisoning, caused by birds ingesting lead shot from the bottom sediments of wetlands, is a significant cause of mortality in many waterbirds. In addition, a large proportion of waterfowl carry lead shot in their tissue as a result of being shot by hunters using lead shot. This represents a source of secondary poisoning for predators and scavengers of waterfowl, particularly raptors. Other animal species in aquatic ecosystems may also be affected, though this question has been little researched. As of 1997, of the twenty-seven countries in which lead shot poisoning in waterfowl has been recorded, only five had instituted nationwide bans on the use of lead shot, and thirteen had taken no action to restrict its use (Callaghan, Kirby, and Hughes 1997).

Little research has been conducted on the effects of shooting disturbance on the movement, distribution, and health of game and nongame animals or how changes in these factors affect ecosystems. D. A. Callaghan and colleagues' review of this question in waterfowl hunting found that both quarry and nonquarry birds (e.g., some species of waders) may be as much as ten times less dense in areas subject to recreational shooting, and some species may entirely bypass major staging areas because of shooting disturbance (e.g., see Figure 5-5). Both the increased movement and the displacement of birds to less than optimal feeding areas may affect survival. Hunting-caused disruption of feeding patterns in snow geese (*Anser caerulescens*) probably results in greater mortality than does offtake from hunting (Frederick, Clark, and Kaas 1987). Hunting disturbance also causes disintegration of family groups in snow geese, which results in decreased survival of young (Prevett and MacInnes 1980).

The massive redistribution of waterbirds, and thus of both their feeding and defecation patterns, caused by shooting may have effects on ecosystem function. Research in Switzerland found higher levels of primary production and invertebrate abundance in hunted than in unhunted areas because of the lower concentrations of waterfowl in the hunted areas (Reichholf 1973).

The effects of disturbance on waterbirds can be ameliorated by banning hunting not only during nesting and fledging periods but also during periods of greatest physiological stress, such as the middle of winter. Many European countries, however, still allow hunting well into or entirely through the middle of winter (Callaghan, Kirby, and Hughes 1997).

Livestock Grazing of Grasslands

Livestock grazing affects the species composition of plant communities in essentially two ways: (1) by the livestock's preferential eating of certain species and (2) by the plant species' differential vulnerability to grazing (Fleischner 1994). N. E. West (1996, p. 336) notes that "Even light livestock use puts inordinate pressure on a few highly palatable species of plants," particularly perennial herbs, resulting in a shift in the competitive balance toward shrubs. Thus, where the goal is to maintain grassland biodiversity, livestock represent a coarse-grained harvesting method with high levels of incidental offtake of nontarget species. Even though some grazing systems may be sustainable in terms of offtake of selected grass species, they often lead to significant changes in species composition and ecosystem dynamics (Solbrig 1993; Walker 1993).

Management Interventions to Increase Productivity or Enhance Quality

Given the opportunity to increase productivity or enhance quality, resource owners are seldom satisfied with harvesting what a habitat naturally produces. Rather, management interventions are usually employed to increase production and profitability (table 6-4). Most interventions fall within one or more of five broad, interrelated categories: (1) enrichment of the habitat, including stocking or planting of the commercially valuable native species; (2) manipulation of abiotic factors and resources to enhance production, harvestability, and ownership of the commodity species; (3) manipulation of biotic factors, particularly control or elimination of species that have a competitive, parasitic, or predatory relationship with the commodity species, and enhancement of food supplies; (4) direct or indirect manipulation of the genetic makeup of the commodity species; and (5) introduction of exotic species or stocks. Management methods are not always readily split into these separate categories. For example, the opening of a forest canopy changes both biotic factors, by reducing the population of the species that are cut, and abiotic factors, by exposing the forest floor and understory to more sunlight and wind.

Table 6-4. *Management Interventions Aimed at Specialization in Production of Wild Species Commodities*

ENRICHMENT OF HABITAT
- Reseeding
- Stocking with young

MANIPULATION OF ABIOTIC FACTORS
- Physical alteration of habitat (control of water levels in wetlands, scarring of forest floor, creation of gaps in canopy, provision of nest boxes)
- Addition of nutrients to system
- Control of frequency and intensity of fire
- Construction of barriers to movement of, or enclosure of, population

MANIPULATION OF BIOTIC FACTORS
- Control of predators
- Control of competitors
- Control of parasites
- Control of organisms that affect the habitat of the target species
- Preferential harvesting of males in polygynous and promiscuous species
- Preferential retention of reproductive stock of the target species during harvesting
- Increased production of species that are a resource (e.g., food, shelter) for the target species
- Management of plant cover and forage production through "strategic grazing" by livestock

MANIPULATION OF GENETIC FACTORS
- Favoring of specific traits through selective harvesting and retention of reproductive stock with desired characteristics
- Introduction of specific traits obtained from selected wild populations
- Introduction of specific traits by genetic engineering

INTRODUCTION OF EXOTIC SPECIES OR STOCKS
- Introduction for harvesting
- Introduction as a management tool (as food or for control of competitors or parasites) to increase production of native species

If modest in scale and intensity, some interventions may have no noticeable effect on the natural ecosystem, and species richness at the local level may even be increased. As interventions intensify, however, changes in the biotic community and in ecosystem structure and processes become increasingly noticeable, often with impoverishing effects on native biodiversity. At the most general level, the ecosystem becomes obviously less natural and more artificial. Ecosystem simplification is usually the end result—and goal—of intensive management for CCU. Following is a review of management interventions used in five of the most common arenas of CCU—timber production, harvest of nontimber plant products from forests, fisheries, recreational hunting of waterfowl, and recreational hunting of large herbivores (mostly ungulates).

Forest Management for Timber Production

Among the various types of wild species use, silviculture probably employs the greatest spectrum of management interventions (table 6-5). Interventions include manipulating the genetics of individual species, manipulating the diversity and abundance of tree species, simplifying forest structure at the stand level, changing forest patterns at the landscape level, and reducing the rotation period (Franklin et al. 1989). The combined effect of these interventions is ecosystem simplification through reduced genetic, species, and structural diversity. Such interventions have been applied for decades in temperate forests, and substantial work is now under way regarding how to modify silvicultural practices to mitigate their effects on biodiversity (Hansen 1997; Hunter 1990; Sjöberg and Lennartsson 1995). In contrast, silvicultural interventions in native forests, apart from the canopy manipulation that logging itself imposes, have been modest to nonexistent in commercial forestry in the tropics. The only exceptions are the monocyclic and polycyclic logging systems used in Southeast Asia and to a very limited extent in western Africa, where prefelling and postfelling treatments to improve regeneration have been conducted, although these practices have been largely abandoned because of high costs and other problems (Whitmore 1991).

Stocking as a management tool ranges from the planting of a few seeds of native stock in gaps left after selective cutting to systematic planting of young trees of exotic species in plantations following clear-cutting. The former technique may have unnoticeable or positive

Table 6-5. *Methods by Which Forests Are Altered and Simplified to Increase Timber Production and as a Consequence of Harvesting Methods*

GENETIC DIVERSITY WITHIN TREE SPECIES AT A PARTICULAR SITE

• Introduction of nonnative stock, resulting in hybridization with native stock
• Use of nursery-grown seedlings from native stock, including genetically improved or cloned seedlings
• Retention during cutting of seed trees ("standards") that have preferred growth forms
• Selective cutting of preferred forms, resulting in dysgenic effects on population
• Alteration of breeding patterns due to fewer trees in logged stands, often resulting in increased inbreeding and reduced heterozygosity

DIVERSITY AMONG TREE SPECIES AT A PARTICULAR SITE

• Planting of rapidly growing exotic species
• Mechanical and chemical elimination of competing trees and other plant species
• Replacement of mixed-species forests with monocultures

STRUCTURAL CHANGES FROM STAND TO LANDSCAPE LEVEL

• Reduction in range of tree sizes and growth forms
• Geometric spacing of trees
• Clear-cutting and monocyclic cutting systems
• Suppression of natural fires and use of prescribed burns
• Sanitation and salvage cuts that remove dead timber and downed logs
• Practices that reduce diversity of species
• Practices that favor temporal simplification (see below)
• Increased number of discrete patches of forest, but reduced range in patch size and increased, straightened, and sharpened boundaries between patches, due to clear-cuts
• Draining of wetlands
• Construction of logging roads

TEMPORAL SIMPLIFICATION

• Early successional stages shortened by use of fast-growing species, use of fertilizer, elimination of competing species, and other interventions
• Late successional stages eliminated because optimal timing of harvesting occurs in transition from young to mature forest; and thus mature and old-growth forests eliminated

Sources: DellSala, Olsen, and Crane 1995; Dudley, Jeanrenaud, and Sullivan 1995; Franklin et al. 1989; Ledig 1992; Whitmore 1991.

effects on biodiversity, whereas the latter, if practiced on a large scale, as is often the case in temperate and boreal forests, generally greatly diminishes biodiversity compared with that in the native forest (Hansen et al. 1991; McShane and McShane-Caluzi 1997; Payne and Bryant 1994). Although the world's temperate and boreal forest coverage has remained stable in recent decades, this masks often significant changes in composition and structure as natural and artificial regeneration of clear-cut stands results in more even-aged, less diverse forest cover (Dudley, Jeanrenaud, and Sullivan 1995). For example, though forest cover is increasing in Sweden, all but a small proportion of Sweden's forests are intensively managed. This includes replacement of mixed forest with single species of conifers, including an exotic from North America, the lodgepole pine (*Pinus contorta*), with site preparation by scarification, thinning to remove other tree species, and application of nitrogen fertilizers. Natural old-growth forest there was reduced by half during the 1980s, and several indicators show a decline in forest biodiversity (Dudley, Jeanrenaud, and Sullivan 1995). In Switzerland, according to T. O. McShane and E. McShane-Caluzi (1997, p. 144), "The emphasis on wood production in this century led management to focus on a limited number of economically valuable species and cutting at specific ages. The result was a simplified forest structure of even-aged trees with few species. . . . Broadleaved trees, formerly the dominant vegetation type on the Swiss Plateau, have been reduced to less than 40 percent of their natural potential due to extensive planting of conifers."

N. Gopinath and P. Gabriel (1997) review how a polycyclic management system for mangrove trees of high commercial value has altered the ecosystem of the Matang Mangrove Forest Reserve in Malaysia. The deliberate removal of most mangrove species and exclusive replanting with *Rhizophora apiculata* in much of the reserve has simplified the structure of the forest, leaving virtually no areas of old primary forest. According to the authors (p. 198), "The management of the reserve has led to a forest type that is structurally different, being more dense and more uniform. The effect of growing trees of the same age and height has destroyed the multicanopied structure of the primary forest, and with it, presumably, the complex food web and ecological niches that this structure engenders."

Apart from the unintentional, often dysgenic changes discussed earlier, artificial regeneration of stands of native species often involves active management for commercially desirable genetic traits. F. T.

Ledig (1992, p. 103) notes that "Domestication involves conscious, directional selection and is very effective in causing divergence from the wild-type." Where seeds collected from slightly different locations, or seeds from stock that has undergone human-mediated selection for desirable traits, are planted, not only is the genetic makeup of the replanted population different from that of the native species, but also nearby natural stands may be contaminated by pollen and seed migration (Ledig 1992). The forest cover in many European countries now consists mainly of exotic species, and some policies promote such trends. For example, a directive of the European Union now requires that seeds from only certain oaks be planted in order to produce straight trunks, thereby almost ensuring that regenerating forests will consist of nonlocal oak (Dudley, Jeanrenaud, and Sullivan 1995).

According to N. Dudley, J.-P. Jeanrenaud, and F. Sullivan (1995, p. 85) "Throughout the world, there is a tendency for natural, mixed forests to be replaced by monocultures. In addition, native tree species are being replaced by a narrow range of high yielding varieties, consisting mainly of conifers and specialized broadleaved trees such as *Eucalyptus* and *Acacia*." This move toward plantation-type forests has been most extensive in countries in temperate regions, with widespread use of exotics. Meanwhile, plantations, also based on exotic species, are rapidly increasing in extent in many tropical forest regions, particularly in Asia and South America (Dudley, Jeanrenaud, and Sullivan 1995). How plantations will affect global biodiversity depends primarily on two factors: (1) the extent to which native stands are replaced by genetically improved or exotic stands, as opposed to reforestation of deforested or degraded sites; and (2) the degree to which enhanced production from such plantations reduces pressures on native stands.

Recent research and experience in managing North American temperate forests reveal how traditional sustained-yield forest management practices may have to be modified, if not totally reconfigured, to maintain both biodiversity and long-term forest health and productivity. Research by A. J. Hansen and co-workers (1995) in coniferous forests of the Pacific Northwest of the United States, for example, indicates that where the goal is to maintain all native species of birds, silvicultural practices should attempt to maintain a range of canopy tree densities and size-class distributions across the landscape. Even-aged, closed-canopy plantations do not need to be maintained because birds that are abundant in this stand type are also numerous in mature and old-growth forests. Management of temperate forests on a rotation basis, whether in 40- or 100-year cycles, eliminates later mature stands and old-growth forests, which have the greatest structural complexity.

Forest management problems in the northern hardwood-conifer region of North America indicate that both natural processes and larger geographic and time scales may need to be incorporated into silvicultural practices. A reciprocal link between the cycling of carbon and that of nitrogen within the forest ecosystem of this region may impose cyclic patterns of productivity as hardwoods and conifers succeed each other. Managing for only one state—hardwoods or conifers—may therefore be impossible (Mladenoff and Pastor 1993). In the same forest region, silvicultural attempts to develop a spruce monoculture free of spruce budworm infestations, by interfering with the natural cycling between fir and aspen (*Populus* spp.) stand dominance, led to reduced productivity and biodiversity (Baskerville 1988). Such long-term, landscape-level processes led D. J. Mladenoff and J. Pastor (1993, p. 162) to propose that "Rather than managing for a sustained level of a particular target population (i.e., at a local level), . . . managers may consider sustaining the cyclic nature of populations at the ecosystem level while maintaining a sustained yield of a target population at the regional level."

The assumptions buttressing the sustained-yield paradigm of forest management that dominated the twentieth century have become increasingly untenable. Potential forest productivity often is not constant, and the observed state of a forest cannot be maintained indefinitely through intensive management without negative consequences for productivity (Mladenoff and Pastor 1993). Moreover, biodiversity is usually significantly compromised wherever intensive silvicultural interventions are used.

Forest Management for Harvest of Nontimber Plant Products

Management interventions for the production of nontimber plant products from forests, such as mushrooms, berries, fruits, and medicinal plants, are little developed. This is attributable to two major factors: first, many nontimber plant products do not generate sufficient profit to merit the expense of intensified production regimes. Second, in part as a consequence of their low profitability, many nontimber forest plant products are subject to open-access harvesting, and thus there is little or no incentive for an individual to invest in long-term production.

Palms are the single most commercially important nontimber plant group on a global scale. Management to increase the production of native species in more or less natural forests is occasionally reported (Anderson et al. 1995; Kiew 1991; Madulid 1991; Pearce 1991; Pedersen and Balslev 1992), but this is clearly the exception rather than the rule, and overall effects on biodiversity of native forests are probably inconsequential at this stage. Even the commercially valuable rattan has been subjected to little intensive management, in large part because it is still an open-access resource (Johnson 1991). Many palm species are commonly cultivated in garden plots and agroforestry. Some, such as the coconut palm (*Cocos nucifera*) and the African oil palm (*Elaeis guineensis*), have been largely domesticated and introduced throughout the tropics; plantations of oil palms pose the greatest threat in terms of displacing native forests (Johnson 1991; personal observation). Many other commercially important palm species are undergoing various degrees of domestication (Johnson 1991), though what effects this may have on wild stands, whether through eventual displacement of native stands or by genetic pollution, is unknown.

Few studies have examined how management for nontimber plant products may affect biodiversity. A. B. Anderson and colleagues (1995) evaluated how tree species diversity was affected by management by local communities primarily for the production of nontimber plant products, such as the fruits and heart of the palm *Eurterpe oleracea* and the seeds of cacao (*Theobroma cacao*), on islands in the Amazon River of Brazil. Forest management involved three strategies: (1) elimination of undesirable competitors; (2) establishment of plantations and dispersal of propagules of desired species; (3) tolerance of numerous species with current value or potential future value. Diversity of native tree species diversity was reduced by roughly half in managed forests.

S. A. Mori (1992) warns against the establishment of extractive reserves, which generally focus on nontimber plant products, as a biodiversity conservation tool in Amazonia. Brazil-nut gatherers and rubber tappers exert an array of pressures on the forests, with negative effects on plant and animal diversity. Where numerous nontimber plant products are harvested and marketed, however, as in much of southern Asia, local people may practice traditional management methods that result in greater biodiversity than do systems that focus on one or a few nontimber commodities (Poffenberger 1990, 1994; Poffenberger and McGean 1993; Singh et al. 1997).

Fisheries

The most common management interventions in fisheries involve the release of genetically altered or distinct stocks of native species and the introduction of exotic species. The multiple pathways by which these practices affect biodiversity (figure 6-1) are common to all introductions, whether in forestry or in game management.

Releases of hatchery stocks as a tool to augment levels of native populations are particularly common in freshwater fisheries and for anadromous species such as salmon. Genetic problems posed by the release of hatchery stocks stem from two factors: (1) the hatchery stock may not be the same as the stock that inhabits the river into which it is introduced; (2) regardless of the origin of the stock, because of the small effective breeding populations retained by hatchery managers, there is a potential for rapid genetic drift in the hatchery population (Waples and Teel 1990). K. Hindar, N. Ryman, and F. Utter (1991, p. 945) conclude that "Where genetic effects on performance traits have been documented, they always appear to be negative in comparison with the unaffected native populations." Apart from genetic consequences, the release of hatchery stocks often affects native populations through direct competition and the introduction of pathogens.

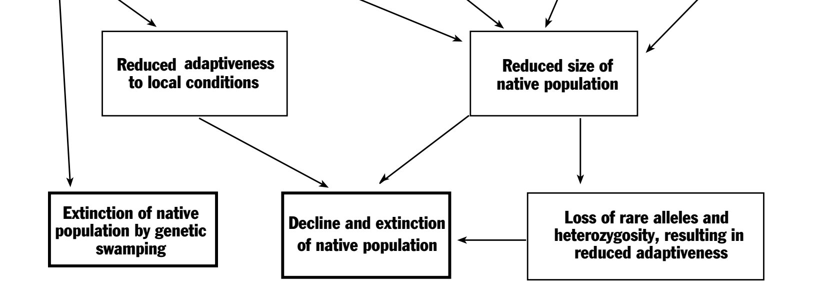

Figure 6-1. *Mechanisms by which introductions of exotic species and stocks lead to decline and extinction of native species.* Source: *Adapted from Carvalho and Hauser 1995.*

The degree to which introductions of hatchery stocks create a problem for native biodiversity also depends, in part, on the status and integrity of existing native stocks. For example, though hatchery stocking programs of the striped bass have altered the composition of all stocks in the United States (Schill and Dorazio 1990), species and ecosystem diversity have benefited from stocking of individual systems where striped bass populations had been lost (Upton 1997).

Because of the high degree of genetic differentiation among salmon stocks in different river systems and even within a river system, hatchery releases can be particularly damaging to genetic diversity in salmon through effects on endemic stocks. In Norway, for example, the artificial reproduction and release into rivers of Atlantic salmon that began around 1850 to enhance recreational fishing has created large declines in native stocks. Although intentional releases may have declined as a problem because of regulations requiring the release of local stock only, a major problem now is the escape of artificially reared salmon from fish farms. Escaped salmon outnumber wild salmon in many Norwegian rivers, and all remaining wild populations are threatened by this trend (Hindar 1992).

R. C. Francis (1997) examines how the salmon-fishing industry develops hatchery programs in an attempt to stabilize and maximize production levels and describes the unpredictable consequences of such programs. In the mid-1970s, for example, catches and runs of pink salmon (*Oncorhynchus gorbuscha*) were at an all-time low throughout the Gulf of Alaska, including Prince William Sound, probably because of a naturally low period of oceanic productivity. A group of fishers and fish processors in Prince William Sound responded by forming a nonprofit hatchery corporation with the objective of creating bountiful harvests even when wild runs were weak. By the mid-1980s, they had created the largest artificial pink salmon run in North America, though the runs exhibited extreme fluctuations in size (figure 6-2). Meanwhile, runs of wild pink salmon declined in the late 1980s and early 1990s despite the fact that harvests of wild populations remained low and this was a period of high oceanic productivity, when wild runs should have remained high. R. C. Francis (1997, p. 650) speculates on the reason for the decline in wild runs: "Hatchery smolts are generally released before wild smolts migrate from their natal streams into the nearshore marine environment. As a result, not only do they swamp the environment due to the recent quantities of releases, but they get a competitive jump on their wild counterparts in the timing of entry." Apart from these direct

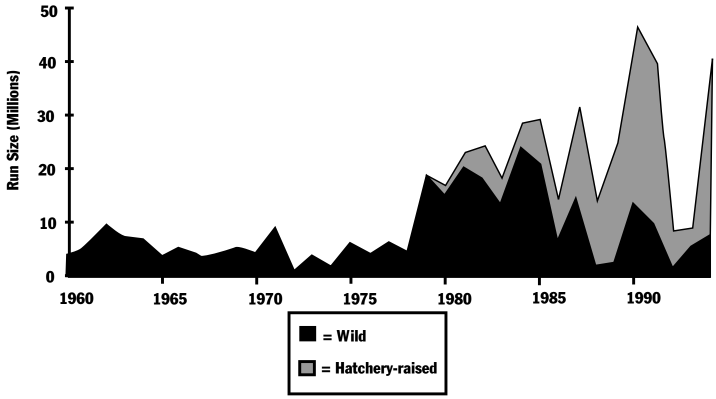

Figure 6-2. *Estimated run sizes of wild and hatchery-raised pink salmon in Prince William Sound, Alaska, 1960–1994.* Source: *D. E. Rogers, pers. comm., 1995, in Francis 1997.*

effects of hatchery salmon on native stocks, hatcheries are at best a palliative that conceals from the public the real problems and dangers that overfishing and development (e.g., dams, deforestation, and degradation of riparian habitats) pose for salmon populations and their habitats (Meffe 1992).

Intensive methods of salmon management are not restricted to hatchery releases and genetic manipulation and to the obvious requirement for devices such as fish ladders to enable migrating fish to get over dams. For example, efforts to rebuild sockeye salmon runs in the Fraser River of British Columbia, Canada, though not involving hatchery releases, may involve constructing spawning channels and increasing the zooplankton food supply for juvenile salmon by fertilizing nursery lakes (Henderson and Healey 1993).

Genetic engineering is another management technology increasingly applied in marine fisheries with potentially far-reaching effects on native populations and ecosystems. Recently, for example, a sterile triploid form of the Pacific oyster (*Crassostrea gigas*), one of the world's most popular culinary oysters, has been created and distributed around the world. Because of its sterility, the triploid form forgoes the normal summer spawning activity (which makes it inedible) and thus is able to direct more energy into growth. Although the triploid form was developed largely for use in fully enclosed systems of oyster farming rather than for wild rearing, the potential for release into the wild and resulting displacement of native stocks, and perhaps for introgression with native stocks if sterility is not complete, is a concern. Other genetic engineering technologies, ranging from hybridization to the insertion of growth hormone genes into oysters, are being tried (Baker 1996).

Oysters are one of several marine animal and plant species that are attracting increasing attention for aquacultural production. Aquaculture is a rapidly growing enterprise, particularly in Southeast Asia, where it grew at a rate of 16 percent annually from 1984 to 1990 (Thuraiŕaja 1994, cited in Clay 1996). Fish culture techniques range from fully self-contained and isolated land-based rearing tanks to various types of intervention in natural systems. Coastal marine waters, because of both their relatively high productivity and their accessibility to human management, are increasingly popular for aquaculture. The rapid rise in world aquacultural production is often cited as a way to decrease the demand for and effects on wild marine food resources, but it comes with potentially significant effects on native biodiversity of coastal waters. Apart from genetic engineering, the inevitable escape of caged exotic fish into the wild, and purposeful introduction of exotic species, aquaculture often involves substantial physical, nutrient, and chemical alteration of aquatic ecosystems. In coastal areas of the Mediterranean Sea, for example, mussels are grown on long lines, oysters on racks or rafts, and fish in submerged and floating cages, with various forms of food and other supplements often provided. The result can be major changes in the biotic community and ecosystem. The structures may attract some species and be avoided by others, and the additional nutrient loads lead to eutrophication of shallow coastal lagoons that are subject to little tidal flushing. In 1990, the 7,500-hectare lagoon Étang de Thau, along the Mediterranean coast of France, had 1,324 hectares devoted to mollusk culture, with 2,816 racks producing 34,000 metric tons of primarily Mediterranean mussels (*Mytilus galloprovincialis*) and the introduced Pacific oyster. Another 25 metric tons of sea bass were produced in cages. Apart from the obvious effects of this scale of management on any semblance of a natural ecosystem, at least nine species of exotic algae, probably introduced incidentally with the oyster, thrive on the oyster lines, and the oyster's introduction may have caused the local disappearance of the Portuguese and European flat oysters (Rosecchi and Charpentier 1995).

