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The Economics of Terrestrial Biodiversity Conservation in Developing Nations

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I. Introduction

Mounting evidence suggests that humans are substantially reducing the diversity of biological resources on the planet (Warrick 1998; Ceballo and Erlich 2002). Some natural scientists predict that a third or more of the species on earth could become extinct in this century. Such losses are encountered in the geological record only as a result of catastrophic events. Moreover, a substantial percentage of global biodiversity can be found in developing nations, including many of the most endangered species. These same nations have limited means for protecting their biodiversity and myriad other pressing social needs that make competing claims on biodiversity. Given that biodiversity loss is fundamentally an economic problem, economic theory and empirical analyses can play an important role in helping to protect biological diversity in developing nations.

A. Defining, Measuring, and Locating Biodiversity

The term "biodiversity" is broad but attempts to describe the variety of living things. Biodiversity can refer to variety in genes, species, and ecosystems. For each level, biodiversity can be measured in a range of ways including counting the number of species or determining the "richness," the relative abundance, or the degree of similarity among species. Biodiversity also includes several attributes such as compositional, structural, and functional biodiversity (Noss, 1990; Franklin, 1993). Some people view biodiversity as an environmental service although there is growing consensus that biodiversity contributes to the creation of environmental services rather than being a service itself (Millennium Ecosystem Assessment, 2003). Appropriate measures of biodiversity also rely on the spatial and temporal scales for biodiversity. In short, this range of levels, scales, and attributes implies that there is no single number that depicts biodiversity.

It is also not clear how many species or genes exist. Biologists have described approximately 1 million species, of which about 100,000 are well-known. Estimates of the total number of species globally range from a few millions to tens of millions (May, 2000). Policies cannot, therefore, be based on an actual count of species but are instead based on indicators of biodiversity. These indicators vary with scale but include concepts such as IUCN Red List indicators (lists of vulnerable species), area in protected status, and levels of ecosystem services.

Species, genes, and ecosystems are not evenly distributed across the globe. In fact, approximately half of all terrestrial species are located in 25 areas that cover about ten percent of land (Myers, 2003; Pimm and Raven, 2000). Approximately 16 of these 25 high-biodiversity, and two-thirds of terrestrial species areas are located in tropical forests, which fall largely in developing countries, especially in the Amazon, Congo, and Southeast Asia (Pimm, MEA, 2003).

Only 12 percent of the original habitat in these 25 areas remains, which implies significant losses of species already (Myers, 2000). Although the non-linear biological relationship between species and area suggests that more than 12 percent of species remain, the rate of species extinction in recent history appears to be many times –perhaps several thousand times – the natural rate of extinction. Among the most important proximate causes of this biodiversity loss are habitat destruction, although hunting, non-native invasive species and climate change are also important in current and future losses (Vitousek et al. 1997; Pimm and Raven, 2000; Thomas et al. 2004). In our analysis of policy responses to biodiversity loss, we concentrate on habitat destruction, but we will briefly discuss responses to other causes of biodiversity loss as well.

B. The Value of Biodiversity

The debate over how much and what kinds of biodiversity should be protected is rife with uncertainty. Scientists do not know how much biological diversity exists on the planet, nor exactly how biological diversity supports the ecological services on which humankind depends. Attempts to highlight the importance of biodiversity point to myriad outputs whose production depends on biodiversity: hydrological services, climate regulation, soil management, pollination services, desalinization, biosphere resilience, tourism, pharmaceutical and industrial chemical research, and consumptive outputs like timber, fuelwood, meat, medicines, fruits, nuts, ornamental plants, domestic pets and a variety of other non-timber ecosystem products. Theoretical and empirical work has identified links between changes in biodiversity and the way ecosystems function (Loreau et al. 2002). Economists (Alexander 2000; Simpson 2000) and biologists (Wilson 1984) have also noted that biodiversity can be valued for nonconsumptive uses such as spiritual or artistic inspiration. Finally, arguments can be made for protecting biodiversity based solely on our current ignorance: there may be substantial value in retaining the option to discover more about biodiversity's importance and hidden role in our lives before we irreversibly extinguish it.

Given all of these potential values, one might then ask, "How much is biodiversity worth?" This question is not only controversial (Alcamo et al. 2003: 127-147), but largely unanswered at this point in time. Economists have made modest and incomplete attempts to value ecosystems and related ecosystem services in developing nations (e.g., Pattanayak and

Kramer 2001; Kramer and Mercer 1997). To our knowledge, no attempt to estimate the value of a specific endangered species has been completed in a developing nation, although a few studies have been completed in the U.S. (see Loomis and White 1996, Reaves et al., and references therein). Even if they had been completed, however, the way in which these values should be aggregated and then incorporated into policy decisions is an open question. The exercise of putting a dollar value on a globally-valued ecosystem (e.g., tropical rain forest) or species (e.g., minke whale) puts extreme theoretical and empirical demands on already controversial valuation methods (Carson 1998).

Given the discussion above about the important benefits of biodiversity protection, one may wonder, "If biodiversity is so valuable, why do we continue to see declines in biodiversity indicators?" If tourism associated with the visitation of ecosystems and wildlife is so important in many developing nations (Wells 1997), why doesn't the tourism industry invest in maintaining one of its most important inputs? Why do water users not invest in protecting the biodiversity that contributes to their maintaining their water supply? Part of the answer lies in the same attribute of biodiversity that makes it so valuable: it is a global resource from which all humans on the planet derive value.

Biodiversity protection is a classic public good: once it is provided, no one can be excluded from the benefits and one person's enjoyment of these benefits does not reduce the benefits available to other people. However, when people destroy biodiversity through their consumptive use of species and habitat, the benefits from that destruction are *private*. Thus people receive tangible private rewards for destroying biodiversity, but people who protect biodiversity have few incentives to offer this protection because they cannot exclude nonpayers from benefiting from that protection.

Markets alone will therefore always undersupply biodiversity. Governments and other actors must use programs and policies to provide the socially optimal level of biodiversity. However, the sheer number of individuals and governments who benefit from biodiversity makes coordination difficult and increase the likelihood of free-riding behavior. Moreover, the beneficiaries of biodiversity protection are often diffuse, while the beneficiaries of alternative uses of biodiversity that leads to its disappearance are often concentrated in small groups that reap large private gains from extinguishing biodiversity. The location of substantial amounts of biodiversity in low-income nations with weak institutions, high discount rates, and pressing social and economic needs only serves to exacerbate the loss of biodiversity.

Further complicating matters, many of the benefits associated with biodiversity protection, such as contributions to global ecosystem functions, the potential for pharmaceutical discoveries, and the existence of charismatic species, accrue to people who are far removed from the sources of biodiversity in developing nations. Without institutions that can transfer some of the global value of protecting biodiversity to local and regional decision makers who bear much of the cost in protecting biodiversity, little progress is likely to be made in stopping the decline in biodiversity in developing nations in the foreseeable future.

II. Targeting Scarce Conservation Funds

Given the limited funds for biodiversity protection, spending conservation funds in a way that ensures that each dollar goes as far as it can in achieving conservation objectives is essential. To allocate conservation resources efficiently, practitioners and policymakers must necessarily integrate benefit and cost data to make good decisions. This advice holds true whether benefits are measured in dollar values or in physical values (e.g., species or ecosystem attributes).

Economic analysis may not be the driving force behind determining the goals of biodiversity conservation investments but cost-effectiveness analysis should inform the decision of where to make those investments (Tacconi and Bennett, 1995).