Construction of artificial reefs is another physical intervention employed to increase the production of fish in coastal waters. Although this may increase not only the production of marketable fish but also the structural and species diversity of the area (J. R. Clark 1996), it clearly reduces the naturalness of coastal ecosystems.

Even without physical interventions, aquaculture can involve massive translocation of eggs or young. Spat of clams and oysters may be captured in one place and sown elsewhere in high densities, thereby altering the biological community of both the capture and seeding sites. Along the coast of the Matang Mangrove Reserve in Malaysia, for example, spat of the blood clam (*Anadara granosa*) are removed by wire mesh from spatfall grounds and sown on mudflats at the rate of 900–2,160 kilograms per hectare, or roughly 4.5–10.8 million spat per hectare (Ng 1994, cited in Gopinath and Gabriel 1997). In 1993, blood clam farming covered more than 5,040 hectares of mudflats along the western coast of the Malay Peninsula (Gopinath and Gabriel 1997).

The large and rapidly expanding industry of shrimp aquaculture illustrates the range of management interventions, from extensive to intensive systems, with far-reaching effects on tropical coastal ecosystems. Shrimp aquaculture accounted for 25 percent of global shrimp production and 50 percent of international trade by the late 1980s (Clay 1996). Most of the recent growth in aquacultural production has occurred in Southeast Asia and, secondarily, in Latin America. Worldwide, more than 600,000 hectares of shrimp aquaculture are under extensive or semi-intensive management that involves massive physical and biological manipulation of mangrove ecosystems and shrimp populations (Clay 1996).

Freshwater fisheries, more than any other form of wild species management, display the human predilection to introduce exotic species if markets are not fully satisfied with what grows naturally in a habitat. The introduction of exotic species of fish for both recreational and food fisheries is a widespread and ongoing international enterprise with destructive consequences for native species and ecosystems. At least 1,354 introductions of 237 species of fish in 140 countries have occurred since the middle of the nineteenth century, and this does not include introductions of marine species, artificial expansion of ranges within countries, and unofficial introductions (Welcomme 1984, 1988). Introduced species that have the least effect on biodiversity are generally small herbivores, insectivores, and omnivores, whereas the greatest effects are caused by large predators, which are favored in recreational fishing (Courtenay and Moyle 1996). The total number of fish species in some lakes may increase when species are introduced, but native species usually suffer (Courtenay and Moyle 1996) (figure 6-3).

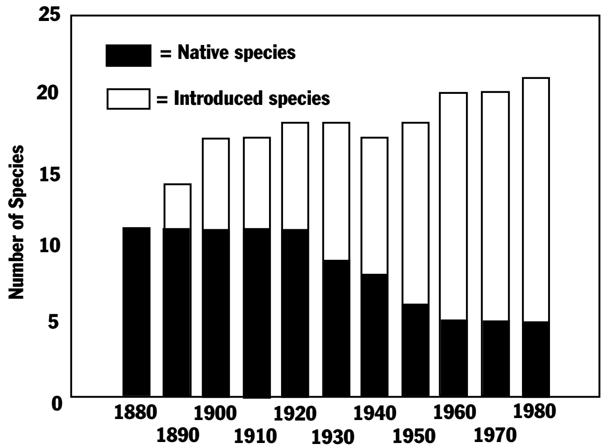

Figure 6-3. *Effects of introduced species of fish on native species in Clear Lake, California, 1880–1980.* Source: *Courtenay and Moyle 1995.*

In North America (Canada, the United States, and Mexico), at least 140 species of freshwater fish have had their ranges expanded through introductions, with an array of negative effects on native species and ecosystems. For example, although introductions of the brown trout (*Salmo trutta*) and rainbow trout (*Salmo gairdneri*) have benefited recreational fisheries, they have displaced the ecologically similar cutthroat trout (*Salmo clarki*) throughout much of the Great Basin of western North America. The pervasiveness of exotic species is suggested by the conservative estimate that 25–50 percent of freshwater fish caught in the continental United States are from introduced populations (Moyle, Li, and Barton 1986).

African lakes have been subjected to at least 147 introductions involving fifty fish species, twenty-three of which came from outside Africa. Eighty-three percent of the introductions were for aquaculture, for creation of a fishery, or for recreational fishing. Although many introductions failed to become established, three introduced species in particular, the Nile perch (*Lates niloticus*), Nile tilapia (*Oreochromis niloticus*), and the freshwater sardine (*Limnothrissa miodon*), appear to have been relatively successful in creating fisheries and increasing per capita human fish consumption in Africa. However, in a number of cases, the benefits in terms of fish yield and socioeconomic return are not sufficient to outweigh the damage done to endemic species that already sustain successful fisheries (Pitcher 1995).

It is often difficult to distinguish between the effects of introductions and of overfishing on biodiversity and ecosystems, though introductions clearly have had significant negative consequences in African lakes (Pitcher and Hart 1995). Roughly two-thirds of the some 300 species of cichlids endemic to Lake Victoria have disappeared since the introduction of the Nile perch in the 1960s, and other major changes in the lake's ecosystem have occurred. Although C. D. N. Barel and colleagues (1991) attribute most of the cichlid decline to predation by the Nile perch, A. W. Kudhongania and D. B. R. Chitamwebwa (1995) cite evidence that overfishing was a more important factor. Regardless of the relative importance of these factors, Lake Victoria—once home to a highly diverse and endemic fish fauna—is now dominated by three fish, two of which are exotic species (Ogutu-Ohwayo 1995).

Waterfowl Management

The long history of waterfowl management in Europe, North America, Australia, and New Zealand should provide a wealth of insights into the biodiversity effects of management methods to improve production. Management interventions for enhancing waterfowl production have focused primarily on manipulation of wetland habitats and on the introduction of nonnative stock and species. Wetland restoration to increase waterfowl production has clearly been of broad benefit to wetland biodiversity, particularly in North America. However, a wide array of management interventions are also carried out on natural wetlands (table 6-6). As in other management systems, judgment of the effects of waterfowl management methods on wetland biodiversity depends in part on the historical effects of human use on the habitat under management (has it already been highly altered?) and in part on the importance of naturalness compared with other management goals. Regardless of these goals, the effects on biodiversity of the interventions listed in table 6-6 remain little researched and poorly understood.

Much wetland management represents efforts to restore degraded or lost wetland habitats. Efforts to encourage farmers to manage rice fields so that they can be used by waterfowl and other wetland species should, in balance, clearly benefit biodiversity at no cost to natural ecosystems. In contrast, though blasting of potholes in a natural wetland with a nearly monotypic cover of vegetation will surely increase both waterfowl habitat and total species diversity, it carries a significant cost to naturalness. Yet other efforts are aimed at creating new wetlands on

Table 6-6. *Management Interventions for Waterfowl Production and Their Effects on Biodiversity*

Intervention	Examples of Biodiversity Effects
Construction of dikes and water control structures to manipulate water levels, including drawdowns, in order to maintain hemimarsh and control fish	Major alterations of plant and animal communities; reduction or extirpation of fish populations; may favor establishment of alien species
Alteration of wetland topography through blasting, dredging, and bulldozing to create potholes, ditches, contour furrows, and resulting spoil ridges in order to enhance mixture of open water and cover for breeding waterfowl	Increased diversity within site; major alteration or loss of natural biotic community
Flooding of bottomland hardwood forests to enhance habitat and increase acorn production	Loss of understory species; change in tree population toward water-tolerant species
Construction of artificial islands for nesting	Significant alteration of natural biotic community
Provision of nest boxes	Effects may be minor, though other birds, mammals, and insects may use nest boxes, with potential changes in biotic community
Farming of grains, such as corn and millet, to increase food	Concentration of birds; shortstopping birds and alteration of migration patterns; outbreaks of density-dependent diseases; displacement of native vegetation by cultivated fields
Use of herbicides, mowing, and controlled burns in wetlands to control vegetation and increase open-water areas	Various changes in plant and animal communities; changes in water chemistry; release of nutrients; increased insolation and resultant earlier warming of soils, and severe damage to bryophytes caused by burning
Use of grazing, mowing, controlled burns, and planting to improve upland vegetation for both wintering and breeding waterfowl	Possible increase in water turbidity, eutrophication, and compaction of soil caused by livestock; significant changes in community structure; if carefully managed, may increase upland plant diversity

INTERVENTION	EXAMPLES OF BIODIVERSITY EFFECTS
Control of fish by drawdowns, application of piscicides, and trawling to reduce water turbidity, competition for food, and predation on ducklings	Alteration of fish community; substantial change in nutrient cycles and structure of invertebrate and macrophyte community
Control of predators of waterfowl eggs and incubating females	Alteration of predator-prey community; other effects largely unknown
Control of muskrats at moderate densities to maintain desirable vegetative cover; control of beavers to prolong beaver pond life	Loss or reduction of large fluctuations in both muskrat numbers and emergent vegetation; reduced periodicity of cycle of beaver pond abandonment and recolonization.
Application of fertilizer to increase productivity of oligotrophic wetlands	Reduced oxygen levels and substantial changes in algal and invertebrate communities

Sources: Callaghan, Kirby, and Hughes 1997; Kadlec and Smith 1992; Payne 1992; Reinecke et al. 1989.

formerly nonwetland sites, and thus an assessment of biodiversity effects must incorporate the loss of the original habitat and its biological community. For example, green-tree reservoirs are created in the United States by annual flooding of bottomland hardwood forests to provide habitat for wintering and migrating waterfowl, but among their more obvious potential consequences are loss of understory species and change in forest composition (Baldassarre and Bolen 1994). Although wetland management for such diverse uses as waterfowl hunting and alligator production is purported to be largely synonymous with management of the wetlands for biodiversity (Joanen et al. 1997; Wentz and Reid 1992), this question has received remarkably little vigorous research. The potential effects, however, on natural wetlands appear significant if strict precautions are not taken in the application of habitat management methods (Callaghan, Kirby, and Hughes 1997).

Because of the high rate of wetland loss around the world, waterfowl management has in large part been a science of doing more with less. How waterfowl managers would like to have their work judged is suggested by comments in a well-known recent waterfowl textbook:

> Whenever the abundance or availability of habitat is reduced, the quality and availability of the remaining habitat gain

> increased significance. This is based on a canon of wildlife management that, within limits, some lesser amount of quality habitat is more important than a greater amount of poorer-quality habitat. Thus, management has evolved as an unequivocal necessity resulting from the direct loss of an immense amount of habitat. And with these losses comes the improbable likelihood that nature can compensate by providing high-quality habitats at the right time and place to meet the life-history requirements of waterfowl. Lacking such compensation, we can scarcely expect to maintain the waterfowl populations witnessed only a few decades ago. (Baldassarre and Bolen 1994, p. 476)

As in forestry and fisheries management, the stocking of domestically reared populations is common in waterfowl management. Genetic contamination of native populations and an increased incidence of disease, caused both by the released birds acting as vectors and by increased concentrations of birds, are potential consequences (Callaghan, Kirby, and Hughes 1997). Establishment of exotic species is also a common method for enhancing waterfowl hunting. The two species most commonly used are the Canada goose (*Branta canadensis*) and the mallard, both native only to the Americas; introduced populations of Canada geese are now found, for example, in Scandinavia, Europe, and New Zealand. Although little research has been conducted on the effects of these introductions, it appears that introduced Canada geese are significantly affecting native waterbird species, with broader implications for ecosystem function (Callaghan, Kirby, and Hughes 1997). Introduction of the mallard into New Zealand has resulted in widespread introgressive hybridization with the Pacific black duck (*Anas superciliosa*) (Gillespie 1985). D. A. Callaghan, J. S. Kirby, and B. Hughes (1997) conclude that introgressive hybridization will probably continue between these two species until Pacific black ducks are either extinct or confined to largely isolated habitats. In North America, both large-scale introductions of mallards and their adaptation to an increase in artificial habitats have caused mallard populations to spread to the east, north, and south. Hybridization with mallards is eroding genetic integrity in the American black duck (*Anas rubripes*) (Merendino, Ankney, and Dennis 1993), has caused the extinction of the pure Mexican duck (*Anas diazi*) genotype in the United States (Callaghan and Green 1993), and threatens the survival of the Florida duck (*Anas fulvigula fulvigula*) (Mazourek and Gray 1994).

Big Game Management

Management of large herbivores for recreational hunting and, in some cases, for production of venison also involves various habitat interventions and manipulation of target species and their predators. Vegetation management for many ungulates is often aimed at opening large, unbroken tracts of woody vegetation or forest to create edge and vegetative diversity. N. F. Payne and F. C. Bryant (1994) provide a long list of management interventions that can be applied to the rangelands of North America to improve habitat for ungulates, though they emphasize the need to maintain overall diversity. Techniques to create edge and openings in shrublands and woodlands range from heavy mechanical manipulation (e.g., plowing, chaining) to application of herbicides. Herbicides, for example, are used to improve elk habitat by top-killing Gambel oak (*Quercus gambelii*) in areas of dense stands. Fertilizer has been recommended as a means to improve forage production on elk winter range (Basile 1970). "Strategic grazing" by livestock has been cited as a tool to improve habitat for mule deer (*Odocoileus hemionus*) (Urness 1990) and elk (Anderson, Franzen, and Melland 1990). Although this type of management can fragment habitats, it is seldom practical on a large scale. How these interventions affect biodiversity is poorly understood.

Management of ungulates for CCU in southern Africa displays a different approach to production specialization. Based on responses to a questionnaire sent to private ranches in South Africa, for example, D. E. Benson (1991, p. 506) reports that landowners who maintain commercial game operations "did not use sophisticated wildlife and habitat management practices." However, management does appear to be intensive and highly specialized in terms of what species are maintained: 37 percent of respondents kept one to four species, 34 percent kept five to nine species, and 29 percent kept more than ten species (Benson 1986, cited in Cumming 1991a), though in most areas there are probably more than twenty species of native ungulates. Further, 81 percent of respondents enclosed their game in fences, enabling them to secure legal and physical property rights to the game (Benson 1991). Game ranches cover at least 160,000 square kilometers of South Africa's farmland, but game operations are generally a secondary activity to, and incorporated with, cattle ranching (Cumming 1991a).

These figures suggest that management by private landowners may be important in conserving commercially important species, but as J. Mcnab (1991, p. 2288) asserts, in southern Africa "very few of the privately owned ranches tolerate the presence of large predators. Similarly, habitats on these ranches are manipulated in favour of the commercial species (usually grassland species) to the detriment of those preferring dense thickets or forest. Hence, many species of birds, small mammals, and reptiles that depend on these habitats suffer. At best, one can say that the conservation value of the privately owned commercial ranches is very limited." R. A. Luxmoore (1989) suggests that management for trophy hunting of large predators in southern Africa may be more compatible with biodiversity because it can be carried out at lower densities and thus carries a lower risk of overgrazing. There is also an incentive to maintain greater species diversity, particularly of the rarer species because they generate higher trophy fees. However, owners of commercial wildlife ranches in Zimbabwe are sometimes reluctant to curb wildlife numbers, resulting in overcrowding of wildlife, habitat impoverishment, and loss of species (G. Child 1996). A rigorous assessment of the biodiversity trade-offs of pure cattle ranching, game ranching, and maintenance of more natural areas seems to be lacking. An important consideration in this case is that without commercial wildlife use, game populations might not exist at all on many ranches.

Control of predators to enhance huntable numbers of ungulates is also an issue in North America, particularly in the far northern part of Alaska in the United States and in the Yukon Territory of Canada. To increase populations of moose and caribou, both major game species in this region, wolf (*Canis lupus*) populations are subjected to regular and intensive control by wildlife management agencies. Bear (*Ursus arctos* and *U. americanus*) numbers are also occasionally controlled to reduce predation on moose. Although the relationship is complex, wolves can strongly limit moose populations, at times to densities of less than one-half the carrying capacity of the environment. Thus, controlling wolf (and bear) numbers is a management tool for substantially increasing huntable numbers of moose (Boertje, Valkenburg, and McNay 1996; Van Ballenberghe and Ballard 1994). One study in Alaska and the Yukon found the sustained yield of moose in areas where wolves and bears were lightly hunted to be 0–18 moose per 1,000 square kilometers, whereas areas of heavy wolf and bear offtake yielded 20–130 moose per 1,000 square kilometers. In one area, wolves and bears killed 54 moose per 1,000 square kilometers, restricting the sustainable yield for humans to 6 moose per 1,000 square kilometers (Gasaway et al. 1992).

In Alaska, wolf numbers are controlled primarily by application of liberal hunting and trapping regulations, which include a nearly nine-month season; allowed hunting and trapping in most national parks and national wildlife refuges; no limit on the number of wolves taken under a trapping license; allowed killing of adults with young and of young themselves; and other measures which are clearly directed toward the control of numbers rather than toward the commercial or recreational value of wolf hunting itself. The result is that an average of 1,000 wolves (an estimated 15–20 percent of the population) are killed annually in Alaska, with more than 1,600 taken during the 1993–1994 reporting period. In addition, sterilization, relocation, and other forms of wolf control are emerging in Alaska and the Yukon. For example, in one area, the Alaska Department of Fish and Game proposed to use trapping, snaring, and relocation to reduce at least thirteen family groups to just the alpha pairs, which would be sterilized and left to defend their territories from colonization by reproductively viable wolves (Haber 1996).

Apart from community changes caused by such strong manipulation of a major predator-prey relationship, constant and intensive exploitation may affect the genetics and social behavior of wolves. Significant genetic differences apparently exist between well-established wolf groups. Heavy hunting and trapping would tend to destabilize wolf social structure and thereby erode such genetic patterns. G. C. Haber (1996, p. 1073) suggests that "It is questionable as to whether a normally ultra-social species 'survives' if its social organization is continually shredded by heavy exploitation." R. D. Boertje, P. Valkenburg, and M. E. McNay (1996), however, argue that wolf control in their study area in Alaska has been beneficial for wolf conservation. They provide evidence indicating that wolf control, by allowing prey densities to rise, eventually resulted in a larger wolf population than if no control had occurred. Opportunities to better understand the biodiversity effects of heavy wolf harvest and control are limited by the scarcity of sites that have remained free of wolf harvest for long periods (Gasaway et al. 1992; Haber 1996).

Selective harvesting of males in polygynous and promiscuous species is a common management strategy to increase sustainable harvest levels and to satisfy hunter preferences. In Canada, for example, more male than female polar bears are killed because recreational hunters prefer males and because the selective removal of males is favored by management regulations for both recreational and subsistence hunting. The anticipated result is an unnatural population structure and skewed sex ratio, as appears to be the case in the western Hudson Bay where the adult population is 60 percent female. Because male and female bears in some regions use different habitats during the spring feeding period, the skewed sex ratio may alter the pattern in which nutrients (primarily in the form of seals) are removed from the environment and may increase intrasexual competition among females, with various potential effects on bear biology and ecosystem function (Derocher and Stirling 1992; Stirling and Øritsland 1995). This management regime is in contrast to traditional subsistence hunting of polar bears in which the sexes are equally harvested (Lee and Taylor 1994).

The pursuit of trophy animals places a premium on certain qualities, as suggested by a 1985 article in the hunting magazine *Outdoor Life* titled "Can Science Produce a Race of Super Bucks?" (Etling 1985). This is a slippery slope in game management. The slide begins with selective breeding within a species such as white-tailed deer in order to increase antler size. The next step is to cross native species with another species, as has occurred in efforts to "improve" mouflons (*Ovis musimon*) by crossing them with domestic sheep and to "upgrade" European roe deer (*Capreolus capreolus*) through interbreeding with Siberian roe deer (*Capreolus pygarus*) (Geist 1995). Finally, the demand for bigger or more exotic trophies has resulted in massive and diverse introduction of exotic species of ungulates. Whether such exotic species are introduced into the wild or into fenced game ranches, the long-term consequences are often the same because captive animals inevitably escape.

Examples of introductions into the wild include the European red deer (*Cervus elaphus*) in New Zealand (Howard 1965) and Argentina (Veblen et al. 1989), the white-tailed deer in Europe (Geist 1995), and the European wild boar (*Sus scrofa*) in the United States (Roth 1997b). The lead in introductions of exotic game belongs to the state of Texas in the southern United States. As of 1994, seventy-one exotic species of ungulates totaling 195,483 animals inhabited 637 ranches there (Traweek 1995). At least six species, among them the nilgai (*Boselaphus tragocamelus*) and the Barbary sheep (*Ammotragus lervia*), are well established in free-ranging, unhusbanded populations (Teer 1997).

Grassland Management

The management of grasslands, shrublands, and woodlands for production of domestic livestock involves almost the same range of management interventions as does silviculture. As in forest management, the results range from nearly natural ecosystems composed of native plant and animal species to ecosystems dominated by exotics. Given the propensity for grasslands to switch between alternative states depending on grazing pressure, fire, and other factors, small changes in management interventions can create potentially large changes in the species composition and ecological structure of grasslands.

The introduction of exotic species is probably the most pervasive management intervention in grasslands managed for domestic stock, with major effects on biodiversity. In northern Australia, for example, the poor nutritional quality of native grasslands is compensated for by oversowing them with exotic species, permitting a doubling of stocking rates (Solbrig 1993). In the steppe grasslands of North America, native plant species have been extensively displaced by introduced Eurasian wheatgrasses and ryegrasses (West 1996). In reviewing the range of "improvements" designed for more and better livestock production in western North America, N. E. West (1996, p. 327) concludes that "The net result has been progressively more widespread and intensive use of a landscape that has become partially 'tamed' from the 'wild.'"

Summary

Populations of wild species of value for CCU generally do not grow in the forms and at the densities preferred for commercial markets. Consequently, diverse management interventions are applied to sustain and enhance the production of wild species of commercial value. Which interventions are used and the extent and intensity of their use depend on various factors—the life history of the species, the characteristics of the ecosystem, the profitability of the CCU product, the socioeconomic goals of the resource owner, the demands of the consumer, and financial and technical resources. Offtake and management interventions may intentionally or incidentally alter the genetic makeup of species, species composition, and ecosystem structure and function. Low-level offtake and management interventions may increase, decrease, or cause no noticeable effects on biodiversity at the local or landscape level. As offtake and interventions increase, however, biodiversity and ecosystem naturalness are inevitably compromised. Maximum sustainable offtake of keystone species and intensive management interventions generally result in genetic change in the target species, substantial loss of native species, and a more simplified ecosystem structure. At the extremes of production specialization and management interventions, such as plantation-type forestry, many forms of aquaculture, and game management within fenced enclosures, the ecosystem becomes more tame than wild, more artificial than natural.

Sustainable offtake and ecological sustainability are two separate, often divergent, goals. Specialized production of a wild species may provide a highly sustainable yield. Yet efforts to increase the yield of a target population, and to increase offtake to near MSY, erode biodiversity and thus ecological sustainability. The dilemma for conservationists is that in some ecosystems, production specialization may help make a commodity species and its habitat economically competitive with alternative uses of land or water. Further, enhanced production of CCU products in some areas may relieve pressures to harvest in others.

Ecological sustainability within a given area is thus a balancing act between the incentives for short-term financial gain that encourage overharvesting on one side and the incentives for long-term investment that create the slippery slope of production specialization on the other. Production specialization, however, must be viewed within the larger context of how it affects the socioeconomic sustainability of the management site and harvest pressures at other sites. For biodiversity conservationists, the challenge is to navigate between the Scylla of overexploitation and the Charybdis of overspecialization.

CHAPTER 7

Conservation Benefits of Commercial Consumptive Use

Sustainability is a goal, like liberty or equality; . . . the realist is as skeptical of claims concerning sustainability as she would be of a claim that perfect liberty had been attained.

—Kai N. Lee (1993)

Those who view CCU as a tool for biodiversity conservation largely focus on ways to increase the flow of socioeconomic benefits to the resource owner, with the expectation that if the incentive structure is right, those benefits will motivate the resource owner to manage the target species sustainably and to maintain the ecosystem it inhabits. Other market values, such as nature tourism, may be used in tandem with CCU as a means of expanding socioeconomic returns based on the natural ecosystem. A more protectionist philosophy, in contrast, has little confidence in the marketing of wild species commodities as a tool for biodiversity conservation. From this viewpoint, if species are overharvested, it is best to shut down the markets and stop all harvesting. Government controls and sanctions and other socioeconomic values, such as ecotourism, are believed to be the best way to achieve recovery and habitat conservation. The concept of developing markets and consumptive uses where none exists is seen as dangerous folly.

What evidence is there that given the right social, economic, and ecological conditions, CCU benefits biodiversity conservation? What are the mechanisms by which these benefits might accrue? The most important and thus most often invoked mechanism by which CCU functions as a conservation tool is its ability to socioeconomically outcompete alternative uses of land and water. However, there are at least four other potential mechanisms (table 7-1), which will be reviewed first.

Table 7-1. *Potential Benefits of Commercial Consumptive Use for Biodiversity Conservation*

Type of Benefit	Potential Examples
Production specialization at one site benefits other sites of high biodiversity	Industrial timber production from forests on highly productive soils reduces pressure for more extensive logging of low-productivity, high-biodiversity forests and may generate revenues for their conservation
Production of wild species commodities may be more ecologically benign than production of substitutes	Some wood-based products require less energy to produce than manufactured synthetic substitutes Coastal fisheries are often more efficient in energy usage and use fewer toxic substances than agriculture to produce the same amount of protein
Harvest of wild populations used to restore and maintain natural processes and biodiversity destroyed or degraded by other human activities	Hunting and culling used to control populations of ungulates where predators have been eliminated, thereby avoiding habitat degradation from overbrowsing Forest cutting used to mimic natural disturbance of wildfires in regions where wildfires have been suppressed
People who harvest and use wild species may be more likely to support biodiversity conservation because of increased awareness of nature and of its benefits	Hunters and fishers develop knowledge of and appreciation for the ecosystems in which they hunt and fish and thus support nature conservation Consumers of forest products and fish, particularly if commodities are green labeled, develop greater awareness of wild species and their habitats and thus support their conservation
Socioeconomic benefits from CCU offset the opportunity costs of alternative land or water uses and the costs of living with pest species	Hunting fees and conservation easements purchased by nonprofit hunting organizations may yield greater profits than cattle ranching, providing an incentive to maintain wildlife and their habitats Socioeconomic returns to the community generated by estuarine fisheries may be greater than the cost of reducing industrial effluents that contaminate the estuary and damage the fisheries Hunting fees for large predators that prey on livestock or for herbivores that damage crops make these species an asset rather than a liability

Reduction of Harvest Pressure on Other Important Sites

Intensive production of wild species commodities in one area may decrease pressures to produce the same commodity in other areas that are ecologically more sensitive or important. In reference to North American ecosystems, H. Salwasser and colleagues (1996, p. 554) suggest that to carry out this strategy, "high productivity sites such as flat ground with deep loamy soils, and featured species such as pines, firs, oaks, elk, and trout should be managed with state-of-the-art efficiency in certain places to sustain the production of resources needed by people, thus meeting human needs with minimal impacts on more fragile ecosystems." K. Sjöberg and T. Lennartsson (1995) suggest a similar strategy for the Nordic countries. Implicit in this strategy is the idea that if more people are employed in relatively environmentally benign production activities, fewer will be engaged in environmentally unfriendly pursuits. For example, would the millions of people who now make a living by fishing the tropical coastal waters of the world have to look inland toward tropical forests for their livelihoods if fishing was no longer an option?

In addition, if it is financially successful, production specialization in one area may help finance efforts to conserve natural ecosystems of high biodiversity value elsewhere. This benefit may be diffuse to the extent that CCU can contribute to economic development, which in turn better enables countries to invest in conserving the nonconsumptive-use values of biodiversity, whether through national parks or by other means. To be viable as a conservation strategy, however, planning at the outset must ensure that the development of CCU production areas does not destroy unique or otherwise important biodiversity sites or values within those areas.

This approach requires trade-offs between production zones and biodiversity conservation zones. Such trade-offs can probably be negotiated more easily at the state or provincial level or the national level than at the international level. New Zealand appears to have made such trade-offs in the use and conservation of its forest estate. The objective of the New Zealand Forest Accord, signed in 1991 by government and nongovernmental institutions, explicitly recognizes the separate production and biodiversity conservation functions of these plantation and natural forests: "To . . . recognise that commercial plantation forests of either introduced or indigenous species are an essential source of perpetually renewable fibre and energy offering an alternative to the depletion of natural forests." New Zealand has 1.3 million hectares of plantations, based primarily on an exotic species, the Monterey pine (*Pinus radiata*), and about five times that area, or 6.5 million hectares, is in natural forest. Thus, roughly five hectares of native forestland are being conserved in a more or less natural state for every one hectare under plantation. Because timber production is concentrated in plantation forests, only 4 percent of New Zealand's natural forests were open to exploitation in 1991. Such a division of forest uses provides a conservation strategy distinct from multiple-use forestry, which attempts to combine the goals of timber production and biodiversity conservation (i.e., ecological sustainability) in management of natural forests (Dudley, Jeanrenaud, and Sullivan 1995).

State ownership of New Zealand's forests (though the plantations have now been largely privatized) allowed this countrywide scale of planning to occur. Such planning is much more difficult where forests are mostly privately owned, as in the Nordic countries. In Sweden, for example, individual, noncorporate ownership accounts for more than one-half of all forestland. Such fragmented private ownership provides little flexibility in allocating forestlands based on their relative importance for biodiversity conservation and timber production. Thus, because the area of protected forestland cannot be greatly expanded, new forest management policies in the Nordic countries are oriented toward multiple-use approaches that attempt to integrate the twin goals of biodiversity conservation and timber production (Sjöberg and Lennartsson 1995).

A. B. Anderson and co-workers (1995) suggest that the high productivity and high density of commercially important species in the floodplain forests of the Amazon estuary could provide a means to relieve pressure from other, less productive upland forests in Amazonian Brazil. Forests dominated by the economically valuable palm *Eurterpe oleracea* cover roughly 10,000 square kilometers in the Amazon estuary. This area has an estimated carrying capacity of 50 inhabitants per square kilometers who could be engaged in both subsistence and commercial uses of forest products, and therefore it could support as many as 500,000 people. Thus, although these floodplain forests cover just 0.3 percent of Amazonian Brazil, they could potentially support 12.7 percent of the region's total rural population.

Unintentional trade-offs between production zones and protection zones occur at the global level with internationally traded commodities such as timber. For example, in response to the recent decline in production of wood from forests in western North America, production is increasing in the southeastern United States and eastern Canada. In addition, the reduction in log exports from western North America to the Pacific basin is being offset by increased logging in New Zealand, Chile, and Russia. Logging in Nordic countries and major plantation regions of the world will probably also increase in response to further reductions in logging in western North America, particularly in British Columbia, Canada. It appears less certain whether and to what degree reductions in supply may also be offset by increased timber harvesting in tropical regions such as Brazil, Indonesia, and Malaysia, both from primary and secondary forests and from plantation forests (Brooks 1995; Sedjo 1995b). The implications of these trade-offs for biodiversity conservation have not been evaluated.

The biodiversity trade-offs from increased forest protection in one region of the world and resulting logging increases in other, often biotically distinct, forests also involve questions of scale. For example, compared with the high-volume forests of northwestern North America, a larger area of forest must be harvested in Siberia and the Russian Far East to produce the same volume of timber. However, because of the much larger area of forest cover in Siberia and the Russian Far East, a smaller proportion of the total forest area in these regions would be needed to produce the same volume (Brooks 1995).

Forest management for production of timber, perhaps more than management for production of any other wild species commodity, involves choices about where to harvest on a global scale to optimize biodiversity conservation benefits. R. A. Sedjo (1995b, p. 26) suggests that "Given the fact that logging restrictions in one region will be offset by logging increases elsewhere, the global issue is not whether to log but where to log." If well planned on a regional and global scale, logging at one site, though perhaps locally degrading for biodiversity, can meet the timber demands that would have resulted in logging and greater biodiversity loss in other regions.

The potential for forest plantations to meet much or most of the world's need for timber, particularly for pulpwood and cheap construction timber (Sawyer 1993), provides another option for trading productive timberlands for natural forests. Sedjo (1995a) estimates that the world's current consumption of industrial wood could be produced on about 200 million hectares of good forestland, or roughly 5 percent of the world's current forest area. Total global coverage of forest plantations around 1990 was estimated at 100 million to 155 million hectares, with approximately 75 percent of this in temperate regions (Evans 1986; Sawyer 1993; Sedjo 1995a). The increase in plantation coverage since that time appears significant, although this is even more difficult to estimate than total coverage because replanting figures are often combined with figures for new plantings. One estimate, however, is that 50 percent of all plantations in developing countries were established during the 1980s (FAO 1995b). Moreover, some thirty developing countries have forestry policies that favor plantation development over natural forest extraction as a source of raw industrial material (Spears 1991). Sedjo (1995a) claims that natural forests no longer provide most of the world's industrial wood production (while not explicitly saying that most production therefore comes from plantations); J. J. Gauthier (1991, cited in Sawyer 1993) calculates that plantations provide just 7–10 percent of the world's industrial wood.

Plantations appear to be a particularly attractive option where they can be established on degraded but potentially productive (in terms of timber) lands. If such plantations can produce more timber more efficiently than do natural forests, market forces may largely reallocate land uses to more productive sites without national or global land-use planning. Planning is needed, however, to ensure that natural forests or other natural ecosystems of importance to biodiversity conservation are not converted to plantations. Moreover, plantation production poses a risk wherever it competes with production from natural forests that depend on timber revenues for justifying their conservation. Declines in timber revenues in natural forests due to more efficient production from plantations could result in conversion of these natural forests to other land uses, particularly in some tropical regions (Sawyer 1993). For example, large-scale and efficient mahogany plantations would probably undercut forest conservation efforts based on management of natural forests for mahogany production on the Yucatán Peninsula of Mexico.

Similar trade-offs exist in the conservation of freshwater and, particularly, marine ecosystems. Global aquacultural production increased in volume by 9 percent annually between 1985 and 1990, when it totaled 11.5 million metric tons (excluding algae), divided roughly equally between freshwater and marine systems (Moiseev 1994). Between 1984 and 1993, aquaculture's contribution to the total amount of fish available for human consumption grew from 12 percent to 22 percent (CGIAR 1995). More than an estimated 450,000 square kilometers of shallow areas in the periphery of the world's oceans could be used for aquaculture, with a potential total production of 40–50 million metric tons of fish and shellfish (equal to roughly half the world's current marine catch) and a similar amount in algae production (Moiseev 1994). However, as noted previously, many current aquacultural techniques, particularly in coastal marine systems, significantly alter and often degrade marine ecosystems and native biodiversity. The ready dispersal of escaped or introduced exotic stocks makes this problem difficult to contain within zones designated for aquaculture. Further, the worldwide demand for fish protein may continue to grow at such a pace that it absorbs much of the increased production from aquaculture. Finally, a large and representative portion of the world's shallow coastal waters, regardless of their suitability for aquaculture, should be set aside for biodiversity conservation. Nevertheless, to the extent that protein produced from aquaculture can substitute for protein from natural-production fisheries, aquaculture may reduce fishing pressure on some native stocks.

The potential for trade-offs between production areas and conservation areas also exists in grassland management. For example, intensive management of grasslands (e.g., seeding and fertilization of forage crops) in some regions of the United States could reduce the area needed for cattle raising by an order of magnitude. Thus, theoretically, ten hectares of grassland reserve could be set aside for every hectare put into intensive production (Hunter and Calhoun 1996).

Ecological Friendliness of Some CCU Products Compared with Substitutes

Production of some wild species commodities may be more ecologically benign than production of substitutes. In contrast to substitution between similar products—plantation timber for natural forest timber, for example—the products here clearly differ.

Fisheries present an interesting case regarding the ecological costs that might result if agricultural production were to replace some fisheries as a source of food. Fossil energy use provides one index for assessing ecological costs since both the extraction and burning of fossil fuels entail various negative environmental effects. Based on fossil energy inputs, such as fuel consumption and energy used in the construction and maintenence of equipment, many fisheries yield more kilocalories of protein per kilocalorie input than farm animal production, and are comparable to some field crops. However, the energy efficiency of fisheries varies widely. In the northeastern United States, for example, inshore pelagic fisheries that use small vessels are four times more efficient than offshore, large-vessel fisheries, although the latter are still more efficient than the production of pork, beef, and chicken. The harvest of lobster and shrimp in the United States is several times less energy efficient than the harvest of important commercial fish such as herring, salmon, cod, haddock, and halibut, and considerably less efficient than livestock production. Overall, the ratio of fossil energy input to protein energy output of the fisheries industry in the United States is roughly comparable to chicken and beef production systems (Pimentel and Pimentel 1996; Pimentel, Shanks, and Rylander 1996; Rawitscher and Mayer 1977). In Japan, coastal whaling is estimated to be 10 to 25 times more energy efficient than agricultural and fisheries production (Freeman 1991). A full comparison of the ecological costs of agricultural and fisheries production must consider other effects such as soil erosion and pesticide contamination associated with agriculture and the biodiversity impacts of fisheries examined in chapter 6.

If timber production were reduced worldwide, substitutes for wood would have to be found, particularly for use in construction and as fuel. The extraction or production of these substitutes might have equal or greater environmental costs. For example, metals, cement, and fossil fuels are obtained by often environmentally damaging mining and quarrying activities. Furthermore, many substitutes require more energy to produce than do wood products (Boyd et al. 1976), which would require the burning of more fossil fuels or construction of more hydroelectric plants. Another argument frequently used in favor of wood products is that few wood substitutes are renewable, recyclable, and biodegradable (Sedjo 1995b). For some uses, however, highly synthetic substitutes may be more environmentally benign than wood-based products (Hocking 1991). Nevertheless, a direct and comprehensive comparison of the types and magnitude of environmental trade-offs involved in the production and use of wood compared with nonwood products has yet to be conducted (Brooks 1995). In addition, as noted previously, an important question in assessing the benefits and costs of producing substitutes concerns what the environmental consequences are if, rather than being engaged in the production of wild species commodities, people make a living in other ways that also directly or indirectly affect biodiversity.

Restoration and Maintenance of Ecosystems

The harvest of wild species may be used as a management method to restore or maintain biodiversity and to create conditions that are perceived as more natural. This concept has generated considerable controversy, as it often entails conflicting judgments, based on both science and aesthetic values, regarding what is natural and how effectively people can replace, through harvesting and management intervention, natural processes that have been lost due to human causes. Two areas in which this tool has commonly been applied are the management of large herbivores and forest management.

Populations of large herbivores, it is argued, frequently exceed the carrying capacity of their habitat because predators that formerly controlled their numbers have been lost and because their movements are restricted by habitat fragmentation, fencing, or human settlement. Numbers of large herbivores, in this line of thought, must therefore be controlled by human intervention to avoid degradation of habitat by overgrazing and overbrowsing.

Significant changes, and often declines, in species diversity and structural complexity have been reported in North America where populations of white-tailed deer, elk, and moose have reached high densities because of the absence or rarity of large predators and because of changing land-use practices. Greater offtake is often recommended as a management response (Teer 1997). Large herbivores have been controlled in South Africa's national parks because fencing obstructs their movements, resulting in habitat degradation (Joubert 1991; Pienaar 1963). The most striking examples of habitat and biodiversity change due to changing population levels of a large herbivore are provided by the African elephant, which, because of habitat loss and increasing confinement to protected areas, often reaches artificially high densities in some regions of Africa. Research suggests that at high population densities elephants can change woodlands to grasslands and degrade biodiversity by destroying riverine forests; at low densities they have little effect on existing biodiversity; and at intermediate densities they may increase biodiversity (Dublin, McShane, and Newby 1997).

J. G. Teer (1997) concludes that hunting and culling are important management tools for keeping ungulate numbers in check and, thereby, for maintaining more natural ecosystem processes and biodiversity. Deciding when to apply these management tools, however, often involves a large dose of subjective value judgments (e.g., Dublin and

Taylor 1996). Although biodiversity may be enhanced by controlling numbers of large herbivores, whether this results in a more natural ecosystem is subject to broad interpretation. Such decisions are also made within the context of the management goals and the socioeconomic and political environment of an area. In Africa, for example, where people are short of protein and wildlife management authorities are chronically short of funds, large natural die-offs of game that could have been harvested to provide meat and revenues may be seen as extremely wasteful (Dublin, McShane, and Newby 1997). And although some contend that elephant numbers should be controlled to avoid habitat degradation, others argue that such intervention, even in parks where elephant numbers may be unusually high, is best avoided (Leakey and Lewin 1995). Scientists and conservationists are similarly divided regarding the effects on habitat of the large elk population in Yellowstone National Park in the United States and the need to cull the herd (Teer 1997).

Rapidly expanding populations of geese in North America and resulting habitat degradation caused by salt marsh "eat outs" and other consequences of large numbers of geese have precipitated calls by waterfowl managers to control goose populations. The rapid rise in these populations is attributed primarily to the abundant waste grain provided by agriculture. Liberal hunting regulations have been ineffective in stopping the rise. This has led to proposals to control their numbers through the creation of much more liberal hunting regulations, including, as a "last resort," the harvest and commercial sale of wild geese as food (Ankney 1996).

Debates regarding the need for and desirability of controlling wildlife populations to protect habitats often hinge on the use of terms such as *overpopulation* and *overgrazing*, which imply that there are "right" population and grazing levels that meet management's (and often the public's) sense of what constitutes the correct appearance and species composition for a given ecosystem. Although population control can be a useful management tool, management goals often appear to be set by reverting to the balance-of-nature maxim, which asserts that populations and ecosystems should vary within narrow bounds (McCullough 1997). There may be an aversion to allowing a population and an ecosystem to fluctuate widely and reach what may be a new dynamic equilibrium because the process and outcome are too messy and unpredictable. The changes may not fit management's or society's notions of what the ecosystem ought to look like and what it should produce in terms of both consumptive- and nonconsumptive-use benefits.

These values and benefits, and the fact that they may require management to deviate from more natural conditions, need to be more explicitly recognized and articulated. The African Elephant Specialist Group of the World Conservation Union (IUCN), for example, has adopted a concept called "preferred management densities," which is being used in several countries. A key element of the concept is that elephant population densities should be kept within acceptable limits in an ecosystem. These limits are based on management objectives—tourism, biodiversity conservation, economic performance, and the like—for the area in question (H. Dublin, pers. comm., 1997).

Compared with well-planned culling programs, recreational hunting is often an ineffective tool for controlling populations. In hunting of both deer and geese, for example, hunters often fail to take the number of individuals recommended by management and allowed by regulations (Ankney 1996; Teer 1997). Trophy hunting is not a tool for controlling elephant numbers in Africa (H. Dublin, pers. comm., 1997). In addition, as noted in chapter 6, trophy hunting in particular does not mimic the patterns of mortality created by nonhuman predators and other natural causes. In Zimbabwe and South Africa, where managers depend on culling to regulate elephant numbers, entire family units are often removed to minimize disruption of elephant social behavior. The sale of elephant products such as skin and ivory (the meat generally goes to local communities) from culled elephants made this a profitable activity until the 1989 CITES ban on trade in elephant products (Dublin, McShane, and Newby 1997). Thus, where control of wildlife populations is deemed necessary, organized culling may often be a more effective tool than recreational hunting.

Clear-cuts and salvage logging have been promoted as tools for mimicking natural disturbances and improving forest health (Lippke and Oliver 1993). For example, periodic fires in the inland forests of the Pacific Northwest of North America were essential in maintaining forest structure, species com position, and ecosystem processes, but since European settlement, natural fires have been suppressed to protect property and timber values. Although on a local scale the effects of clear-cuts may superficially resemble the effects of a fire on vegetation structure, clear-cutting has been performed more frequently and on a far greater spatial scale than disturbances caused by fire in this region before European settlement. Compared with the uneven effects and patchy distribution of natural fires, continued clear-cutting in such a highly disturbed landscape is unlikely to provide refugia for plants and animals that recolonize logged sites. Unlike fire, persistent logging can significantly deplete nutrients, minerals, and elements that require centuries to accumulate in forest soils (DellaSala et al. 1995). In addition, the fallen logs and tip-up mounds resulting from fires and windstorms create a structural complexity on the forest floor that may be important in maintaining many forest species (Halpern and Spies 1995). This complexity is lost in the traditional practice of removing all merchantable wood during clear-cutting operations. Similar issues regarding the maintenance of structural complexity and nutrient levels arise when salvage logging is proposed to reduce dead wood and thus, it is argued, reduce fire hazard in forests that have experienced mortality from epizootics and previous fires (DellaSala et al. 1995).

Maintenance of the Link between Humans and Nature

The harvest of wild species and use of their products can serve to maintain an awareness of the link between human welfare and natural ecological systems, and more generally, it may simply elevate human awareness of and appreciation for wild species and nature. This is Aldo Leopold's (1949, p. 178) "split-rail value," the sense that "there is value in any experience that reminds us of our dependence on the soil-plant-animal-man food chain, and of the fundamental organization of the biota." Although Leopold primarily had in mind hunting, fishing, and other outdoor pursuits, the urbanite's equivalent split-rail experience might be found in an appreciation for hardwood furniture, natural medicinals, fresh fish at the supermarket, and a Brazil-nut crunch in ice cream. Green labeling of wild species commodities can enhance consumers' awareness of the human-nature connection. The conservation benefit of this increased awareness may operate through the various mechanisms—votes, donations, personal choices, and so on—by which an individual can support biodiversity conservation. Such conservation benefits and the role of CCU in creating them are difficult to evaluate, but the split-rail value of CCU seems potentially significant.

Offsetting the Opportunity Costs of Alternative Uses

The most direct role of CCU in conservation is its contribution to offsetting the opportunity costs of alternative uses of land and water that result in degradation or loss of the natural ecosystem. In some cases, the alternative use is obvious and localized, such as conversion of a forest into a cornfield or of a wetland into a rice paddy. In other cases, the alternative use is subtle and dispersed, such as use of the world's oceans for disposal of toxic wastes. Here, the opportunity cost is the money governments and corporations save by not having to dispose of wastes in more environmentally benign ways. Sometimes the alternative use is represented by the landowner's decision to eliminate a wild species that damages property and crops or endangers lives. Thus, the opportunity cost with which fee hunting must compete may not be the profits earned from a distinctly different land use, but may simply be the "cost of living with wildlife" (Modise 1996). Noneconomic values may weigh heavily in these often subtle choices made by landowners.

Although the importance of CCU as a mechanism to justify the conservation of natural ecosystems in the face of alternative land and water uses has received considerable attention from conservationists, examples tend to be anecdotal and actual opportunity costs are seldom calculated. Following is an examination of how three broad categories of CCU—recreational hunting, management of forests for timber and nontimber products, and marine fisheries—have contributed to conserving biodiversity by competing with alternative uses of land and water.

Recreational Hunting

Recreational hunting (as well as recreational fishing) provides several market mechanisms for generating revenues to compete with alternative land uses. As previously noted, recreational hunting and fishing are unique among the various types of CCU in that the product for which the consumer pays is more than just the organism itself or a product from the organism. Value is also determined by the environment in which hunting and fishing take place. The average hunter would not pay to shoot an animal locked up in a pen or to walk with gun in hand through a natural landscape where there is no game.

Safari Hunting in Africa

Safari hunting in Zimbabwe has been referred to as "low-impact tourism" because safari hunters tend to travel in smaller numbers than do other tourists and they are usually satisfied with more basic amenities. In addition, safari hunters pay much more to hunt than other tourists pay to view wildlife; a single foreign hunter may spend more than $40,000 on a single trip to Zimbabwe (CAMPFIRE n.d.). In South Africa, the average foreign hunter spends seven times more per day than the "normal" tourist (Meiring 1994, cited in Crowe et al. 1997). Thus, in one sense, a portion of the revenues generated by recreational hunting could justifiably be attributed to a nonconsumptive use—nature tourism.

Recreational hunting, together with culling of game for the venison market and live game sales, has been widely cited as important for maintaining wildlife and wildlands in the face of competing land uses, largely livestock grazing and subsistence agriculture, in southern Africa. D. H. M. Cumming (1991b) provides a comprehensive review of the competing interests of wildlife conservation and the cattle industry in Zimbabwe since the turn of the twentieth century. From 1900 to 1990, the human population grew from less than 500,000 to 10 million, and from 1900 to 1976, the number of cattle grew from less than 50,000 to about 6.5 million. Wildlife was viewed largely as a threat to livestock because it competed for forage and served as a reservoir for disease. As a result, wildlife populations declined throughout the country. Facilitated by changes in wildlife policies that gave private landowners greater flexibility to earn revenues from and manage wildlife on their property, by the early 1960s game animal populations began to recover as a formal game-ranching industry, initially based largely on the venison market but later based increasingly on safari hunting, developed on commercial farmland. The area of commercial farmland dedicated to commercial wildlife use grew from 350 square kilometers in 1960 to 27,000 square kilometers by 1990. The government also greatly expanded its direct involvement in safari hunting during this period with the development of state-owned safari areas for recreational hunting, much of it accomplished through leases to commercial safari operations.