When prioritizing biodiversity protection investments, however, academics and advocates often focus solely on the benefits that each parcel contributes toward the policy objective, while government agencies often focus solely on acquiring land as cheaply as possible with only a vague notion of the benefits provided by each acquired parcel.[1] In a study of prioritizing investments for biodiversity conservation in the United States, biologists (Dobson *et al.*, 1997) found that endangered species in the U.S. were concentrated spatially and suggested that conservationists focus their investments on a small number of geographic areas. Economists (Ando *et al.*, 1998) responded by pointing out that variability in economic factors was just as important as ecological variability in efficient species conservation. Ando *et al.* found that given a target of conserving 453 endangered species, the approach that considers both economic and ecological variability cost less than one-sixth the cost of the approach that only considers ecological variability. A similar debate developed over ecosystem conservation investments at the global scale (Mittermeir *et al.*, 1998; Balmford *et al.*, 2000). Cost-efficient conservation strategies were also examined by Polasky *et al.* (2001) for the case of species conservation in Oregon. They demonstrated substantial gains could be realized if policymakers considered both costs and benefits simultaneously rather than just costs or benefits alone. Such gains are

[1] Examples of conservation approaches characterized by seeking out the cheapest land can be found in the first nine contract sign-ups of the U.S. Conservation Reserve Program, which attempted to maximize the contracted land area given the available budget, and the establishment of protected areas in Madagascar before 1990, which were overwhelmingly located in steep, marginal lands that were far from infrastructure (Green and Sussman, 1990). Even The Nature Conservancy (TNC), a well-known conservation group, found itself in a situation in which it had been emphasizing maximum land acquisition given its budget. When new TNC president Steve McCormick asked his staff to explain to him how TNC was successful (Knudson, 2001), they responded with the number of acres TNC had protected: “And I say, 'OK, but how does that translate into the preservation of biological diversity? How does it accomplish our mission? And they can't tell me. "

particularly important when one realizes that economic costs are often positively correlated with political conflict and that reducing political opposition to protecting biodiversity can be just as useful as using monetary budgets efficiently.

Although consideration of costs and benefits simultaneously leads, by definition, to more cost-efficient environmental policy outcomes, data collection and analysis can be expensive. Ferraro (2003) illustrates how the correlation and the relative heterogeneity of costs and benefits across the policy landscape determine the magnitude of the potential gains from integrating cost and benefit data in policy design and analysis. In a specific allocation to biodiversity conservation, he argues that biodiversity conservation efforts would benefit from more investment in research that estimates the costs of biodiversity protection across the globe.

When the tradeoffs between different types of environmental services and biodiversity are not expressed in dollar terms, an alternative approach to evaluate biodiversity investments is multicriteria analysis (MCA). MCA enables analysts to find ranges over which the tradeoffs between two desired services or biodiversity types are particularly difficult or simple. Any set of parcels that does not fall on the tradeoff curves defined in MCA are necessarily inefficient from an economic perspective; the system of protected lands could achieve higher levels of benefits for the costs incurred. In Papua New Guinea, for example, one study compared a plan to achieve a goal of representing 10-percent of all vegetation types to a plan to achieve a biodiversity target using a trade-offs based approach with a timber volume index and found that tradeoffs based plan achieved the biodiversity target and cost 7% less than the percent-target plan, which achieved only 70% of the biodiversity target (Faith, et al., 2001).

A final difficulty associated with cost-efficiently targeting scarce conservation funds across the landscape is the dynamic nature of the threats to biodiversity. Conservation funds are raised and disbursed over time. Given the irreversibility associated with ecosystem conversion and species extinction, decision makers are faced with the question, "Should current funds be spent on species and ecosystems that are likely to disappear soon or should they be spent on species and ecosystems that are not in danger of disappearing anytime soon?"

At first glance, the choice seems obvious: protect the most endangered. This is in fact the approach of many conservation organizations (e.g., "hotspot" prioritization by Conservation International; Ecoregion 200 prioritization by World Wildlife Fund). However, the most endangered species and ecosystems are also often the most expensive to save. They are endangered precisely because there is much value derived from extinguishing them. A conservation agent who annually allocates funds to the most urgent cases may find that, at the end of several decades, fewer species and ecosystems were protected in comparison to the approach of a conservation agent who allocated funds to the many more ecosystems and species that are cheaper to protect because they are not yet endangered. Costello and Polasky (2002) present a formal way of thinking about making sequential conservation investments in the face of potential irreversible losses.