Meanwhile, Zimbabwe's CAMPFIRE program, officially initiated in 1989, has provided a framework and support for developing community-based wildlife management on communal lands, which cover 42 percent of the country (B. Child 1996; Cumming 1991a). In 1995, recreational hunting accounted for 93 percent of total revenues from all consumptive and nonconsumptive (i.e., tourism) uses of wildlife on communal lands (CAMPFIRE Collaborative Group n.d.). Although the sustainability of the CAMPFIRE approach has been questioned (Barrett and Arcese 1995; but see the retort in Murphree 1996), its potential as a conservation tool is indicated by the fact that as of 1993 nearly 400,000 people on some 30,000 square kilometers of land had enlisted in the program (I. Bond 1993; CAMPFIRE n.d.).

Apart from the obvious benefit to game populations, a comprehensive assessment of the biodiversity benefits of these game management programs has not been conducted in Zimbabwe. According to Cumming (1991b), however, populations of endangered species such as black rhinoceros, white rhinoceros (*Ceratotherium simum*), and cheetah (*Acinonyx jubatus*) have increased on commercial ranches over the past thirty years. Although black rhino and cheetah cannot be hunted, their presence is an added attraction to both fee-paying hunters and photographers. Recovery of the tsessebe (*Damaliscus lunatus*), formerly endangered in Zimbabwe, occurred initially on commercial ranches, and live sales have now dispersed the species throughout the farming areas of the country. Leopards, despite livestock predation problems, are better tolerated on farmlands because they can be included in safari hunts. Cumming also notes that after fifteen years of operations as a safari area, it was clear that the state-run Matetsi Safari Area had "recovered ecologically" from its status as commercial farmland. Wild populations of the Nile crocodile (*Crocodylus niloticus*) have also been protected and have increased in recent decades, as they are a source of eggs for the expanding crocodile farming industry. Communities in the CAMPFIRE program have reduced tree cutting and annual burning to improve wildlife habitat, and large areas of land are being zoned and set aside for wildlife (CAMPFIRE n.d.). Cumming (1991b, pp. 19–20) concludes that "There is little doubt that the commercial consumptive use of wildlife in Zimbabwe has permitted some 50,000 square kilometers to be retained under wildlife utilization. . . . The greater proportion of this land would not now be under wildlife if consumptive use and sale of [wildlife] products had not been possible. Were commercial sale of products to be stopped tomorrow much of this land, which amounts to more than twice the area of National Parks, would be put to other uses" (figure 7-1).

Wildlife production systems appear to have a significant potential to outcompete cattle ranching over larger areas in Zimbabwe and elsewhere in southern Africa. In Zimbabwe, financial returns on private ranches could be slightly to significantly greater from wildlife production than from cattle in the three driest regions in the country, which represent 82 percent of the land area. As explained in chapter 4, this advantage often disappears on communal lands because the govern-

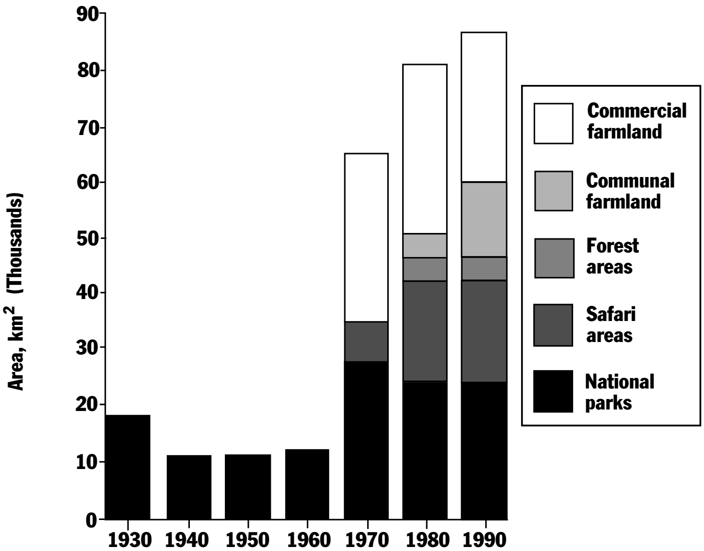

Figure 7-1. *Increase in land area available to wildlife in Zimbabwe, 1930–1990.* Source: *Cumming 1991b.*

ment retains some of the wildlife revenues (I. Bond 1993). A comparison of the financial returns from game ranching and cattle ranching on the Rooipoort ranch in the Cape Province of South Africa also indicated that game animals are more profitable than cattle in this region. Return on operating expenses from game sales and hunting was three times greater than return from the cattle operation. The profitability of wildlife use could be further increased if the ranch put more effort into expanding its hunting operation, including the hunting of upland game birds (Crowe et al. 1997).

Botswana, Namibia, and Zambia all have programs under way whereby greater wildlife ownership and management responsibility are being devolved to local communities (Hirschoff, Metcalfe, and Rihoy 1996; Steiner and Rihoy 1995). The resulting land area that could be devoted to wildlife management rather than alternative land uses is significant. In Zambia, for example, where World Bank reports indicate that only 1.2 million of a potential 9 million hectares are being used for agriculture, wildlife use may be a competitive option for much of this land (Steiner and Rihoy 1995). In Botswana, wildlife management areas, in which wildlife use is the designated primary form of land use, represent more than 20 percent of the country's land area (Modise 1996; Steiner and Rihoy 1995). A. S. Steiner and E. Rihoy (1995, p. 6) conclude that "In view of its size, low population density and the fact that much of Botswana is unsuitable for arable agriculture it is likely that wildlife management will develop into an increasingly competitive form of land use." J. I. Barnes (1996), however, warns that at the present 3.8 percent growth rate of Botswana's livestock population, within fifteen years livestock will occupy some 5 million hectares of land presently allocated to wildlife, which will mean conversion of some 68 percent of the current wet-season dispersal range of the elephant. Barnes's economic analysis indicates that the only obvious means to avoid this scenario is for wildlife, particularly elephants, to recapture much (roughly half) of the direct-use value lost after the 1989 CITES ban on trade in ivory and other elephant products. He suggests that increased revenues from both safari hunting and sale of various elephant products will be crucial in achieving this and that nonconsumptive-use and nonuse values should also be developed to maximize the value of wildlife and its competitiveness as a land-use option. Rapid population growth and immigration to communal lands represent serious challenges to the future of wildlife programs and their ability to outcompete alternative land uses throughout southern Africa.

Trophy Hunting in Asia

In Pakistan, the recent development of trophy-hunting programs for three species of caprinid—the markhor, the urial (*Ovis vignei*), and the ibex (*Capra ibex*)—is converting these species into an asset for local people. The most successful is the Torghar Conservation Project in Baluchistan Province, which was initiated in 1986 by local tribal people. Based on a primarily European clientele, a conservative level of offtake, and trophy fees of roughly $25,000 for markhor and $11,000 for urial, the project has been self-sufficient since its inception, with a total income of $460,000 as of 1996. By 1994, thirty-three local game guards, paid entirely by hunting revenues, were protecting approximately 1,000 square kilometers of land and virtually all poaching had been eliminated. Both markhor and urial populations, which had been almost extirpated in the region, have grown steadily since the beginning of the project. Broader biodiversity benefits include apparent reduction of heavy hunting pressure on other species, such as the Indian wolf, and plans to reduce grazing pressure by livestock (Johnson 1997).

A hunting program initiated in 1985 in the seventy-five-square-kilometer Dulan International Hunting Area, in the province of Qinghai, People's Republic of China, also shows signs of improving local incentives for wildlife conservation. Based on trophy fees as high as $2,400 for the blue sheep (*Pseudois nayaur*), the primary game species, as well as other ancillary service and license fees, foreign hunters generated gross revenues of $560,000 for the area between 1985 and 1991. A large share of the revenues went to government agencies, which reinvested a portion in the management program, and local people benefited from increased income and funding for local elementary schools. In addition, many families interviewed indicated that they valued the interaction with foreign visitors, a form of cultural currency for wildlife conservation. The program appears to have won the support of local people and reduced poaching within the hunting area. Proposals to allow hunting of the argali (*Ovis ammon*), which commands a trophy fee of $18,000, could, if sustainable, greatly increase income for the program (Liu 1995).

Recreational Hunting in North America

Recreational hunting has been widely cited as an effective tool for habitat conservation in Canada and the United States. Three significant markets or quasi-markets exist for the flow of monetary benefits from hunting into habitat conservation: (1) government fees and taxes levied on hunters and their equipment (guns and ammunition); (2) fees paid by hunters to private landowners for the right to hunt on their land; (3) contributions by hunters to nonprofit organizations that work to conserve habitat of huntable species. In the United States, for example, both the federal government and state governments charge fees for hunting migratory waterfowl, and individual states have license fees for hunting nonmigratory species, thus creating a quasi-market whereby hunters can choose among the "offers" made by states. The significance of government fees paid by hunters is illustrated by the Migratory Bird Hunting Permit, commonly known as the Duck Stamp. Since its inception in 1934, more than $442 million has been collected by the Duck Stamp program, which has been used to purchase or lease more than 1.6 million hectares of wetlands (R. Graves, pers. comm., 1997).

Wildlife in Canada and the United States, as in most countries, is publicly owned, but owners of private land can charge trespass fees for access to wildlife and thus exercise de facto ownership of it (Benson 1992). Although many authors have attempted to link financial returns from fee hunting and game ranching to conservation (Burger and Teer 1981; Langner 1987; Schenck et al. 1987; Teer, Burger, and Deknatel 1983; Wesley 1987; White 1986), D. E. Benson (1991, p. 498) correctly notes that "Few examples exist in the US to enable a critical review of the quantity or quality of private wildlife enterprises and their contribution to wildlife conservation."

The most significant fee hunting region in North America is in the state of Texas in the United States, where landowners received $100–$300 million from hunting leases in 1987 (Steinbach et al. 1987). Although G. V. Burger and J. G. Teer concluded that the lease system was important in averting the conversion of range resources to other crops in Texas, they also noted that few landowners invest much capital in habitat management. More recently, Teer (pers. comm., 1994) comments that though "livestock takes second place in economic returns to the hunting lease system on great acreages west of the 100th meridian in Texas," as yet "there are very little data to demonstrate habitat improvement and protection of biodiversity derived from the hunting lease system." Fee hunting on private lands in Texas has resulted in a notably perverse economic incentive in terms of biodiversity conservation by encouraging the massive introduction of exotic ungulates (see chapter 6).

Results of a study of fee hunting of waterfowl on private lands in the state of Oregon in the northwestern United States indicated that financial returns, as well as aesthetic appreciation and personal enjoyment from hunting, were incentives for farmers to improve waterfowl habitat. Although farmers identified crop depredation by waterfowl as a significant problem, there was no conclusive evidence that this was a deterrent to improved waterfowl management on private lands (Rasker, Johnson, and Cleaves 1991).

The growth of nonprofit, largely hunter-based organizations in North America is an increasingly significant market mechanism through which individuals can pay to maintain both hunting and nonhunting values of wildlife. Two examples of nonprofit organizations formed and supported largely by the hunting community in North America are Ducks Unlimited and the Rocky Mountain Elk Foundation.

Ducks Unlimited, founded in 1937 and with 580,000 members in 1996, has raised nearly $1 billion since its inception for waterfowl and wetland conservation. Ducks Unlimited reports that in 1995, in partnership with private landowners and governments, it improved or created 173,470 hectares of waterfowl habitat in North America and that it has helped restore and conserve 2.95 million hectares of habitat over its history. Eighty-eight percent of this area is in Canada and Mexico, though by far the bulk of Ducks Unlimited's income is from the United States (Ducks Unlimited 1996). The broader biodiversity effects of wetland management programs supported by Ducks Unlimited, particularly activities affecting natural wetlands, are largely undocumented.

The Rocky Mountain Elk Foundation, with 98,000 members from twenty-eight countries, was founded in 1984 by hunters interested primarily in maintaining elk numbers and habitat. In 1995, it raised more than $9 million from its membership, and in its brief history it claims to have helped conserve 720,000 hectares of elk habitat and spent more than $50 million on projects (Rocky Mountain Elk Foundation 1995).

Again, as in the concept of the hunter as tourist, members make contributions to these organizations not only for the purpose of improving hunting but also to promote broader goals of habitat conservation.

Forest Management for Timber and Nontimber Products

The opportunity cost of alternative uses for much of the world's forestlands is relatively low, if not essentially zero. The world's boreal forests and temperate coniferous forests (particularly those in mountainous regions) are generally on lands that are unsuitable for productive agriculture or livestock operations. In many cases, the most significant competing land use is unsustainable logging, with its resulting land degradation, or conversion of forestlands to various intensities of silvicultural production, including monoculture plantations. Recreational and residential development (e.g., ski resorts, subdivisions) and mining pose high opportunity costs and serious environmental problems in some cases, but these represent a small portion of the total land area. Some of the world's wet tropical forests also present few viable alternative uses of the land, but high population growth, poverty, and a shortage of good agricultural land mean that even very low monetary returns from farming and cattle ranching on tropical forestlands, aided by proagricultural policies and subsidies, are often sufficient to cause forest conversion.

The following examples illustrate how forest use and conservation can be competitive with alternative land uses. Some of the questions and challenges facing this conservation approach are also examined.

India

Restoration of forestland in India, achieved by giving local communities a greater stake in forest management, is an example in which both the commercial and subsistence use of wild species products are providing an incentive to improve degraded lands. Reforestation provides an alternative use for India's nearly 175 million hectares of unproductive wastelands. However, two major approaches to reforestation are under way in India. One involves largely government-sponsored initiatives to reforest with monoculture plantations that focus on wood production, commonly using exotic species such as eucalyptus. The other involves community-based initiatives that rely on largely natural forest regeneration and a wide mix of forest products. In the industrialized and more fertile regions of northern and northwestern India, the tendency has been toward plantation forestry, in part because of a strong demand for timber there. In the less fertile and more poverty-ridden states of eastern India, however, regeneration of natural forests under community-based management is evolving much more rapidly than plantation forestry and is reclaiming large areas of wasteland. In the states of Bihar, Orissa, and West Bengal, for example, more than 2.76 million hectares of forest are regenerating through community-based management and producing a diversity of timber and nontimber forest products (Poffenberger 1995; Sharma 1993, 1995; Singh et al. 1997).

Despite these promising trends, the demand for land presents a daunting challenge to the future of forest management in India. India's forest-dependent populations require an estimated minimum of 0.5 hectare of forestland per capita, but the current average is 0.1 hectare per capita. Meanwhile, India's population continues to grow toward 1 billion, with rapid expansion into poor rural areas (Poffenberger 1995).

Quintana Roo, Mexico

The Plan Piloto Forestal in the state of Quintana Roo on Mexico's Yucatán Peninsula offers another example in which commodity production from a forest may be checking the advance of alternative land uses. In contrast to the prominence of nontimber products in community-based forest management in India, revenues from communally managed lands of the Plan Piloto Forestal are based principally on timber and, to a lesser extent, chicle. In 1993, revenues per communal member ranged from $200 to $1,435. These revenues are generally split, part of them funding improvements in public infrastructure in the communities and part of them distributed as cash payments to communal members. This often represents communal members' sole or most important source of cash, as production from their agricultural lands goes largely to meet subsistence needs. Revenues from the forest probably could have been significantly greater had mahogany not been intensively harvested under a concession to a logging company before management was turned over to the communities in 1983 (Kiernan and Freese 1997).

Thus far, the Plan Piloto Forestal has been rapidly adopted on both communal and native people's lands in Quintana Roo, with significant benefits for biodiversity. According to M. Kiernan and C. H. Freese (1997, p. 118), "Viewed from the broader landscape perspective, the Permanent Forests of the Plan Piloto Forestal form a regional matrix of forest cover of some 500,000 ha that apparently harbors the full array of floral and faunal diversity indigenous to that region." The value of these forests, however, rests almost entirely on production of wood and chicle. Thus, at this early stage of the program, it remains uncertain whether these are sufficient to successfully outcompete crop agriculture and cattle ranching for use of the land, activities that often benefit from government subsidies. The advantage of long-term forest production, particularly given the soils and climate of Quintana Roo, is that timber productivity should be more stable than agricultural productivity and, in contrast to agricultural products, mahogany has one of the most stable markets in the world.

Northwestern United States

Natural forest management must often compete with intensive silviculture and plantation forestry in the coniferous forests of temperate regions. A. J. Hansen and colleagues (1995) conducted a simulation experiment to compare the economic returns of plantation management with those of more natural management systems in western Oregon of the northwestern United States, where western hemlock (*Tsuga heterophylla*), Douglas fir (*Pseudotsuga menziesii*), and western red cedar (*Thuja plicata*) are dominant tree species. Although silvicultural practices in national forests in this region have recently begun including retention of canopy trees, snags, and coarse woody debris within har-

Table 7-2. *Simulated Average Cumulative Value of Wood Products under Different Silvicultural Treatments, U.S. Pacific Northwest*

	Value (U.S.$/ha) According to Rotation Age		
Retention Level	240 yr.	80 yr.	40 yr.
0 trees/ha	393,677	416,257	402,335
5 trees/ha	317,524	341,817	333,786
50 trees/ha	199,416	215,562	225,304
150 trees/ha	148,219	147,312	150,247

Source: Hansen et al. 1995.

vest units, most private lands are in short-rotation plantations using a clear-cut system. To assess the effects on economic productivity produced by changes in rotation age and in the level of live canopy-tree retention in harvest units, Hansen and colleagues' analysis included rotation periods ranging from 40 to 240 years and canopy-tree retention ranging from zero to 150 trees per hectare. Longer rotation periods and greater levels of canopy-tree retention create patterns of disturbance and succession that more closely mimic those in natural forests and create higher levels of structural and species diversity than is found in plantation forests of the region.

Comparing the results of management alternatives over a 240-year period, the simulation predicted that wood production would drop substantially with either increasing canopy-tree retention or longer rotation periods. However, the cumulative value of wood products over the 240-year period would decline very little for longer rotation periods (table 7-2). Although increasing canopy-tree retention still produces significant drops in revenues, it does not drop as fast as wood volume decreases. The reason for this disparity is the higher value of the large-dimension, high-quality trees that could be harvested under higher canopy-tree retention and longer rotation periods, which partially compensate for lower overall productivity. According to Hansen and colleagues (1995, p. 549) "This suggests that, in the absence of discounting, longer rotations do not reduce long-term economic returns compared with short rotations." However, they note that most forest economists do discount, and at the conventional discount rate of 4 percent used by the U.S. Department of Agriculture's Forest Service, longer rotations may not be as profitable as short rotations. Apart from the discount rate, another factor that might favor plantation-type forestry is the higher cost of planning and logging incurred with canopy-tree retention compared with clear-cutting.

Variables not accounted for in this analysis that could tip the scale toward more natural forest management (i.e., retention of more canopy trees and longer rotations) include the possibility that forest productivity will decline over time with intensive silvicultural practices, the recent accelerated increase in value of large-dimension, high-quality logs compared with wood products as a whole, and the omission of revenues generated by fishing, mushroom harvesting, recreation, and other forest uses. C. Best and L. Wayburn (1995, p. 7) suggest that in some forests of this region, "emerging markets for mushrooms, understory florals and greens, lesser known commercial tree species, medicinals, and other special products are beginning to yield income that can rival that of timber." Although the ability of timber and nontimber commodities from natural forest management to economically outcompete plantation forestry in this region has yet to be demonstrated, the potential appears to exist for some forestlands.

Amazonia

The development of markets for nontimber forest products, as opposed to that for timber, has received considerable attention as a way for natural forests both to provide a livelihood for residents of tropical forest regions and to compete with alternative land uses there (Gradwohl and Greenberg 1988; Peluso 1992; Peters, Gentry, and Mendelsohn 1989; Schwartzman 1992; Vásquez and Gentry 1989). The development of extractive reserves in Brazil in the late 1980s and early 1990s constitutes a large-scale experiment on the socioeconomic and ecological sustainability of forest extractivism and its ability to outcompete alternative land uses. Due largely to Brazilian rubber tappers' demands that forested lands be protected from colonization by farmers and cattle ranchers, by 1994 Brazil had created nineteen extractive reserves covering 3,090,348 hectares (Pinzón Rueda 1995). Thus, the basis for creation of the reserves involved both an attempt to resolve social conflicts and at least a tacit recognition that forest extractivism was a viable land-use alternative to farming and cattle ranching. Although the primary economic activity in extractive reserves is based on nontimber plant products and, to a lesser degree, fish and wildlife products, small-scale agriculture is also practiced, mainly to meet subsistence needs. Further, modest management interventions are applied in some forests to produce key commodities such as rubber, Brazil nuts, and various products from several species of palm (Ruiz Murrieta and Pinzón Rueda 1995).

The socioeconomic and ecological sustainability of extractivism, however, remain in doubt. In reviewing the history of extractivism, particularly rubber tapping, in Amazonia, J. O. Browder (1992, p. 176) concludes that "Extractive economies are unstable over time and not indefinitely self-sustainable." N. Salafsky, B. L. Dugelby, and J. W. Terborgh (1993, p. 50) caution that "Extractive reserves are not the panacea some people would have them be." Many of the shortcomings posed by nontimber commodities as a viable form of forest use can be traced to several socioeconomic factors—poverty and indebtedness to traders, transient living, open-access resource use, distance from markets—that deter long-term investment in more sustainable practices. Official designation of extractive reserves in Brazil is an attempt to address some of these problems. The viability of extractive reserves, and more generally of commerce in nontimber forest products, as a means to outcompete alternative forms of land use in tropical forests remains to be adequately tested and documented, though there are clear indications that in some forests it can be highly competitive.

Several attempts have been made to estimate the economic value of nontimber products harvested from tropical forests. R. Godoy, R. Lubowski, and A. Markandaya (1993) found a median value of about $50 per hectare per year, with a range of $1 to $422 per hectare per year, for nontimber forest products reported in twenty-four studies from tropical regions in Asia, Africa, and Latin America. Potentially higher economic returns can be obtained from oligarchic forests in Amazonia. C. M. Peters (1992) estimated that the market value of fruits was $4,242 per hectare per year for the understory species *Grias peruviana* and $6,660 per hectare per year for the shrub *Myrciaria dubia,* both of which form dense stands in alluvial conditions in northern Amazonian Peru. But these estimates are for highly localized conditions and thus cannot be extrapolated to large areas. Godoy and coworkers point out the difficulty of interpreting and comparing these studies because economic valuation methods differ widely; there is a tendency to examine plant products (mostly) or animal products, but not both; and little attention is given to sustainability.

These factors and our ignorance of the opportunity costs of alternative land uses make it difficult to assess the value of nontimber forest commodities as a conservation tool. For example, C. M. Peters, A. H.

Gentry, and R. O. Mendelsohn (1989) calculated, based on sustainable offtake, a net value of $422 per hectare per year from harvesting fruits and latex and a net value of approximately $15 per hectare per year from harvesting timber in a region of northern Amazonian Peru. Godoy and co-workers, however, suggest that had Peters and co-workers based their estimate on product flow (the quantity actually used by people) rather than on inventory (the stock quantity in the forest), the value of fruits and latex would be only $15 per hectare per year, or less than 4 percent of their original calculation and the same figure as calculated for timber.

A. B. Anderson and E. M. Ioris (1992) report that deforestation for agriculture and timber extraction had been common in the Amazon estuary, but when strong markets for the fruit of the palm *Eurterpe oleracea* and other forest products developed in the nearby city of Belém, island residents turned increasingly to managed and natural forests as the major form of land use. A study of land-use patterns on three islands in the Amazon estuary found that an average of 2 percent of land was devoted to home gardens, 1 percent to swidden agriculture, 1 percent to perennial cash crops and plantations, 55 percent to managed forest (with reduced diversity of tree species compared with unmanaged forest), and 41 percent to unmanaged forest (Anderson et al. 1995). Although the fruit of *E. oleracea* provides as much as 80 percent of the annual income of rural families in this region, families avoid extreme specialization in production of this fruit because other factors act as incentives to maintain a diversity of forest products. Because *E. oleracea* fruit is abundant only in the dry season, diversification to meet subsistence needs when household income is low becomes the principal economic strategy in the rainy season. Diversification also minimizes the risks inherent in dependence on a single product; a diversified forest with diverse products allows families to switch readily to production of other forest products should prices fall for the fruit of *E. oleracea*. Another factor favoring forest cover over agriculture or plantation-type forestry as a land use, particularly for low-income producers, are the minimal material and labor requirements of floodplain forest management (Anderson and Ioris 1992).

Estuarine floodplain forests in the Amazon present a unique advantage in terms of nontimber forest commodities as a viable land-use option in that, compared with upland forests, they exhibit low biodiversity and are often dominated by one or a few species of substantial economic importance (Anderson et al. 1995). A similar pattern emerges in Amazonian Peru, where, according to O. Phillips (1993, pp. 27–29), "Substantial forest fruit production is found mainly in palm-rich swamps and frequently inundated floodplains; areas where, in spite of the nutrient-rich soils, flooding makes agriculture difficult or even impossible." He concludes that only in these relatively rare forest types is harvest of forest fruit the most productive land-use option on an area basis; otherwise, traditional and commercial agriculture, with the exception of cattle ranching, are the most efficient food production systems on most soil types. Further, management and marketing of nontimber products as a land-use option is of limited potential for new immigrants colonizing regions of Amazonia because of land tenure issues and their general ignorance regarding the forest environment and its products (Phillips 1993).

In contrast to Phillips's conclusions, A. Grimes and colleagues (1994) found the net present economic value of fruits, medicinal bark, and resins to be higher in upland forests than in floodplain forests in Amazonian Ecuador. They also found that regardless of forest type (upland or alluvial), collection of these products yielded a greater economic return than did competing land uses (table 7-3). The relative advantage of natural forest production would be even greater had they included the value of medicinal herbs, flowers, and game. Although their results indicate that forests currently used for natural production earn more than those employed in competing land uses, they caution against inferring that all remaining forests should be used for nontimber forest products, since the increase in supply resulting from such use could significantly decrease prices.

Table 7-3. *Net Present Value of Alternative Land Uses in the Upper Napo Region of Amazonian Ecuador (5% discount rate)*

Land Use	Net Present Value (U.S.$/ha)
NTFP, upland plot A[a]	2,939
NTFP, upland plot B	2,721
NTFP, upland plot C	1,257
Timber, upland plot A	188
Agriculture	<500
Cattle ranching	57–287
Local land prices	50–220

Source: Grimes et al. 1994.

[a] NTFP = nontimber forest products. Plot A was assumed to be more profitable for timber than plot B or plot C.

Other Tropical Regions

The ability of nontimber forest products to compete effectively with alternative land uses in the tropical forests of Africa and Southeast Asia has not been so extensively researched as in Latin America, though the diversity and economic value of nontimber products there are significant. As noted earlier, nontimber products play a major role in forest restoration in India, and the maintenance of numerous forest patches by rural communities in Thailand (see chapter 4) appears to be based primarily on the value of their nontimber products (Poffenberger and McGean 1993).

D. C. Lawrence, M. Leighton, and D. R. Peart (1995) argue that because of low product density in primary forests near Gunung Palung National Park in West Kalimantan, Indonesia, extractive reserves or buffer zones may not provide sufficient economic incentive to protect primary forests. N. Salafsky, B. L. Dugelby, and J. W. Terborgh (1993) arrived at the same conclusion in their review of the potential for nontimber products as a forest conservation strategy in West Kalimantan. However, given the right social and institutional framework in Indonesia, rattan, Southeast Asia's most valuable (but overharvested) nontimber forest product, could provide a cornerstone for forest conservation (Peluso 1992; Siebert 1993). Success in natural forest management in Indonesia, as in many other forest regions, may require that managers "simultaneously manage timber and nontimber resources to compete economically with agricultural alternatives or timber mining" (Siebert 1993, p. 749).

In Africa, although diverse nontimber forest (and woodland) products are commercially harvested for food, medicines, and other uses (e.g., Cunningham 1993), attempts to use them as a primary tool for habitat conservation seem poorly developed. For example, A. B. Cunningham (1994), citing problems of overexploitation, recommends that conservation efforts focus on ways to deflect the demand for harvesting commercially important wild plant species from priority conservation sites in southern Africa by promoting domestic cultivation of such plants and the use of substitutes. One place where economic incentives based on nontimber products show promise is in and around Kasungu National Park, Malawi, where programs for harvest of the mopani worm (*Gonimbrasia belina* and *Gynanisa maia*) and for beekeeping, both dependent on woodlands within the park, have been developed. Estimated potential combined economic returns from mopani worms and beekeeping are more than twice the combined returns from typical crops of maize, beans, and groundnuts but are roughly half those from tobacco. Regardless of the relative profitability of these enterprises, use of the park for mopani worm collection and beekeeping has begun to engender local support for conservation (Munthali and Mughogho 1992).

Marine Fisheries

Well-documented cases in which market-based incentives in fisheries have stopped or mitigated destruction or degradation of aquatic ecosystems are even rarer than terrestrially based examples. In part, this may be because the high financial stakes often involved in coastal development activities, such as construction of harbors, tourist resorts, and residential developments, readily overwhelm fisheries-generated revenues. In addition, aquatic ecosystems often are not amenable to outright conversion to an alternative use as a forest may be converted to a cornfield. The use of aquatic ecosystems as large waste disposal systems, with the world's oceans the largest, is a less visible alternative use that nevertheless has significant consequences for biodiversity. Such pollution may come from marine-based sources, such as ships and the open-ocean dumping of wastes, or from distant human activities, such as upstream land-use practices. In either case, contaminants can disperse far and wide through aquatic ecosystems. Thus, as long as the oceans remain an inexpensive option for intentional or unintentional disposal of contaminants, such practices in fact represent an alternative use that may compete with fisheries and the maintenance of natural ocean ecosystems. The cultural and amenity values of some coastal ecosystems, and the resulting diverse interest groups and political decisions that affect coastal resources, further obscure the specific role fisheries markets play in coastal zone management.

Examples from the Philippines, salmon fisheries, and international treaties that affect ocean ecosystems illustrate how commercial fisheries may operate as a conservation tool.

The Philippines

G. Hodgson and J. A. Dixon (1988) analyzed the potential economic effects of a logging concession in the watershed of Bacuit Bay on Palawan Island in the Philippines. Bacuit Bay contains important fish resources and coral reefs that support fisheries and tourism industries, which would be negatively affected by logging-induced erosion and increased sediment loads in the bay. The study compared the net present economic values of tourism, fisheries, and logging over a ten-year period for two options: (1) no further logging in the watershed in the Bacuit basin and (2) continued logging for five years, at the end of which period the entire basin would have been logged (table 7-4). In essence, the alternative use in this case was partial conversion of the bay to a sink for absorbing the external environmental effects—increased erosion—of logging. The analysis revealed that the option of no further logging would generate 1.5–2.0 times more revenue than would the option of continued logging. Tourism constituted the primary source of revenue under both options, whereas revenues from fishing and from logging were similar under the continued logging scenario. Inclusion of revenues from the tuna fisheries that exist off the coast, which may be justifiable given the possible dependence of tuna on food resources from the bay, would increase fisheries revenues by more than 50 percent. The results of this study helped lead to a logging ban in the Bacuit watershed and elsewhere on Palawan (F. Romero, pers. comm. 1997).

A recent proposal to construct a large cement plant along the coast of the region of Luzón in the Philippines was also stopped, due in part

Table 7-4. *Net Present Value of Tourism, Fisheries, and Logging in Bacuit Bay, Philippines, 1987–1996*

	Net Present Value (U.S.$, Millions)	
	No Logging	Logging
10% discount rate		
Tourism	25.5	6.3
Fisheries	17.2	9.1
Logging	0	9.8
Total	42.7	25.2
15% discount rate		
Tourism	19.5	5.6
Fisheries	14.1	7.9
Logging	0	8.6
Total	33.6	22.1

Source: Hodgson and Dixon 1988.

to the degrading effects it would have had on coastal fisheries. The fishery-based income in the area, which would have been affected by siltation from quarrying activities and port development, is estimated at $1.37 million annually, compared with the less than $1 million the municipality would have earned from the cement plant (F. Romero, pers. comm., 1997).

Salmon Fisheries

The well-developed and economically important salmon fisheries in North America and Scandinavia appear to be important in maintaining and restoring natural ecosystems. Canada's Fraser River is the world's largest single producer of salmon, with an average annual catch in recent years of more than 8 million fish for the food market, valued at $187 million (Henderson and Healey 1993). This is, however, substantially less than the Fraser River system once produced, as changing land-use practices as well as natural changes in the river basin have reduced salmon runs. As noted in chapter 6, the Canadian government now wishes to double production of the economically most important species, the sockeye salmon. Although some of the proposed management practices, such as construction of spawning channels and fertilization of nursery lakes, create artificial conditions, other efforts appear important in restoring and maintaining natural ecosystems. As part of the rebuilding effort, Canada's Department of Fisheries and Oceans (1986, cited in Henderson and Healey 1993) emphasizes no net loss of fish habitat. One of the biggest socioeconomic trade-offs in the salmon restoration effort will take place in the arid middle and upper portions of the Fraser River basin, where sockeye salmon compete with other users, such as agriculture and forestry, for a limited water supply. Logging, for example, will have to be curtailed to prevent siltation of major sockeye spawning areas (Henderson and Healey 1993). More broadly, the Canadian government's "Green Plan" calls for a cleanup of the Fraser River (*Canada's Green Plan* 1990, cited in Henderson and Healey 1993). Recently, commercial prices for wild-caught salmon have plummeted because of market competition from large increases in the production of farmed salmon. The economic value of salmon sport fishing on Canada's west coast, however, has increased to approximately $420 million. Thus, the total value of the fisheries in the Fraser and other rivers of the region remains high and a significant influence on river and land management (Cerneteg 1997).

Efforts in the western United States to restore salmon runs and their aquatic ecosystems are based in large part on the commercial importance of salmon fisheries there (Lee 1993a). In Norway, a nationwide plan for coastal use includes a list of salmon rivers worthy of protection (Hindar 1992).

Salmon fisheries also act as a market-based incentive for conservation in the open marine realm. United Nations Resolution 44/224, sponsored by Canada and the United States, placed a moratorium on high-seas driftnetting in 1992. The goal of the moratorium is to conserve marine mammals and birds, as well as fish, that are caught in drift nets, thereby helping to maintain a more intact ecosystem in the North Pacific Ocean. A major reason for Canada and the United States to push for the moratorium, however, was to prevent depletion of commercial salmon stocks by high-seas drift net operations (Henderson and Healey 1993). Thus, the commercial importance of salmon, as well as its cultural importance in many North American communities, is contributing to maintenance of the North Pacific ecosystem. These conservation gains, however, are greatly tempered by the reduction in salmon numbers and changes in salmon genetics caused by fishing and hatchery programs (see chapter 6).

International Treaties

The value of marine fisheries has played a key role in the development of marine treaties that aim to prevent the degradation of ocean ecosystems and living resources. The Ministerial Declaration of the Second International Conference on the Protection of the North Sea states that the parties "accept the principle of safeguarding the marine ecosystem of the North Sea by reducing polluting emissions of substances that are persistent, toxic and liable to bioaccumulate . . . especially when there is reason to assume that certain damage or harmful effects on the living resources of the sea are likely to be caused by such substances" (Dickson 1996, p. 4). Similar calls for control and reduction of marine pollutants and other potential causes of marine ecosystem degradation, such as mineral mining, introduction of exotic species, and overfishing, can be found in several other international agreements, such as the Agreement on the Prevention of Marine Pollution from Land Based Sources, the International Convention for the Prevention of Pollution from Ships, and the Antarctic Treaty System (Dickson 1996; Norse 1993). Thus, without the economic as well as human-nutritional values of marine fisheries, the scope and strength of agreements to protect marine ecosystems would be considerably less than they are today, and marine ecosystems would be more subject to degradation by contaminants. Much remains to be done, however, to further strengthen and better implement these agreements.

Summary

Better monitoring of CCU programs and comparative studies of their benefits and costs are needed to clarify how and under what circumstances CCU benefits biodiversity conservation. Nevertheless, if well managed, CCU can be an important tool that works in multiple ways to conserve natural ecosystems and wild species. CCU from highly productive lands and waters may divert pressure to harvest from, and may generate funding for, other ecosystems important for biodiversity conservation. The production of wild species commodities may be ecologically less costly than the production of substitutes, and the harvest of wild species can even be a means to restore or maintain biodiversity and natural ecosystems that have been degraded by human activities. The socioeconomic benefits of CCU can, alone or in tandem with other biodiversity-based values, sometimes outcompete the socioeconomic benefits of alternative uses of land and water that would degrade or destroy a natural ecosystem. Finally, use of wild species and their products may serve to educate and sensitize individuals and society to the importance of biodiversity for human well-being, both material and spiritual.

CCU is often a difficult and complicated tool to wield, however, as various social, economic, and ecological conditions must be in place for it to work. If these conditions are not in place, there is abundant evidence for how readily CCU can lead to overexploitation of wild populations and degradation of natural ecosystems. Much more research and experimentation with alternative approaches, including better evaluation of ongoing programs of CCU purported to be of conservation benefit, are needed to improve our understanding of what the best conditions for sustainability are and how to manage for them. Substantial gains in the sustainability and conservation benefits of CCU could be made now, however, by applying what we already know to improving management and policies.

CHAPTER 8

Managing Commercial Consumptive Use for Biodiversity Conservation

To choose what is best for the near future is easy. To choose what is best for the distant future is also easy. But to choose what is best for both the near and distant futures is a hard task, often internally contradictory, and requiring ethical codes yet to be formulated.

—E. O. Wilson (1984)

Managing CCU so that it benefits, or is at least compatible with, biodiversity conservation requires choices regarding the allocation and use of the earth's natural ecosystems. Those choices will be made and influenced by a diversity of stakeholders. Stakeholders concerned with CCU management, whether resource owners, nonprofit biodiversity conservation groups, government policy makers, or concerned citizens at the ballot box, will make their decisions based on various social, economic, and ecological factors. These factors interact, usually with consequences that require trade-offs among competing stakeholder interests, between economic growth and ecological sustainability, and between current and future benefits. Managing CCU for biodiversity conservation requires an understanding of these interactions and trade-offs.

More broadly, it requires an understanding that the need for trade-offs, for compromises, increases as the world's consumption increases and that more trade-offs imply more biodiversity sacrifices. This is where considerations about the allocation of the earth's natural capital (the trade-offs) must be viewed within the larger context of scale, of global human consumption.

Strengthening the Link between CCU and Biodiversity Conservation

Stakeholders for whom biodiversity is an important goal face four interrelated tasks to strengthen the link between CCU and biodiversity conservation:

1. *Stop overexploitation and restore depleted populations and degraded ecosystems*. Overexploitation has two consequences for biodiversity. First, depletion of a population, particularly of a keystone species, may in itself lead to significant ecosystem change and biodiversity loss. Second, even though overexploitation of a commercially valuable species may provide short-term benefits by increasing profits and thus protecting the ecosystem from alternative uses of its land or water, it eventually leads to declines in harvest levels and revenues. Unless the depleted population and its commercial value can be quickly restored, the ecosystem is rendered more vulnerable because alternative land or water uses may become socioeconomically more attractive.

2. *Avoid unnecessary specialization in commodity production from natural ecosystems*. Simplification of the world's remaining natural ecosystems through specialization in commodity production, whether of domesticated or wild species, should be avoided, particularly in ecosystems of significant biodiversity value. Economic specialization that requires ecosystem simplification should focus on areas that are already highly altered or degraded. How to produce commodities while maintaining native biodiversity remains a significant research and management challenge and will vary widely among ecosystems. Related to tasks (3) and (4) that follow is the question of what level of specialization and resulting sacrifice of biodiversity may be appropriate or necessary within a given area, whether to make it competitive with alternative uses or to ease harvest pressures on sites of high conservation priority.

3. *Wisely allocate land and water uses*. Greater attention must be given to allocation of the intensity of use of ecosystems, from designation as protected areas to maintenance as natural or seminatural areas subject to various types and intensities of CCU to conversion to domesticated production. This allocation will depend in large part on the relative socioeconomic competitiveness of alter-

native uses of the land or water and will be periodically challenged and reevaluated as values and markets change. How natural ecosystems fare in this allocation of land and water uses will depend on our ability to incorporate other biodiversity values into the balance sheet (see task 4).

4. *Incorporate noncommodity values of biodiversity into decision making.* The more visible and tangible are the noncommodity values of biodiversity, both monetary and nonmonetary, the greater will be their competitiveness in decisions about land and water use. To the extent that other biodiversity-based values (e.g., nature tourism, ecosystem services, existence values, etc.) can be added to (or, in some cases, replace) CCU values, natural ecosystems will stand a better chance of outcompeting alternative uses, and pressures to simplify ecosystems for commodity production will be lessened.

Both overharvesting and overspecialization can erode biodiversity. However, high levels of offtake and intensive management interventions may be necessary for socioeconomic sustainability and competitiveness with alternative uses. Thus, ecological sustainability and socioeconomic sustainability are inversely related regarding the direct effects of increased offtake and specialization (figure 8-1). However, where CCU is the primary source of socioeconomic benefits for the resource owner and where alternative uses of land or water are a threat, ecological sustainability may be indirectly enhanced by the greater rev-

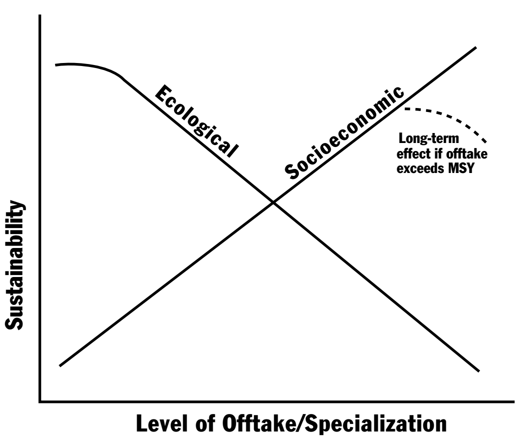

Figure 8-1. *Direct effect of increasing levels of offtake and production specialization on ecological and socioeconomic sustainability where CCU is the primary source of revenue. Not considered here is the benefit increased offtake and specialization might provide for ecological sustainability by outcompeting alternative uses of land and water.*

enues resulting from higher offtake (up to a point) and greater specialization. In some cases, even overharvesting may be a strategy for maintaining the short- and medium-term socioeconomic sustainability of a natural ecosystem.

Where complete conversion of natural ecosystems to domesticated production is not an imminent threat, as in vast boreal forests or the open ocean, increasing the level of harvest or specialization to increase profits is questionable as a conservation tool. Although enhanced profitability of CCU may enable managers to invest more in research and sound management practices, such benefits quickly diminish when increased profits are sought by raising harvest levels. Greater intensities of use and habitat manipulation—for example, maximizing the sustainable harvest of timber in boreal forests or of bluefin tuna (*Thunnus thynnus*)—will require greater investments in research and management to ensure sustainability. Although there may be socioeconomic benefits, there is no net gain for biodiversity conservation in this scenario, only the increased risk of overharvesting and ecological degradation.

Averting Overexploitation

Multiple factors, which vary according to socioeconomic and ecological conditions, can lead to overexploitation (figure 8-2). The following review, though not an exhaustive description of the issues that must be addressed, highlights the most important ones.

Securing Resource Rights and Avoiding Open-Access Problems

Open-access regimes, combined with high prices for CCU products, stands as the single biggest proximate cause of overexploitation. Where resources are difficult to protect despite clearly defined ownership, high prices can result in uncontrollable clandestine harvesting and thus de facto open-access regimes. There are two principal solutions to this: (1) better define and enforce ownership and use rights where feasible; (2) establish and enforce regulations that control harvest levels. Sound government leadership is required in both instances. Ownership and use rights can most readily be assigned and enforced for terrestrial, freshwater, and near coastal ecosystems.

However, the lack of clear resource ownership in some regions, such

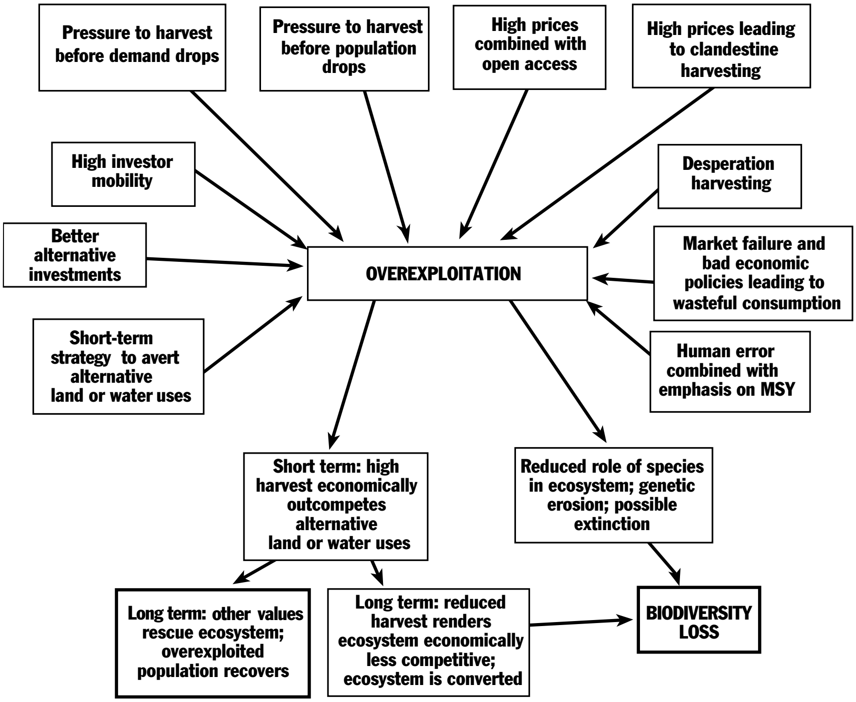

Figure 8-2. *Causes and effects of overexploitation of wild species.*

as many areas in the Amazon and Congo basins, generally reflects the low monetary returns from the regions' natural resources (Bromley 1991; Swanson 1994) and the failure of governments to recognize the socioeconomic benefits natural resources provide to local people through subsistence use and informal markets. Governments thus do not assign and enforce ownership rights, including their own. A vicious cycle results whereby open-access regimes lead to overexploitation of natural resources and thus loss of their socioeconomic value, thereby further eroding any incentives for governments to invest in these regions.

Who receives ownership and use rights and how they are conveyed are crucial questions whose answers depend on various socioeconomic, ecological, and cultural factors. In some cases, the best solution will be to reestablish traditional rights of local communities or to promote private ownership; in others, sustainability and biodiversity conservation may be best served by government ownership. The principal challenge here often centers on the mismatch of social and ecological scales. Ecosystem processes usually operate at scales much larger than those of reasonable units of proprietorship. Regardless of how ownership is defined, and because of this mismatch of scale and the negative externalities that result, some level of government oversight will often be necessary to ensure sustainability and to represent broader societal interests in biodiversity. Where possible, however, government must adopt a comanagement approach with local communities and resource users. This approach should involve a search for synergies between the knowledge generated by Western science and the experiential knowledge of resource users and traditional communities.

Assigning and enforcing ownership and use rights for highly migratory species and for fisheries beyond the nearshore environment is much more difficult, if not impossible. Although mechanisms to privatize fisheries resources, such as individual transferable quotas (ITQs), may be effective in some circumstances, fisheries managers and governments will have to develop more effective means of preventing overfishing if the world's fisheries are to recover and be sustainably managed. This should include less reliance on quota systems, greater reliance on adaptive management, and stronger enforcement programs. Beyond the 200-mile limit of the exclusive economic zone (EEZ), sustainability will require a more concerted effort to give teeth to international fishing agreements that regulate harvest and trade in overexploited species.

Promoting Long-Term Stewardship and Discouraging Resource Mining

An economically rational resource owner may liquidate a high-value, slow-growing wild species commodity in order to reinvest the proceeds into something earning a higher rate of return. Such activities should pose less of a problem where resource owners have a long and stable history in an area and thus fidelity to and dependence on the long-term sustainability of the resources under management. These conditions may apply to government ownership where government agencies have the resources, incentives, and public support for sound management. Where resource ownership is fluid or alternative investments promise a higher yield than can be realized through sustainable resource use, government regulation and enforcement may be necessary to prevent resource mining. This is particularly true for individuals and corporations with a history of mobile and changing investments. Short-term concessions for resource use, particularly for logging, also preclude long-term stewardship and invite resource mining and therefore should be avoided.

Economic specialization, however, is encouraged by secure ownership; thus, in areas of conservation priority, other measures may be necessary to avert ecosystem simplification by intensive management. In addition, major increases in the price received for a wild species commodity, as often results in the conversion from local to international markets, can disrupt the social stability and traditional resource management regimes of local communities.

At the international level, CITES, the Convention on Biological Diversity, and other agreements involving trade in wild species commodities must be strengthened to prevent resource mining.

Averting Desperation Harvesting

Desperate circumstances lead to short-term solutions. In CCU, this generally means overharvesting. Desperation comes in many forms—a poverty-stricken forest-dwelling family living from day to day, a debt-ridden fisher struggling to make monthly payments on a new boat, a government weakened by its inability to repay its foreign debt. All are versions of the high-discount-rate syndrome. Where wild species commodities can help resolve an economic crisis, rational decisions lead to the same effect: better to harvest the resource now, thereby ensuring survival over the short term, and deal with the consequences of resource depletion later. The conservation strategy must first be to avert scenarios that expose resource owners and harvesters to crisis-creating situations in which overexploitation is the only way out. Thus, for example, owners of fishing fleets should be discouraged from overcapitalizing during good fishing years because this leads to desperate conditions during poor fishing years. Government subsidies that keep such industries afloat during desperate, low-production years should be avoided. In such cases, government investments should go toward buyouts to reduce the size of the fishing fleet and toward helping fishers move into other employment sectors. The same problems are widespread in the timber industry.

There is, however, a role for subsidies from government or other sectors to help meet socioeconomic needs during the recovery period of an overexploited resource. Without such assistance, whether in the fishing communities of coastal Canada or the forest communities of

Amazonia, communities may suffer and disintegrate from economic stress, and barring strict enforcement, desperation harvesting of the dwindling resource will continue.

We must also discourage the development of international monetary policies that require governments and their rural citizens to overexploit natural resources in order to meet both national financial obligations and basic human needs. Structural adjustment loans, for example, should not require depletion of a country's living natural resources to finance macroeconomic imbalances (Reed 1996).

Dealing with Ecological, Economic, and Social Uncertainties

Those who depend on the harvest of living resources for their livelihood are subject to the uncertainties of both the marketplace and the environment. Demand and prices may be strong one year and weak the next. Populations of wild species that sustain a large harvest one year may produce no surplus or may have migrated elsewhere the following. If a price or population is viewed as unusually high, the harvester may choose to mine the resource and reap the profits, not because of uncertainty but because of the certainty that the price or population will eventually go back down. Where wide fluctuations in markets and production occur, it may be important not only to provide incentives and regulations that prevent overharvesting during periods of high prices, but also to encourage the investment of earnings from high-profit years as a buffer against years of low prices or low harvest levels. The frequent alternative use of such profits, with generally negative consequences for ecological sustainability, is greater capitalization either in harvesting capability (e.g., expanded fishing fleets) or in management interventions to increase and stabilize productivity (e.g., fish hatcheries).

Ecological uncertainty also leads to miscalculation of harvestable quotas. As discussed later, new approaches in adaptive management and the application of the precautionary principle are needed for better monitoring and more prudent harvesting of wild populations.

Uncertainty about tenure, harvest rights, and benefits from a resource also encourage resource mining. Both political instability and major swings in government regulations affecting resource ownership create disincentives for sustainability. Clarification and stabilization of private tenure and regulations that govern it are among the most important tasks facing conservationists. Resource rights to high-value wild species commodities are often challenged. High prices encourage risk taking by clandestine harvesters and attract politically savvy entrepreneurs who, whether through legitimate or corrupt political avenues, pressure for policy changes that may infringe on the rights of traditional resource harvesters. Thus, when new markets emerge for wild species commodities, close attention must be paid to the problems of clandestine harvesting, illegal markets, and pernicious policy changes.

Getting Market Signals Right and Reducing Wasteful Consumption

Overexploitation of some wild species commodities can be reduced by ensuring that the price paid by consumers reflects the full environmental costs, including the negative externalities, of resource use. Correcting the market's failure to cover environmental costs will increase the price consumers pay for the commodity and thereby reduce wasteful consumption and, ultimately, the quantity of the commodity demanded (Pearce 1995). Governments and international financial institutions have developed economic policies that send the wrong economic signals and favor overexploitation of natural resources. Changing existing policies that undermine sustainability and preventing the development of new ones are some of the most important policy tasks facing conservationists. Mechanisms must also be sought to ensure that the full environmental costs of resource use are reflected in resource prices paid by consumers. This can be done both voluntarily, through green marketing, and through taxes designed to address the negative environmental externalities of CCU. Similarly, governments should collect royalties and fees for use of government-owned lands that reflect the true value of the resource, including the externalities of its use, to ensure that resources are not undervalued and to encourage their efficient use.

Large subsidies for the timber and fisheries industries must be curtailed so that the true value of a resource is reflected in the price consumers pay. Removal of subsidies will tend to drive prices higher and lower consumption and harvest rates. It should also generate greater revenues per unit of harvest, enabling more funds to be directed toward sustainable management.

National accounting systems that address the depletion of natural capital are needed to provide a meaningful yardstick for making policy decisions about resource use and for measuring the true effects of macroeconomic policies on resource use. Environmental accounting systems will change, for example, the way we evaluate the effects of structural adjustment programs and free trade agreements. Although some elements of global trade may benefit biodiversity conservation, current global trade agreements carry significant risks for the ecological sustainability of CCU. More detailed analyses and substantial reform are needed so that trade agreements, particularly WTO, encourage nations and industries to internalize the environmental costs of CCU, to ensure that production specialization does not compromise biodiversity, and to avoid any propensity transnational corporations may have for a "mine-and-switch" investment strategy in living natural resources.

Harnessing Consumer Power

The discriminating consumer can provide a strong incentive for resource producers to practice sustainable management. This requires that consumers care enough about sustainability to make discriminating choices in their buying. Such choices will be made only if consumers receive reliable information about the product and have confidence in the veracity of the information. To the extent that ecologically sustainable harvest practices and the product tracking required of green labeling involve greater costs, consumers must be prepared to pay more for green-labeled products. In a broader sense, green labeling provides a market mechanism for consumers to pay for a wide array of biodiversity values they hold for the ecosystem from which the wild species commodity is harvested. Thus, the buyer of a can of sustainably harvested Brazil nuts may be paying not only for the sustainability of the product but also for other biodiversity values he or she derives from Amazonian forests, such as option value or existence value (Swanson 1994).

Green labeling requires that the producer be rewarded for producing an ecologically sustainable product. This provides both the financial incentive to develop sustainable practices and the financial capability to cover the additional costs of sustainable practices as opposed to overexploitation. Where CCU is ecologically sustainable and crucial for averting alternative uses of land or water, it may be important to increase consumer demand for the product. Certification systems must be developed to distinguish good management from bad and to give consumers an understandable and reliable method for buying ecologically sustainable CCU products.

Increased consumer awareness can also reduce demand for wild species commodities. Whether in the illegal drug trade or the illegal wildlife trade, experience indicates that where consumer demand remains high, any attempt to reduce or stop trade through trade bans and sanctions is an expensive and usually futile exercise. Working to reduce consumer demand or to redirect it toward more sustainable and conservation-benefiting forms of CCU may often be a more cost-effective long-term management technique.

Managing Human Error: The Precautionary Principle and Adaptive Management

The unpredictability of natural systems, human ignorance and miscalculation, and imprecise harvesting techniques can, despite good intentions, result in harvest levels that exceed the sustainable levels set by management. The probability of this occurring is greatest where maximum sustained yield (MSY) is sought. Under MSY, a slight miscalculation can readily lead to overharvesting, a decline in the population to below the MSY level, and ecosystem degradation where keystone species are involved.

The precautionary principle has been proposed as a policy tool to limit the degree to which goals of sustainable offtake or ecological sustainability are put at risk by harvest and management. Much of the impetus for the precautionary principle was the increasing recognition of the prominence of uncertainty in human-environment interactions and of our inability to accurately predict the environmental consequences of human activities (Dovers and Handmer 1995). Similarly, science's inability to correctly model and predict how populations and ecosystems will respond to management led to the development of adaptive management as a way to combine scientific methods and management interventions. Adaptive management is a process in which mistakes are expected and are to be learned from, but it is also precautionary in that irreversible mistakes (e.g., extirpation of a stock by overharvesting) are to be avoided (Lee 1993a; Walters and Holling 1990).

A predecessor to the precautionary principle was the "safe minimum standard" (Ciriacy-Wantrup 1952), which primarily concerned averting irreversible or difficult-to-reverse consequences of unwise resource management, such as destruction of breeding stock or its habitat or contamination of groundwater. Initial development of the precautionary principle, however, has largely involved minimizing environmental risks posed by pollution. This was the intent when the precautionary principle was first internationally adopted, at the 1987 Second International Conference of the Protection of the North Sea: "In order to protect the North Sea from possibly damaging effects of the most dangerous substances, a precautionary approach is necessary which may require action to control inputs of such substances even before a causal link has been established by absolutely clear scientific evidence" (Cameron and Abouchar 1991, p. 5).

Application of the precautionary approach has recently gained increasing attention regarding the harvest and management of wild species. For example, in 1991, the Commission for the Conservation of Antarctic Marine Living Resources adopted measures requiring that members proposing to develop new fisheries notify the commission in advance and provide an assessment of potential effects of the new fisheries on dependent and associated species (Mangel et al. 1996). CITES has also adopted as policy a precautionary approach regarding the harvest of wild species for international trade; in particular the scientific authority of the permit-granting country is required to affirm that the "proposed export will not be detrimental to the survival of the species" (Favre 1995, p. 338). In addition, groups generally aligned with more strict animal rights causes have called for the precautionary principle to be applied to CCU such that no use is allowed unless it has been *proved* to be safe (Dickson 1996).

As these examples illustrate, a central element of the precautionary principle is a shifting of the burden of proof. For example, in proposing a set of principles for the management of wild species (but never specifically referring to the precautionary principle), M. Mangel and coworkers (1996, p. 345) state that "It is generally appropriate to assume that, until proven otherwise, use of wild living resources will have unacceptable effects on both the target resource and on other components of the ecosystem. This changes the working hypothesis from 'use of the resource will have no effect' to 'use of the resource will have serious effects.' It also changes the burden of proof from those responsible for conserving the resource to those who want to use the resource."

This requires a fundamental change in the way science traditionally approaches a problem. Conventional science is averse to accepting a hypothesis as true when it is in fact false (referred to as a type I error); scientific rigor and careers are better served by rejecting a hypothesis that is in fact true (type II error). The problem with this approach has been demonstrated in fisheries in which reductions in harvest have been delayed because scientists hesitated to declare that something was true—the populations were declining—until they obtained more evidence (Lee 1993a; Peterman 1990). Where there is concern about effects on populations and ecosystems, whether from the potential effects of harvesting or of pollution, science will better serve ecological sustainability by being more concerned with avoiding type II errors. The risks are too high to do otherwise. Rational people do not play Russian roulette because, even though the odds (one in six) are strongly against the bullet being in the chamber, the cost of a type II error (concluding that the bullet is not in the chamber when in fact it is) is rather severe.

Both the precautionary principle and adaptive management have their origin in dealing with large-scale activities and environmental effects such as industrial pollution, large commercial fisheries, and industrial-level forestry. Those who might harm the environment by their activities are seen as having the resources and ability to make significant adjustments in their activities (e.g., changing harvest levels or harvest technology), and in general these enterprises are not viewed as providing benefits for biodiversity conservation. Further, such large-scale CCU programs could attract financial and technical resources, from both industry and government, to develop the monitoring and other investments adaptive management requires. However, a new set of issues and critics (Dickson 1996; Freese 1996; Morrill 1996; Warren 1993) has emerged to challenge the precautionary principle, particularly where it is applied to small-scale users in developing countries and where the resource use might be an important socioeconomic justification for conserving natural ecosystems.

Greater uncertainty about the ecological consequences of a given harvest level means that the potential margin of error is greater and therefore, under the precautionary principle, that harvest levels should be set more conservatively. More research and information, to the extent that they reduce uncertainty and the potential margin of error, allow higher harvest rates. This corollary of the precautionary principle poses problems regarding socioeconomic equity between North and South, haves and have-nots. For example, resource managers in tropical biomes often have little money and technological capability compared with their temperate-zone counterparts and manage multiple species in often poorly understood ecosystems. They therefore seldom attain the level of understanding of their target species and associated ecosystems held by their peers in the industrialized world. The same disparities arise regarding the investments in research and monitoring large corporations can make compared with the investments individuals or capital-poor community-based managers can make.

The resulting issue is whether it is practical and ethical to require, say, forest extractivists or artisanal fishers to forgo a large part of their potential harvest to keep the risk of overharvesting to some minimum level. At the national scale, as in marine fisheries, the research and monitoring practiced by industrialized nations far exceeds that in developing nations, and thus under the precautionary approach the former's marine fisheries quotas will on average be higher. It is understandable if some view the precautionary principle as another industrialized-world scheme for the rich to get richer while the poor get poorer. The precautionary principle has also been criticized for not addressing *why* a use is taking place and how its curtailment may affect those dependent on the resource and for largely removing decision making from the user—the person or group most directly affected—and giving it to a bureaucracy, which does not have to bear the consequences of the decision (Dickson 1996) and is often captured by special interest groups.

Misapplication of the precautionary principle could also undermine biodiversity conservation efforts. The need to avert the conversion of natural ecosystems to alternative uses of land or water may place a short-term premium on maintaining socioeconomic sustainability, which may require placing ecological sustainability, including sustainability of offtake, at greater risk.

These considerations require that the precautionary principle be applied with a sensitivity to different levels and measures of sustainability and risk. In some circumstances, though the risk of overharvesting a population may be significant and thus not precautionary at the population level, the socioeconomic sustainability it buys over the short term may reflect a highly precautionary strategy for maintaining the natural ecosystem it inhabits. Flexibility in terms of what is at risk—short-term levels of offtake, a population, the survival of a species, the loss of an ecosystem—needs to be considered. Irreversible harm to a population or ecosystem must weigh much more heavily than reversible consequences. Indeed, taking risks with reversible consequences enhances our ability to learn from management. As D. Bodanksy (1991, p. 43) notes, "The precautionary principle seems to suggest that the choice is between risk and caution, but often the choice is between one risk and another."

To apply a stronger precautionary approach at all levels of risk, with both biodiversity conservation and socioeconomic equity as goals, will require a much greater North-to-South flow of investments in research, monitoring, and management. Further, if it is to be applied under conditions of impoverishment, where livelihoods depend directly on the use of wild species, compensation may be required for local stakeholders who forgo higher harvests to satisfy the desire of conservation stakeholders for reduced ecological risk.

Given the limited financial and technical resources in most of the world, new, low-cost methods for designing and managing sustainable harvest programs, such as the harvest refugia concept (see chapter 5), development of community-based approaches to monitoring and research, and greater reliance on local knowledge and traditions, must be pursued. At the other extreme, as suggested by C. J. Walters and C. S. Holling (1990), new technologies such as satellite image analysis may also offer less costly, though also less precise, methods for monitoring and experimenting, whether in managing forests or big game. These are the challenges for a new adaptive management regime tailored to the realities of most of the world.

Managing for Diversification versus Specialization

Economic specialization must be viewed at two levels with regard to ecological sustainability. At the global level, increased yields of wild species commodities from highly productive sites can relieve pressure to harvest from, and can generate funding for, sites that should be protected for their biodiversity value. At the local level, within a given management site, economic specialization inevitably compromises biodiversity values. How can this loss of biodiversity be minimized within a given site?

Any of five ecological, socioeconomic, and cultural factors may convince a resource owner to maintain biodiversity rather than specialize in production (figure 8-3): (1) in many ecosystems, loss of biodiversity or natural processes can, over the long run, erode productivity (Holling et al. 1995); (2) biodiversity provides resilience to perturbations and adaptiveness to long-term change (Holling et al. 1995); (3) biodiversity maintains options, providing a hedge against changing human values and the possibilities that today's weed will be tomorrow's miracle plant (Burton et al. 1992) and that new markets may emerge to pay for non-CCU values of biodiversity (Pearce and Moran 1994); (4) a biologically diverse landscape provides security and stability to communities in

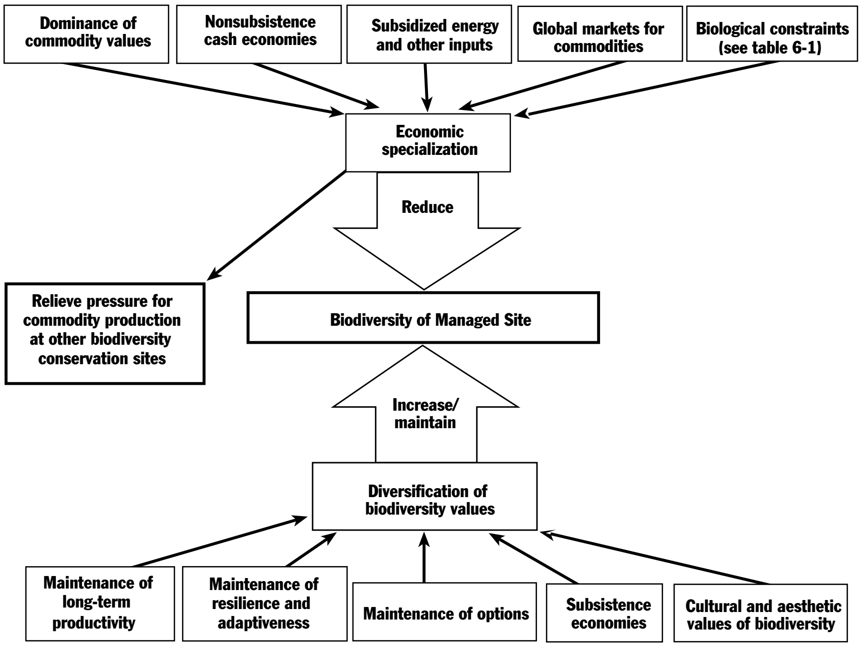

Figure 8-3. *Factors that favor economic specialization or diversification in management of natural ecosystems and their effects on biodiversity.*

subsistence economies (Gadgil, Berkes, and Folke 1993); (5) for a number of reasons, including cultural and aesthetic values, natural ecosystems and biodiversity can hold intrinsic value for the owner. All these factors can be effective tools both in maintaining biodiversity where commodity specialization is possible and in justifying the maintenance of natural ecosystems in the face of competing uses of land or water.

Productivity

Even in cases in which the production of one or a few wild species commodities is the primary goal, there is evidence that maintenance of biodiversity and natural ecosystem processes can be important in ensuring the long-term productivity of the target population (see chapter 5). Those components of native biodiversity and ecological processes that may be crucial for long-term productivity are difficult to predict, and the degree to which they must be maintained varies greatly among ecosystems. Pesticides, irrigation, and fertilizers are often used to compensate for natural controls and processes that are lost as ecosystems are simplified for commodity production. If the full environmental costs of these inputs had to be paid by the producer and consumer in the form of higher prices, greater emphasis might be given to maintaining biodiversity and natural ecological processes of importance to productivity.

A producer's concern for long-term declines in productivity will be offset by the discount rate he or she applies to future revenues. A loss in future revenues due to a slow decline in productivity from, say, a forest may be more than offset by the higher yields and revenues obtained initially by homogenizing stand structure and depleting forest nutrients through clear-cutting. How to minimize the effects of such discounting practices remains a major challenge to sustainability.

Resilience and Adaptiveness

Closely linked to the previous point is the importance of biodiversity and natural ecological processes in maintaining resilience of ecosystems and adaptiveness of both ecosystems and individual populations. Commodity production in many ecosystems can benefit when this concept is incorporated into management practices. The potential applications and benefits are broad. For example, biodiversity and natural ecosystem dynamics in grasslands are often crucial in maintaining both short- and long-term productivity through periods of drought, and maintenance of genetic diversity in fish stocks provides a buffer against the production vagaries of individual stocks and enables stocks to adapt to both short- and long-term changes in their ecosystems. Thus, greater attention to resilience and adaptiveness can counter the tendency to simplify ecosystems for commodity production.

Maintenance of Options

Resource managers who wish to maintain economic options for use of their land or water will be inclined to maintain greater biodiversity. Individuals who maintain a large portfolio of potential biodiversity products will be better prepared to capitalize on new markets that emerge for one or more of those products. Recent rapid and significant changes in consumer preferences and prices paid for various timber products and luxury items such as furs are evidence of how quickly markets can change. Moreover, resource owners who specialize in commodity production to the detriment of biodiversity may also be reducing current and future revenue options from noncommodity values of their land or water. Ecosystem valuations (see chapter 3) demonstrate that the sum of recreational, functional, and other values of biodiversity within a given ecosystem is often several times greater than the ecosystem's commodity values. New and expanding markets—nature tourism, conservation easements, green labeling—are developing to pay landowners for these values. Thus, prudent resource owners, particularly owners of sites with great biodiversity value, will avoid ecosystem simplification and biodiversity loss lest they foreclose current and future revenue-generating options. Moreover, choices made now regarding ecosystem management can have long-term consequences for maintaining options. An even-aged stand of forest developed for timber production may require more than a century to be managed back to a natural-looking, more biologically diverse state that would be of value for nature tourism or for a conservation easement purchased to maintain its plant and animal diversity. Where market-based incentives fall short, as they often do, governments must develop tax incentives and other policy mechanisms to encourage ecologically sustainable practices.

Security and Stability

People who live in stable communities with largely subsistence economies, where options for purchasing goods produced elsewhere are limited, generally have a greater need to maintain biodiversity than do those in cash-based economies. For such people, biodiversity confers security and stability, since there are no substitutes for the essential and diverse services and products nearby natural ecosystems provide. Because resource tenure rights of subsistence-based communities often are not legally well defined or enforced, care must be exercised to avoid well-financed and politically influential commodity production schemes that may disrupt traditional tenure rights and erode the diverse products and services provided by natural ecosystems. Even communities with significant cash economies may depend on or prefer to use a diversity of natural products and services from nearby ecosystems. Subsistence uses and informal markets for many wild species commodities, such as nontimber forest products and bush meat, are generally not recognized in national accounts, and thus their socioeconomic importance is at best weakly considered in development planning that affects natural ecosystems. Local communities' needs for the diverse products and services of natural ecosystems is a potentially important argument for maintaining biodiversity in areas proposed for commodity production schemes, such as plantation forestry and coastal shrimp farming.

Cultural and Aesthetic Values

Resource owners do not reduce all resource production decisions to the bottom line of the ledger sheet. Various nonmonetary factors often weigh significantly in their decisions. Some manage their land or water to maintain a lifestyle of importance to them, which they want to pass on to their heirs. Others value wild species and ecosystems for aesthetic, educational, or scientific reasons, or they feel a moral obligation toward conserving biodiversity. Whatever the basis, the more people know about wild species and natural ecosystems, the greater is their appreciation for them and the more prone they are to preserve them regardless of monetary considerations. Thus, educating resource producers about biodiversity, its societal benefits, and their role in conserving it may mitigate incentives to homogenize ecosystems for commodity production.

A Conceptual Economic and Ecological Framework for CCU

The relation between CCU and biodiversity conservation largely involves the interaction of economic and ecological factors, which change and affect each other across the spectrum of options for resource use and management. Decision making about CCU management and, more broadly, about allocation of land and water to different types and intensities of resource use and management, may therefore be facilitated by a conceptual framework that integrates economic and ecological factors.

Effects of Intensities of Use on Key Economic and Ecological Variables

The two graphs in figure 8-4 illustrate the effects on key ecological and economic variables of different intensities of use and management of both wild and domesticated species for market commodities on a given

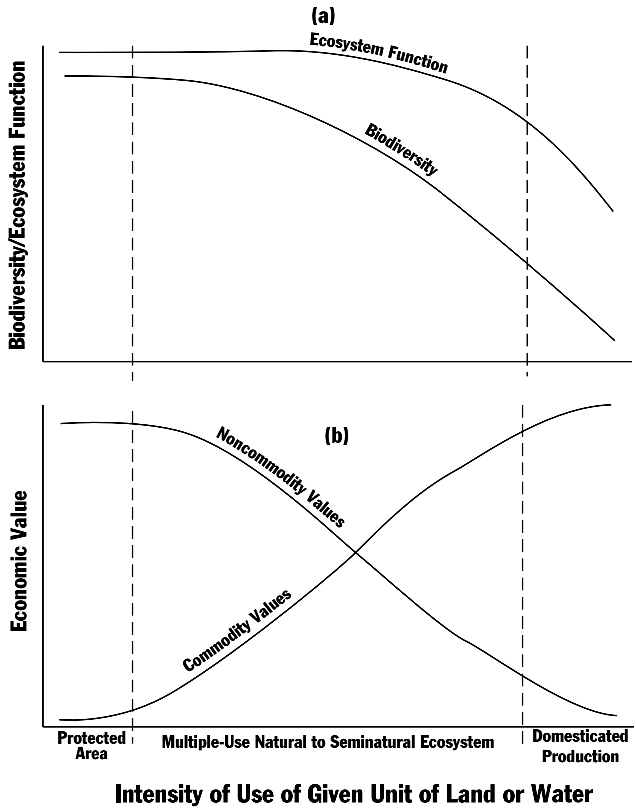

Figure 8-4. *Effects of increasing intensity of use of a given unit of land or water on (a) biodiversity and ecosystem function; (b) commodity and noncommodity values of an ecosystem.*

unit of land or water. In this framework, a given unit of land or water may be managed as (1) a fully protected area where there is no or minimal consumptive use of wild species; (2) a multiple-use natural or seminatural ecosystem where management may range from moderate levels of offtake to high offtake and intensive habitat management; or (3) a monoculture or its ecological equivalent, whether a cornfield, single-species forest plantation, or shrimp farm. These three levels of use correspond to the triad approach of land-use allocation proposed by M. L. Hunter Jr. and A. Calhoun (1996). Distinctions between the three levels of land use are generally more blurred than is indicated by the vertical dashed lines in the figure. Low levels of CCU are permitted in many protected areas, for example, and agroforestry sites and shaded plantations of coffee and cacao may fall in a gray area between seminatural ecosystems and domesticated monocultures.

Natural or seminatural ecosystems that are available for multiple uses represent well more than half of the terrestrial realm and nearly all of the marine realm (see chapter 1). Of concern here are those factors that influence whether a given ecosystem or area will be managed intensively for commodity production, thus falling on the right side of the graph, or not intensively (in terms of harvest levels and manipulation of the ecosystem) thus falling on the left side of the graph. What are the economic and ecological consequences of these choices?

Figure 8-4(a) shows that biodiversity within a given area will usually be greatest under no or low intensities of human use and intervention. (The exact shape of the curves is not important, but the overall patterns are.) Replacing "biodiversity" with "native ecosystem integrity" or some similar concept of naturalness would yield a similar relationship. Eventually, however, increasingly intensive management for commodity production from a natural ecosystem begins to erode its biodiversity. Biodiversity continues to decline with greater intensities of use until the point of full transition to monocrop production systems, where biodiversity reaches its lowest level. Meanwhile, ecological functions, such as nutrient cycling, efficiency of energy capture, and productivity, decline more slowly than biodiversity, at least initially, because of ecological redundancy among species within an ecosystem. It is such redundancies, particularly with regard to factors affecting productivity, that enable commodity managers to depress or eliminate populations of some species and alter ecosystem structure to direct more nutrients, energy, and space toward production of the commodity species. To the extent those functions important for productivity of the commodity species begin to decline with increasing intensities of use, they are replaced, as indicated earlier, by generally energy-intensive inputs such as irrigation, fertilizers, and pesticides.

Figure 8-4(b) illustrates the trade-off between commodity and noncommodity economic values as a function of intensity of use. The commodity value of an ecosystem increases at the expense of noncommodity values under increasingly intensive use. Because commodities are often the only tangible economic value of a given unit of land or water for the resource owner, the commodity value is often the same as the total market value of the ecosystem. Obvious exceptions to this are popular parks and reserves that earn substantial revenues from nature tourism. The second curve in figure 8-4(b) shows the value of the diverse and generally less visible noncommodity values of a natural ecosystem and of biodiversity, for which true markets are poorly developed. In economic terms, the noncommodity-value curve could be considered the sum of positive externalities provided by the ecosystem (e.g., watershed protection, biodiversity values, climate regulation) and the negative externalities caused by human use (e.g., increased downstream siltation, reduced biodiversity, climate change). At low intensities of use, that sum will be positive, but as use intensifies, the sum may become negative (Freese 1996).

The sum of the commodity and noncommodity values at any given intensity of use theoretically represents the total economic value of a unit of land or water. In terms of benefits to society, the point of optimal economic use of that land is defined by the point at which the sum is greatest. Thus, for natural ecosystems of high biodiversity value, the challenge for conservationists is to keep that optimal level of use as far to the left of the graph as possible—to prevent the slide down the slippery slope of economic specialization. At the same time, conservationists must identify, from both a biodiversity conservation perspective and an economic development perspective, those ecosystems most suitable for economic specialization and commodity production.

Considerations in Allocating Land and Water Use

Biodiversity conservation will not be served, however, if conservationists attempt to maintain management of every unit of land or water toward the left side of the graph. If we assume no decrease in demand for commodities from natural ecosystems and domesticated monocultures (to the contrary, it will surely increase as human population and consumption increase), it will be important for some areas to be devoted to intensive production of commodities so that other areas can be devoted to biodiversity conservation (Hunter and Calhoun 1995; Lugo and Brown 1995; Salwasser et al. 1996). Although A. E. Lugo and S. Brown (1996, p. 289) conclude that "The time has come to assign a use to every square kilometer on the planet," significant decisions about land and water allocation have already been made on a global scale. Almost all of the earth's most fertile and productive lands have been under cultivation for hundreds, and in some cases thousands, of years (Huston 1993). Very little of this land is in protected area status or harbors anything resembling its native biodiversity. The remaining land is, on average, significantly less productive for monocrop agriculture or livestock production. Nevertheless, much of this less productive land, such as semiarid steppes and grasslands and significant areas of both temperate and tropical forests, has also been converted to intensive or semi-intensive production of livestock, agricultural crops, and timber. The marginal productivity of these lands, the loss of government subsidies for agricultural production in some cases, and the increasingly important noncommodity values of their biodiversity create the potential for these lands to be converted back to more natural, biologically diverse states. Until the end of the nineteenth century, for example, Scandinavian forests were being lost to livestock grazing, agriculture, and unsustainable forest management practices. Strong timber markets in the twentieth century helped fuel extensive plantation-type reforestation of these lands. More recently, sagging agricultural profits and stronger public interest in the biodiversity values of forests have stimulated additional reforestation, with legislatively mandated trends toward more natural forest cover (Fritzbøger and Søndergaard 1995; Hytönen and Blöndal 1995).

However, as suggested in chapter 7, the best conservation strategy for some degraded, formerly forested lands may be to convert them to highly productive plantation forests in order to concentrate wood production within a limited percentage of the earth's forest estate. Although native species are ecologically preferable, exotic species that provide higher yields may be used where they do not threaten to colonize natural ecosystems. To what extent specialized production of other wild (or semiwild) species commodities can relieve pressure on more vulnerable and high-conservation-priority ecosystems remains to be seen. Aquaculture would seem to offer some limited opportunities where total confinement is possible or where native species are used in more open systems, but the risks of escaped stocks, genetic alteration, and competitive displacement of and predation on native species present significant problems for this approach. Further specialization in the production of game in wetlands and terrestrial habitats, though it may be important in isolated circumstances to produce revenues for habitat conservation, generally offers more risks than benefits for biodiversity and should be avoided.

Areas of climatic and geophysical extremes, including areas with nutrient-poor soils (e.g., alpine, tundra and high boreal regions, deserts, and many tropical rain forests), are least amenable to any sustainable system of commodity production, and large expanses of most of these regions remain more or less intact. Such areas understandably contain a disproportionate share of the earth's protected area coverage because their protection does not have to compete with commodity uses of the land (Huston 1993).

Species diversity is a unimodal function of productivity in many, if not most, terrestrial and marine regions of the earth (figure 8-5), with the exact form of the hump-shaped relationship varying depending on the taxonomic group and region. Although there is considerable debate about the strength of this relationship and the underlying causes (Huston 1993; Rosenzweig and Abramsky 1993; Tilman and Pacala 1993; Wright, Currie, and Maurer 1993), it suggests a useful way to examine the trade-offs between commodity production and biodiversity conservation. If species diversity is usually relatively low on highly productive sites, then the conversion of the earth's most productive lands to monocrop production has, to date, probably had less of an effect on species diversity than if the same level of production had been achieved randomly over the landscape. However, a growing world population and increasing per capita consumption will create inexorably rising demands for the most basic commodities, such as food and heating and

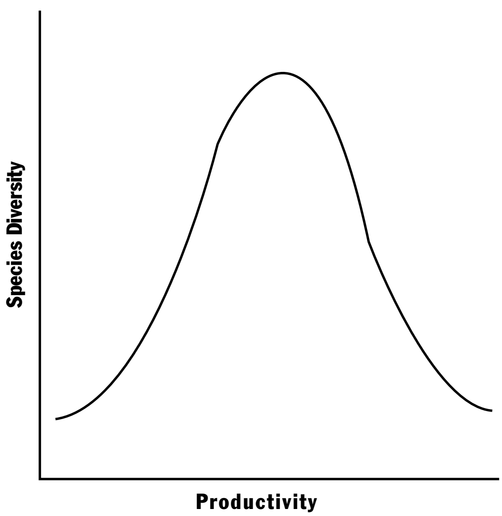

Figure 8-5. *Common, generalized relationship between productivity and species diversity.* Source: *From data in Huston 1993; Rosenzweig and Abramsky 1993; Tilman and Pacala 1993.*

construction material. This will tend to make commodity production on less fertile lands increasingly profitable in the future, whether at commercial or subsistence levels. As a result, the advancing frontier of commodity production—whether wheat fields, tree plantations, or aquaculture—will be decreasingly productive per unit of land or water, but the loss of species diversity will be increasingly severe in many regions as moderately productive sites are occupied. A countercurrent to this trend, however, may be that biodiversity and natural ecosystems will attain greater economic value as the supply of both declines. Thus, society should be increasingly willing to invest in their conservation.

We need to better understand the link between productivity and biodiversity to find optimal solutions to the problem of conserving biodiversity while simultaneously providing a high and sustainable production of photosynthetic-based commodities. To the extent that relationships between productivity and biodiversity exhibit regional and global patterns, such solutions will require international coordination. M. Huston (1993), for example, suggests that the conservation of plant biodiversity might be served by an increasingly global economy by enabling agricultural production to be concentrated in regions of greatest productivity, thereby sparing areas of high plant diversity where productivity is often lower. He cautions, however, that high productivity appears to be correlated with high diversity in other taxonomic groups, such as marine mammals and predatory birds, and thus different strategies will be required for these groups. Moreover, species diversity is only one of several components of biodiversity that must be considered. Solutions to these problems also require a better understanding of how biodiversity can be maintained in production areas and how it contributes to the long-term productivity of CCU products.

Regardless of the above considerations, it will be crucial to set aside and restore highly productive areas, some of which have been almost completely converted to commodity production (e.g., moist temperate grasslands, fertile floodplains, many coastal waters), to ensure full representation of biodiversity in the world's network of reserves. Fortunately, high-productivity, low-biodiversity ecosystems should be relatively easy to restore compared to low-productivity, high-biodiversity ones.

Few places on earth have the right combination of ecological and socioeconomic conditions for biodiversity conservation to be adequately served solely by the union of private property rights and unregulated markets. Southern Africa, where recreational hunting often provides both the economically most profitable land use and an incentive for landowners to maintain significant levels of biodiversity, is one of the few exceptions. CCU markets and natural ecosystems are usually much less compatible. Indeed, trophy hunting is one of the few exception that proves the rule, since it involves a strong element of nonconsumptive use in the form of nature tourism.

Thus, conservation of priority sites for biodiversity is unlikely to happen in a laissez-faire economy in which markets exist often exclusively for the commodity values of natural ecosystems while noncommodity and nonuse values provide no monetary return to the resource owner. In such cases, commodity production goals usually dictate decisions about resource use. Again, the exceptions are those few natural ecosystems in which nature tourism may be the most profitable use of land or water (with its own ecological costs) or ecosystems in climatic extremes or extreme isolation.

Further, spiritual, ethical, and other noninstrumental values of biodiversity cannot be adequately addressed in a free market system. Current market systems continue to favor a world increasingly dominated by a mosaic of monocultures and seminatural ecosystems that are moderately to intensively managed for commodity production. The consequence is significant losses of biodiversity on a global scale. To avert this scenario, we must create new markets or quasi-markets to help pay for the noncommodity values of biodiversity, and society, whether acting through governments or as individuals, must be much more committed to protecting the noncommodity values of biodiversity.

Managing CCU within a Larger World of Biodiversity Values

Although CCU values are but a subset of a much larger world of values, both economic and noneconomic, that society derives from biodiversity and natural ecosystems, finding effective ways to account for these societal values in the management of natural resources is a daunting but crucial task. Developing means of paying for these values, in both monetary and nonmonetary currencies, can only enhance attempts to make CCU ecologically sustainable. Where wild populations are overharvested, payment for other biodiversity values can reduce overexploitation both by providing alternative sources of revenue to resource managers and by increasing funding for improved management. Where wild species commodities now dominate management decisions in natural ecosystems, payment for other biodiversity values can tip the scale toward managing for more natural ecosystems and greater biodiversity. Where decisions are to be made regarding alternative uses of land and water, such payments will make protected areas a more competitive option, and areas of domesticated production and intensively managed CCU can be more readily viewed as biodiversity conservation tools confined to areas best suited for commodity production.

To place and manage CCU within the larger context of other biodiversity values, four major steps must be taken: (1) identify the biodiversity values of an ecosystem; (2) define and, where possible, quantify the social and economic benefits of these values; (3) develop ways to pay for or account for these values; (4) ensure that payment results in management that benefits biodiversity.

Identifying the Values

The resource owner can generally capture through the marketplace the economic value of direct uses of biodiversity, such as commodity production and nature tourism, but not indirect use, option, bequest, and existence values. Conservationists must develop more comprehensive approaches to identifying all benefits, both monetary and nonmonetary, provided by areas of priority for biodiversity conservation. We need to increase our knowledge of and better educate the public about biodiversity and its benefits, since neither emotional and spiritual attachment nor more instrumental values will develop for those components of biodiversity that are unknown or unexplained. This requires that we know well the products and services that a natural ecosystem provides and how it provides them (Janzen 1994).

Biodiversity conservationists must avoid putting all their eggs in the basket of instrumental biodiversity values. M. Sagoff (1995, p. 618) cautions that "Advances in technology may one by one expunge the instrumental reasons for protecting nature, leaving us only with our cultural commitments and moral intuitions. To argue for environmental protection on utilitarian grounds—because of carrying capacity or sources of raw materials and sinks for wastes—is therefore to erect only a fragile and temporary defense for the spontaneous wonder and glory of the natural world."

Stakeholders who benefit from the biodiversity of a natural ecosystem must be identified. For example, in a coastal estuary, these include, among others, the local fishing community that attaches commercial, recreational, and cultural values to the estuary; commercial fishing fleets operating offshore that depend on the estuary as a nursery for the fish they catch; hunters who shoot ducks that depend on the estuary; consumers who eat fish and game caught from the estuary; individuals and nonprofit conservation groups that value the estuary's biodiversity; academicians who conduct research on estuarine organisms and ecology; and coastal communities that benefit from the storm protection the estuary provides. Moreover, the world's population, both present and future, benefits from the carbon storage function and future use and nonuse options of the estuarine ecosystem.

Many of these stakeholders are free riders in that they are not paying for the benefits they receive from the estuary. In many cases, however, stakeholders may view the so-called free ride as a right for which they should not have to pay. Clean air and water are universal examples, but some may place biodiversity in this same category. Regardless, all of these stakeholders are beneficiaries of the natural ecosystem from whom payment in one form or another can be pursued.

Defining and Quantifying the Benefits

Once we understand the ecosystem and its beneficiaries (stakeholders), we can begin to define more precisely how they benefit, in what currencies they benefit, and in what currencies they can pay for those benefits. Even though monetary currencies dominate decisions about managing natural ecosytems, we should avoid the convenient trap of reducing all values to dollars and cents. As D. H. Janzen (1994, p. 11) observes, "Throughout the tropics, lured by the ecotourism dollar, there has been a very strong tendency to use the dollar as the primary currency in valuating conserved wildlands. While this has its good points, what seems to be forgotten . . . is that the 'poor' national user of a conserved wildland pays in votes . . . and in emotional attachment to the conserved wildland."

One can theoretically attach a dollar value to a sacred forest's spiritual importance to villagers based on their willingness to pay for conserving that forest, but such measures are impractical and meaningless when a community neither has money nor has ever contemplated the idea that such values must be paid for. The value of a coral reef can be assessed not only in terms of the grant monies a marine biologist attracts and spends on university overhead, travel companies, and local guest houses, but also in terms of the knowledge gained through research. The biologist, university, and local guest house can pay for these benefits by making monetary contributions to coral reef conservation efforts and by educating decision makers about the coral reef and developing local appreciation for the reef through public education. The net present value of future pharmaceuticals derived from a rain forest can be estimated, but how do we value in dollars the lives saved by those drugs? Although there may be practical approaches to estimating the monetary value of lives lost, as insurance companies do when they impute greater monetary value to lives lost in rich countries than those lost in poor countries (Fearnside 1997), many would agree that this is unethical.

Thus, the benefits of biodiversity come in many different currencies—human health and lives, knowledge and appreciation, and spiritual and ethical as well as monetary values. Similarly, there are diverse currencies by which people pay for these benefits—votes, education, persuasion, civil and uncivil disobedience, money, even lives. All these currencies must be considered in any thoughtful and ethical approach to managing the earth's resources lest we allow economic considerations, and thus the world's wealthy, to dominate the decision-making process. Yet it is the wealthy individuals and nations that, though often exerting a disproportionate pressure on the world's natural capital, are also best able to pay for maintaining the societal benefits of biodiversity. Similarly, though there is concern about the effects of global markets on biodiversity, it is global markets that increasingly allow the wealthy on one side of the globe to pay for biodiversity values on the other side.

As the ecosystem valuations presented in table 3-1 indicate, where economic values alone are considered, CCU values generally represent but a minor fraction of the total value of natural ecosystems. Although many of the economic values assigned to biodiversity and natural ecosystems are rough approximations, it is clear that their significance cannot be ignored. Once the principal stakeholders have been identified and these biodiversity values estimated, the challenge remains of finding ways to pay for them.

Paying for the Benefits

There are basically three aspects to the problem of paying for biodiversity values: (1) convincing biodiversity beneficiaries that they should pay, (2) finding a mechanism through which payment can be made, and

(3) getting the payment into the right hands so that it serves as an incentive for biodiversity conservation. The first aspect of the problem primarily involves educating people about the benefits they receive and the threats to those benefits as biodiversity declines. Different perspectives about who is responsible for biodiversity protection present a problem. At the international level, developing countries with high biodiversity may negotiate payments from developed countries with the implied threat that they will cut their forests down if payments are not made. But the attitude in developed countries often seems to be that it is the responsibility of countries with high biodiversity to protect their own natural heritage and thus payments are optional (Fearnside 1997). Clearly, neither extreme view is right or constructive, and the answer is that biodiversity is a shared responsibility among nations.

Nature tourism, purchase of green-labeled products, debt-for-nature swaps, taxes funneled to intergovernmental monetary instruments such as the Global Environmental Facility, contributions to nonprofit conservation organizations, carbon offset payments, and purchase of development rights are among existing and emerging market mechanisms that enable near and distant stakeholders to pay to maintain biodiversity and natural ecosystems (Pearce 1995; chapter 3 of this book). Considerable progress remains to be made in improving the magnitude and efficiency of some of these markets and in developing other market mechanisms.

These markets, however, should not divert attention from the importance of governments, from local to national levels, in providing mechanisms for giving voice to biodiversity values through votes and other nonmonetary currencies. The world's system of parks and reserves would surely be much less developed today had we waited for the results of ecosystem valuations and biodiversity markets to justify and pay for them.

Getting monetary payments into the right hands is crucial if payments are to work as an incentive for biodiversity conservation. The premium paid for a green-labeled product provides one of the most direct methods to get money from those who value biodiversity to those who manage it. Nature tourism also has this potential. At the other extreme are intergovernmental payments by which, whether via multilateral banks or bilateral assistance programs, governments of developed countries pay governments of developing countries for conserving biodiversity. P. M. Fearnside (1997, p. 68) asks, for example, "If the nations of the world miraculously agreed to pay handsomely for the environmental services of the Amazonian rainforest and sent the government [of Brazil] a check, how much of this money would actually go to the principal objectives: maintaining the forest and supporting the region's population?" Even if the problems of corruption and government agencies taking more than their fair share could be avoided, such payments must be dispersed in a way that creates conservation incentives for local resource owners and harvesters. This requires close attention to developing both a well-defined and well-enforced system of resource tenure and use rights and a regulatory framework that encourages sustainability and protects against practices that would jeopardize biodiversity.

The effectiveness of payments for biodiversity payments can be enhanced if we reconsider the meaning and terminology of grants given by foundations, nonprofit organizations, bilateral agencies, and others who support biodiversity conservation. Rather than providing financial "assistance" or "subsidies," terms that convey the sense of a handout, grants are in fact payments from willing-to-pay beneficiaries of biodiversity to those responsible for providing the benefits—the stewards of natural ecosystems. This implies a longer-term commitment on the part of both beneficiaries and providers, the kind of commitment biodiversity conservation requires. Conservation trust funds, jointly endowed and overseen by both biodiversity beneficiaries and providers, provide a mechanism for applying this philosophy.

Ensuring That Payments Benefit Biodiversity

Finally, any program of payments for biodiversity requires assurance that the recipient has the capability to manage for biodiversity and that the results of such management will be monitored. A producer of green-labeled fish products must clearly understand what ecological sustainability, as a certification criterion, means in terms of harvesting and management practices. Further, the producer must have the skills and means to carry out such practices. Similarly, a buyer of development rights of a natural wetland who wishes to maintain the wetland's biodiversity value must clearly understand and be able to implement management measures required to maintain that value. The dynamic and unpredictable nature of ecosystems can make setting and implementing such management targets a formidable task that requires close and ongoing communication between buyers and sellers (in whatever currency) of biodiversity values.

Global Ecological Sustainability

The search for ecological sustainability in the use of wild species commodities involves finding the right alignment of a host of social, economic, and ecological conditions. The conditions and alignment, however, are never perfect. Although various factors—property rights, responsibility to future generations, respect for the spatial and temporal scales of ecological processes—are largely universal in terms of their importance to ecological sustainability, solutions to specific management problems must be local, since so much depends on the social, economic, and ecological context of each area and type of use. Yet even though solutions are necessarily largely local, they must be molded with a global perspective. National and international coordination and priority setting are needed to develop the right policy framework for biodiversity-friendly markets and to develop intelligent trade-offs in allocating uses of land and water and conservation resources.

There is one overarching universal, however, that cuts across collective efforts and will determine their success. The product of the world's human population and per capita consumption—the global scale of resource use—has only one possible relationship with ecological sustainability (figure 8-6). If we fail to deal effectively with population and consumption, any victories of ecological sustainability will be only local and temporary and biodiversity will continue its slide down the slopes of overexploitation and economic specialization.

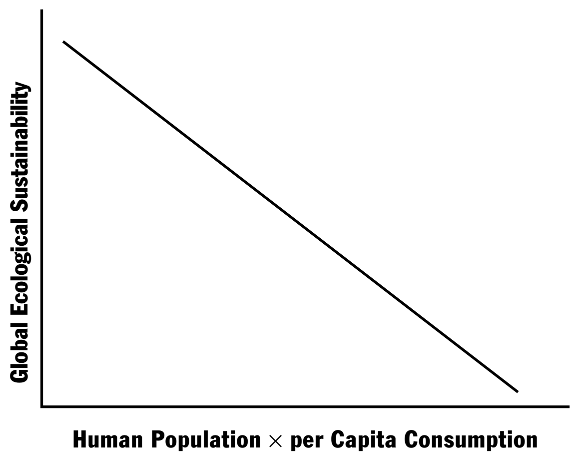

Figure 8-6. *Global ecological sustainability as the product of human population size and per capita consumption.*

Commercial consumptive use of wild species can be a tool for biodiversity conservation, but it carries both an economic and an ecological price. Implementing more sustainable practices will require making greater financial investments. Some of these investments will come from the consumer as the true costs of wild species use are passed on, and some will come from governments (and thus their constituencies) if they act responsibly to protect the societal values of biodiversity for current and future generations. No matter how well we succeed in getting the economics right, however, the use and management of wild species also will inevitably entail an ecological price as ecosystems are manipulated to invest more of their resources in production goals.

But perhaps most important, ecological sustainability requires a social investment whereby intrinsic value is assigned to the concepts of stewardship and natural heritage, to taking responsibility for managing and preserving biodiversity for others. "Community" as a focus for social responsibility must be defined broadly to include natural and seminatural ecosystems and their inhabitant species. This demands an ethic simultaneously more altruistic, in terms of the natural legacy shared with current and future generations, and more biocentric, in terms of consideration of the intrinsic values of nature. If this social investment can be made, the uses we make of wild species for their instrumental values can be a stronger ally in our efforts to restore and maintain the largest possible range of the earth's biodiversity.

Appendix: WWF Guidelines for the Commercial Consumptive Use of Wild Species

The following fifteen guidelines were developed by WWF for managing CCU for the benefit of biodiversity and natural ecosystem conservation (Freese 1996). The guidelines go beyond strictly economic concerns and consumptive use to provide a framework in which conservationists can approach all forms of use, consumptive and nonconsumptive, with a concern for diverse values, both economic and noneconomic.

The guidelines are meant to serve as a broad and flexible blueprint to be adapted to the specific conditions of each region and individual management situation. They provide a combination of general principles to follow and strategies or actions to consider in linking CCU to biodiversity conservation. The guidelines should be considered and applied as a whole, not individually, and are best interpreted within the context of the issues and concerns raised in this book. Further, the guidelines are presented with the expectation that they will evolve and improve as experience is acquired.

Finally, the guidelines are only as good as the intentions and judgment of those who use them. Managing CCU for the benefit of nature conservation requires the best of both.

Five Basic Assumptions

The guidelines were developed within a framework of five basic assumptions:

1. The world's protected areas are critical but far from sufficient for conserving biodiversity. The future of much of the earth's biodiversity depends on how we use and manage unprotected areas of existing or potential value to conservation. CCU has an important role in such use and management.

2. There is redundancy in the ecological roles of species in most ecosystems. Thus, ecosystems can be managed to favor populations of some species to the detriment of others and of biodiversity in general without necessarily undermining important ecosystem functions such as primary productivity and nutrient cycling. The threshold beyond which management intervention and use begin to significantly impair ecosystem functions will vary among ecosystems and is poorly understood.

3. Human intervention in an ecosystem for commercial purposes inevitably alters and generally simplifies, at some scale, ecosystem structure, composition, and function. Thus, the employment of CCU as a conservation tool may imply a trade-off between the ultimate conservation goal of maintaining natural ecosystems and the biotic impoverishment and loss of "naturalness" of those same ecosystems due to human use.

4. A policy of no use, however, will often lead to even greater biotic impoverishment as natural ecosystems are converted to other uses that yield greater economic returns. We must therefore minimize the adverse effects of CCU while using it as a tool to preclude conversion of natural ecosystems to alternative uses.

5. Human population and per capita consumption cannot continue to grow indefinitely in a world of finite resources. Under a scenario of continued growth, the scale of demand will only further overwhelm any semblance of ecological sustainability as the world's lands and waters are managed with increasing intensity to serve human needs. If we do not change course, an ever-larger share of the earth's biodiversity will be lost either to overexploitation or by the conversion of natural ecosystems to domestic forms of single-commodity production.

The Fifteen Guidelines

Guideline 1. Wild species have intrinsic worth and nourish human cultural and spiritual values that should balance the influence of economic forces in nature conservation.

Implicit in this guideline is that nature and the biodiversity it represents should be conserved regardless of any economic benefits they might provide. This serves as an overarching philosophy for approaching the management of nature for human benefit.

Humankind exerts a global and dominating influence on the future of the natural world, and degradation of nature has often resulted from unwise management directed toward its commercial use. We can help prevent the perverse effects of commercial forces by giving increased priority to those values of nature that cannot and should not be subject to purely monetary measures of worth. These values must be considered in decisions regarding the commercial consumptive use of wild species.

Cultural and spiritual values attached to nature vary among people and cultures. In much of the world, however, society is increasingly disconnected from nature, a trend that must be reversed if greater attention is to be given to noneconomic values. This will require that we find ways to expand and deepen human experience with and knowledge of the natural world (see also guideline 13).

Guideline 2. CCU should be promoted only where it is likely to create conservation benefit.

The challenge here is to determine the net benefit CCU may provide to conservation, particularly in its role of maintaining natural ecosystems by helping offset the opportunity costs of alternative uses of land and water and by relieving development pressures on other areas of importance for biodiversity conservation. This requires an understanding of the pressures exerted for alternative uses, the probability that CCU will deter such uses, and the negative effects on, and risks to, biodiversity posed by CCU. It also requires a global view of how CCU in one area may affect other production sectors and natural ecosystems. To achieve such an understanding, we must assess the environmental costs of producing substitutes for a CCU product and the relative suitability and importance of different areas for commodity production and biodiversity conservation.

Where CCU currently exists but is poorly managed, the focus should be on improving management and mitigating negative effects on biodiversity. Uses of no conservation benefit must closely adhere to the precautionary guidelines. Thus, in cases such as those involving open marine fisheries, in which any conservation benefit derived from the use of some species may be marginal or nonexistent, the burden of proof is on the users to demonstrate that the risks of overharvesting and of eroding biodiversity are negligible. Where CCU produces a significant conservation benefit, such as preclusion of competing land or water uses, the precautionary principle must be applied with flexibility to ensure that attempts to reduce the risk of overharvesting are balanced by attempts to avert the loss of the socioeconomic benefits required to maintain the natural ecosystem. In addition, if a proposed use serves no basic human need, conservationists should be prepared to oppose it if it provides no conservation benefit or, regardless of conservation benefit, if it involves a violation of basic human rights or wanton inhumane treatment of animals.

Guideline 3. A CCU program should preserve current and future options by maintaining biodiversity and preventing irreversible changes in the ecosystem.

We poorly understand how biodiversity and ecosystem functions currently benefit society and are even more ignorant of potential future products and services from nature. Maintaining options, therefore, requires maintenance of the full range of biodiversity at the genetic, species, and ecosystem levels. This, along with the need to maintain basic ecosystem functions, implies that ecosystem changes caused by CCU should be reversible. Where the scale and type of CCU may irreversibly reduce biodiversity or jeopardize ecosystem functions, conservation strategies should include nonuse zones or protected areas where biodiversity and ecosystem functions are fully maintained. Such areas are important not only for their current and future benefits to society but also as benchmarks against which to assess the effects of human intervention and as research laboratories for improving our understanding of how to manage ecosystems.

Furthermore, we must manage for unpredictable environmental changes, both at the local and global levels. To do this, management must ensure that organisms and ecosystems maintain their resilience to disturbances in their environment and their adaptiveness to future change.

Guideline 4. Natural ecological fluctuations and processes and the life histories of organisms should provide the blueprint for the design of CCU programs.

The long-term health and productivity of ecosystems often depend on maintenance of natural ecosystem fluctuations and processes. Ecosystems are dynamic, and therefore yields of particular wild species commodities may vary greatly over time. In contrast, commercial markets prefer a steady and predictable flow of goods, generally with a goal of maximizing the flow. Management should approach such goals with caution because attempts to maximize and stabilize productivity, particularly of one or a few species, may undermine ecosystem processes, interrupt life cycles of the managed species, and jeopardize biodiversity and long-term productivity. In some cases, short-term socioeconomic gains from efforts to maximize and stabilize productivity may be important for offsetting the opportunity costs of alternative uses of land or water, but if such short-term gains can in fact be realized, they must be balanced against the risks to long-term socioeconomic and ecological sustainability. For these same reasons, the risks posed by introducing exotic species or stocks into natural ecosystems or into areas from which they may invade natural ecosystems should be assiduously avoided.

Management under this guideline means that sustainable harvest levels for any one species may fluctuate, and goals that call for unvarying levels of maximum sustained yield must be abandoned. Producers and marketers must exercise flexibility and restraint for such management systems to work. Periods of high productivity may lead to overcapitalization, as has occurred in many marine fisheries. This leads to a resistance to reduce harvest levels during periods of low productivity. Diversification of uses from an ecosystem provides a buffer to this problem, since periods of low productivity in one product may be balanced by high productivity in another (see guideline 8).

Management must be particularly sensitive to those life history characteristics of individual species that make them susceptible to overharvesting and population decline. For example, long-lived, slow-reproducing species, such as whales, elephants, and primates, can withstand only low levels of adult mortality if populations are to be maintained. Fast-reproducing species, such as many fish, pose a different set of problems. These species are often characterized by large and unpredictable population fluctuations, and establishing safe harvest levels for them is therefore difficult. If intensive harvesting occurs during years of natural population decline, the population may be driven to dangerously low levels or extinction.

This guideline also emphasizes the importance of considering landscape-level interactions. Nutrients, energy, and organisms move among ecosystems. Thus, although management must be tailored to the properties of particular species and ecosystems, it must also consider interactions at the landscape level and even larger spatial scales. Migratory species merit special caution.

Finally, consumptive-use programs should avoid and, to the extent possible, reverse trends in habitat fragmentation at both local and regional scales.

Guideline 5. Adaptive management is required to cope with uncertainty in both ecological and socioeconomic systems.

We can never perfectly predict the behavior of either ecological or socioeconomic systems. Those employing an adaptive management approach to CCU recognize this uncertainty, plan for the unexpected, expect management mistakes to be made, and view mistakes as opportunities to learn. This philosophy and attitude must be adopted at all levels, from local resource users and managers to researchers and top-level policy makers.

Adaptive management is an iterative approach that links research and management. A premium is placed on learning from management interventions and on adjusting policy and management in response to new information. Both local resource users and policy makers must be integrated into this process if scientifically sound management measures are to be accepted and implemented.

Adaptive management requires investment in ongoing monitoring and analysis and creation of feedback loops among researchers, field managers, resource users, and policy makers. Where new CCU programs are proposed, up-front investment in gathering baseline ecological and socioeconomic information is important in designing initial management interventions and understanding their subsequent effects.

Adaptive management often entails substantial investments above and beyond normal management operations. In regions with extremely limited financial and technical resources, innovative methods of adaptive management are needed, such as enlistment of local resource harvesters and users in monitoring and data analysis.

Guideline 6. Knowledge and skills from local user groups and traditional resource management systems should be integrated with knowledge from scientific research to improve the design and monitoring of CCU programs.

The knowledge of local resource users, particularly in traditional or long-standing systems of wild species management, can help guide research and complement knowledge provided by Western science in designing effective CCU management programs. In addition, as suggested in guideline 5, local resource users' firsthand experience and day-to-day interaction with the species and ecosystems under management can be tapped to improve monitoring and adaptive management approaches. Where traditional systems have been ecologically sustainable, greater attention should be given to retaining these systems rather than supplanting them with more centralized, top-down approaches to management.

Traditional systems alone, however, may not be sufficient when species use moves from subsistence use and local markets to the more intensive use and management interventions inherent in national and international market economies. In these cases, Western science can provide new information and insights for management and help reduce the risks of overexploitation and broader ecosystem effects. Again, however, Western science and approaches to management should attempt to build on, rather than replace, local knowledge and traditions.

Guideline 7. Revenues from CCU should be sufficient and distributed so as to cover the costs of ecologically sustainable use and to create incentives for conserving biodiversity and natural ecosystems, but CCU should not be expected to carry the full costs of conservation.

Revenues obtained from CCU must be sufficient to cover both the management costs of sustainable harvest programs and the costs of mitigating any negative effects the programs impose on biodiversity or the ecosystem. Further, for CCU to be of conservation benefit, the revenues it generates must contribute to offsetting the opportunity costs of alternative uses of the land or water. It is crucial that these costs be fully accounted for in the price that consumers pay for wild species products; this is the essence of green marketing (see guideline 14). Care must be taken to ensure that revenues are in fact distributed in such a way that they provide the incentives and funds needed for sound management. Thus, a large share of the revenues must go back to those stakeholders responsible for decisions and management regarding use of the land or water.

Reliance on CCU alone, however, to economically justify conservation of natural ecosystems invites economic specialization and ecological simplification; in the extreme, the natural ecosystem will be converted to other uses if other biodiversity values are not incorporated into decision making. As guideline 8 indicates, the more we can quantify and capture various biodiversity values, the greater will be our chances of averting resource uses that are inimical to biodiversity.

Guideline 8. Diversified economic benefits from biodiversity and natural ecosystems, secured in part by getting free riders of biodiversity-based values to pay their fair share, should be promoted.

As suggested in the discussion of guideline 7, diversified economic benefits from a natural ecosystem will help avert the process of ecosystem simplification that CCU markets often favor and should improve the ability of the natural ecosystem to compete economically with alternative uses of the land or water. Greater diversification of benefits should also result in greater ecological and economic robustness, and thus sustainability, of the system. Periods of low productivity of some species may be compensated for by the high productivity of others, and a diversified array of products buffers the ups and downs experienced by individual products in the market.

There are three basic approaches to diversifying benefits. One is to find new uses, consumptive and nonconsumptive, for components of biodiversity that have little or no current market value. Efforts to find uses and markets for the wood of lesser-known species of tropical trees are an example. This approach, of course, is not without considerable risk, as new and expanding markets for a species or natural ecosystem can rapidly lead to overexploitation and abuse if sound management methods and controls are not in place.

The second approach is to capture the value of existing biodiversity benefits. Free riders of biodiversity benefits, whether based on use or nonuse values, must begin to pay their fair share so that the full spectrum of biodiversity values is incorporated into decision making. In some cases, it may be easy to identify and secure payment from free riders, such as from a nature tourism company that is using a wildland for free. In other cases, new market mechanisms must be sought that enable other, often more distant, beneficiaries to pay for the maintenance of biodiversity and ecosystem functions. Where the benefit is more of a public good, such as the carbon sequestration value of a forest or option values offered by biodiversity, it may be difficult to identify who should pay or how they should pay. In such cases, governments and intergovernmental institutions may be required to facilitate international agreements and transfer payments for maintaining these societal benefits.

The third approach to diversifying benefits is through value-added activities, such as the local processing of logs into sawed lumber or local processing and packaging of medicinal plants, that increase the overall revenues generated from a wild species commodity. This can reduce pressures to increase revenues through higher and potentially unsustainable harvest rates.

An underlying risk of these approaches is immigration to the area by people wishing to participate in the new revenue opportunities. Such population growth may undermine efforts to achieve sustainability in resource use and cancel any socioeconomic gains made by the original residents. Establishment of secure resource rights for the original residents will help curb this problem (see guideline 12).

Guideline 9. To make natural ecosystems competitive with alternative uses of land and water, some biodiversity may need to be sacrificed.

Many natural ecosystems can be maintained only if they can compete economically with alternative uses of the land or water, particularly the production of commodity crops and livestock. In general, landowners, whether private or public, will decide which land-use option to pursue based on opportunity costs of land, capital, and labor.

Despite our best efforts at capturing the numerous noncommodity values of biodiversity, in some cases specialization in wild species with high market value may provide the only viable strategy for competing with alternative land or water uses. Such a strategy places a premium on increasing revenues from the species being marketed per unit of land or water under management. Managers will therefore attempt to increase production and improve the quality of the wild species commodity while suppressing or eliminating less valued and competing components of biodiversity. Above some threshold, such management inevitably leads to decreased species diversity, altered and often decreased genetic diversity in the managed species, changes in ecological processes (e.g., generally more energy and nutrients directed toward the commercialized species), a more simplified ecosystem structure, and other ecological trade-offs in the management area.

Thus, although economic specialization in certain wild species commodities implies biodiversity trade-offs, without it some natural ecosystems will be converted to more profitable forms of land or water use with much greater effects on biodiversity. Moreover, as noted in the discussion of guideline 2, intensive production in one area may relieve pressures for commodity production in other areas of high conservation priority.

The caveat to this management approach is that there is mounting evidence that long-term sustainable offtake of selected species may require maintenance of a greater degree of natural ecosystem processes and diversity than was previously thought (see guideline 3). Simplification of the ecosystem may yield short-term gains, but the price may be long-term declines in productivity. Further, to the extent that ecosystem changes are irreversible or only slowly reversible, ecosystem simplification limits the ability of the land manager to respond to changing markets and values that may favor new products and services from natural ecosystems.

We need to develop benchmarks of biodiversity and ecosystem integrity to help determine the type and magnitude of change acceptable in such intensively managed natural ecosystems and to aid us in monitoring those changes.

Guideline 10. Benchmarks of biodiversity and ecosystem integrity should be developed and used to set conservation goals, to define acceptable limits of change due to CCU, and to provide a benchmark against which to monitor change.

As indicated in the discussion of guideline 9, CCU and other human activities alter biodiversity and ecosystems in diverse and sometimes subtle ways. Although the general goal of biodiversity conservation is to maintain native biodiversity and ecosystem processes, specific goals are poorly defined and means of evaluating efforts to meet those goals are not well developed. Setting of such goals is problematic for various reasons—the difficulty of defining *native* and *natural,* our ignorance about biodiversity and processes in any given ecosystem, the dynamic nature of ecosystems, and the expense and difficulty of monitoring and evaluating ecosystem change.

Despite these difficulties, the need to determine the conservation benefit of a particular management objective and to set limits to, and monitor biodiversity changes induced by, CCU programs requires that greater attention be given to establishing benchmarks of biodiversity and ecosystem integrity. These benchmarks can then be used to help define biodiversity conservation goals and to evaluate ecosystem change. Such benchmarks cannot be static but must recognize the inherent fluctuations and long-term changes that characterize most ecosystems. They must also recognize that few if any ecosystems are "pristine" in the sense of being free of human influence; rather, the challenge will be to decide what level and form of additional anthropogenic change, if any, is acceptable.

Our ability to identify benchmarks and set conservation goals for a particular ecosystem will benefit from greater knowledge of that ecosystem's biodiversity and processes. However, science can take us only so far. It can tell us the number of species and how many of each are in a forest or estuary, but it cannot tell us how much we should be willing to change those species compositions when managing an ecosystem for human use. CCU will often require compromises with our loftiest conservation goals. Both our goals and acceptable levels of compromise must in large part be based on the diverse values stakeholders bring to the table. The challenge then lies in articulating biodiversity-based values, educating all stakeholders to these values, and sensitizing them to their importance.

Guideline 11. Ecological and socioeconomic subsidies will often be required to make the transition toward sustainability.

The transition toward sustainable use, both consumptive and nonconsumptive, of wild species is a slow process. Years—often decades—may be required to develop effective management interventions, to allow the recovery of populations depleted by overuse, and to increase revenues to the point of socioeconomic sustainability. Yet decisions about the "best" use of the land or water are generally based on short-term economic and political criteria. This may result in the natural ecosystem being converted to other uses before long-term returns from management can be realized.

Financial investments or subsidies from outside sources may therefore be required to maintain socioeconomic and political support while revenue-generating activities from the ecosystem are materializing. These funds may be used to support community services, as wages for resource management efforts, or as outright cash payments to key local stakeholders. Such investments may come from nonlocal stakeholders (e.g., conservation organizations or bilateral assistance agencies) who wish to restore or maintain the natural ecosystem and its biodiversity values. Such investments should be viewed as valid payments for biodiversity values and not as subsidies.

Often, however, external sources of financial support may be inadequate or nonexistent. In such cases, socioeconomic needs must be met by the use of natural resources, even though this may entail overharvesting of some species and may carry broader risks of ecosystem degradation. Use of an ecosystem's resources in this way poses high long-term risks, since once the target species are economically depleted, other biodiversity-based sources of revenue must be found. This strategy should be tried only if there is no alternative and only if significant and irreversible losses of biodiversity can be prevented.

Guideline 12. Management of wild species should be balanced between assignment of resource tenure and management responsibility to the lowest level commensurate with the scale of resource use and regulation by broader authorities for the public good.

A dual mechanism of checks and balances is generally required in order to represent fairly the interests of two primary groups of stakeholders. One group comprises those who live near, and often depend directly on, products and services from natural ecosystems. For these local stakeholders to be good resource stewards, they must have secure tenure or usufructuary rights, and they must receive their fair share of benefits derived from resource use. This tenet simply recognizes that incentives for sustainable resource management are crucial for those who most directly exert control over the resource. Although such rights are often best vested in individuals or local communities, corporations or government agencies may also be suitable tenure holders if they have the incentives and capabilities for sound management. Regardless of who the current tenure holders are, we should ensure that their tenure rights are legitimate and were not obtained by depriving traditional holders of those rights without fair compensation.

Substantially greater oversight from higher authorities will generally be necessary for migratory or highly mobile species, since no one individual or community has exclusive rights to use of such species and none will feel solely responsible for their management. This need is greatest in the management of living resources on the high seas, where international accords and national goodwill and restraint provide the only means for sound stewardship.

The other body of stakeholders whose interests must be addressed is society at large and future generations who value indirect-use, option, bequest, and existence values of natural ecosystems. These values often are not addressed by traditional market-based incentives that motivate local stakeholders and owners. For example, the benefits to downstream stakeholders of watershed protection and the broader public health benefits of pharmaceuticals developed from wild plants are not readily paid for by the relevant beneficiaries. Many stakeholders may consider such biodiversity benefits to be rights for which they should not have to pay. In these cases, action by government is needed to secure payment from free riders via taxes or other mechanisms for paying resource owners and managers to maintain biodiversity benefits. In addition to incentives, sanctions will often be required to direct resource owners toward maintaining the societal benefits of biodiversity.

Guideline 13. The full value of biodiversity and ecosystem services should be reflected in decision making about management of a natural ecosystem.

Policy and management decisions tend to be based on a small part of the full array of benefits provided by natural ecosystems and biodiversity and on short-term rather than long-term economic and ecological criteria. Such decisions favor the most visible commodity values over less visible economic and noneconomic values. There is considerable inertia in existing political and economic systems that caters to well-established economic stakeholders at the expense of new ones. Discounting and the silence of future generations favor short-term gains over long-term sustainability. Resource depletion is not accounted for in prices or in national accounting, and thus the mining of living resources appears economically rational. Further, macroeconomic policies often fail to account for biodiversity values and to favor overconsumption. The consequences are economic specialization, ecosystem simplification, and overexploitation—or, worse, complete conversion of the native ecosystem to alternative uses.

This problem can be rectified only if we address the motives and incentives that influence decision makers, from top government officials to local landowners. In general, decision makers must see that someone is willing to pay for biodiversity values. Payment comes in diverse currencies: it may be monetary or in the form of votes or appeals to spiritual and cultural values. The latter cannot and should not be reduced to pure expressions of monetary worth (see guideline 1).

Education and awareness are perhaps the most fundamental strategies for achieving such change. One does not value what one does not understand, and the effects on others of biodiversity loss may be delayed, indirect, and unintended. Decision makers, from consumers and managers of CCU products to policy makers, should make more enlightened choices that benefit biodiversity conservation as they learn more about the full range of biodiversity values. Free riders must first realize how they benefit from biodiversity and ecosystem services before they can be expected to pay (whether through the marketplace or through taxation), vote, or lobby to maintain those benefits.

Guideline 14. Market demand should be more effectively used as a tool to promote better management.

The consumer is the other major decision maker who affects how species and ecosystems are managed. Indeed, for consumptive uses of wild species, the consumer is arguably the single most important decision maker. There is no use of a product without consumer demand for it. Excessive use occurs only if the demand is too great, but strong demand, particularly from environmentally conscientious consumers, is necessary for CCU to be an effective conservation tool. An educated and aware consumer can exercise substantial influence over decisions made by the natural resource manager, owner, or policy maker. Public awareness campaigns can reduce consumer demand for wild species products that are being overexploited, can increase demand for "environmentally friendly" products, and can lead to policy changes through the political process.

Green marketing requires that consumers exercise a preference for products obtained from wild species that are well managed. Because good management often entails additional costs, the consumer pays a premium for the green product. To be effective, a large part of this premium must find its way back to the resource owner or manager to provide an incentive for, and cover the costs of, ecologically sound management.

Green marketing also requires that consumers be aware of the environmental effects of their buying decisions and that credible methods are in place to certify environmentally sound products. Further, for some wild species products, the commodity trading structure can be a major impediment. Traders often want to keep producers and consumers apart because they do not want products to be differentiated, as green labeling does, and thereby require special handling. As a result, those willing to produce environmentally sound products often must bear the additional costs themselves rather than pass them on to the environmentally concerned consumer.

When attempts to influence consumer demand and choice fail, trade bans and other trade regulations, though of limited effectiveness for some products, may be the only alternative for curtailing the flow of unsustainably harvested wild species products.

Guideline 15. Better national and international coordination and action are required to improve the sustainability of CCU and its use as a conservation tool and to mitigate its negative effects on biodiversity.

In an era of increasing ecological interdependence and economic globalization, improved coordination of national and international policies is required. Such coordination should include harmonization of regulatory controls, improvement of national and international monitoring of trade, and development of mechanisms that use market forces to encourage conservation. In particular, new national and international market mechanisms must be found for near and distant stakeholders to pay for the benefits they receive from biodiversity, thereby reducing the need for CCU to shoulder the burden of economic justification for natural ecosystem and biodiversity conservation. Finally, ways must be found to incorporate the noneconomic values of biodiversity into the policies and decision-making processes of national and international institutions.

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