III. Policies to Conserve Biodiversity

Despite the obstacles to raising conservation funds and targeting them efficiently across the global landscape, there has been widespread experimentation with policies to conserve biodiversity in developing nations. We discuss the most popular and promising of these policies below. However, for reasons we discuss at the end of this section, there have been few *empirical* analyses of these policies. Thus, much of our evaluation of their effectiveness is based on theory, simulations, rough case studies, and anecdotes. Even within developed nations, there is a paucity of empirical work that can guide implementation in developing nations. The evaluation of biodiversity conservation policy therefore lags substantially behind evaluations of other social policies (e.g., health, crime, labor). Advances in biodiversity policy evaluation represent one of the most critical needs in biodiversity conservation at the beginning of the 21st Century.

A. Protected Areas

Defining areas as "protected" and establishing restrictions on their use is the most common policy to protect biodiversity worldwide. In this classic attempt to provide a public good through government fiat, biodiversity is supplied through "fences and fines." In 2003, 10.8 percent of the earth's terrestrial area was designated protected, including 12.6 percent of the land area of developing countries (EarthTrends Data Tables, 2003). Margules and Pressey (2000) state that "reserves alone are not adequate for nature conservation but they are the cornerstone on which regional strategies are built. Reserves have two main roles. They should sample or represent biodiversity of each region and they should separate this biodiversity from processes that threaten its persistence." Myers, however, reports that only 37 percent of the earth's 25 biodiversity "hotspots" (threatened areas with high rates of endemism) are in protected areas (Myers, 2003). Thus, although establishing protected areas is the most common policy to protect biodiversity, much of global biodiversity is outside of protected areas.

As discussed in the previous sections, conservation funds are limited and thus care must be made in allocating them. Which areas to include in a park or reserve system is a complicated choice that should reflect, among other things, the regional distribution of biodiversity, opportunity costs of the land use restrictions, and the impact on rural people. Reserve site selection algorithms exist that determine sets of protected areas that achieve conservation goals under specified constraints (e.g. Rosing et al. 2002). Although these algorithms are rarely implemented in developing country settings, they provide a framework for identifying tradeoffs based on the available data and the conservation goals. From the perspective of economics, two factors are critical in determining which areas to include in a protected area system: the goals of the reserve system and the cost effectiveness of plans to attain those goals. In the previous section, we discussed aspects of cost-effective targeting of conservation funds. Thus in this section, we focus on delineating the goals of protected area system.

Many protected area policies to date are based on conserving fractions of land area rather than conserving levels of biodiversity. The UNEP World Conservation Monitoring Centre (WCMC, 2002) reports, for example, that the IV World Congress of Protected Areas set a target of 10% of the earth's land area for conservation. Such area targets say nothing about how much biodiversity is being conserved (Pressey, 1997; Barnard, et al. 1998). In practice, such area targets have often lead to systems of reserves that are made up of lands that are readily available or have low opportunity costs rather than lands that are true priorities for conservation. In some cases, degraded land that provides few biodiversity or environmental service benefits is included in these systems.

Even when protected area establishment is guided by biological criteria, decisions about which land parcels to include and what degree of restriction to place on their use require assessments of tradeoffs. The IUCN defines 6 categories of protected areas, each with a different degree of restrictiveness, beginning with "strict nature reserves" with no uses other than scientific research permitted (including tourism) followed by "national parks" which permit recreational activities and ending with areas that are "managed for sustainable use of natural ecosystems" with various extractive uses permitted (IUCN, 2003). The existence of these categories highlights a fundamental issue with protecting biodiversity in developing countries: people often rely on resources within protected areas and human uses may not be compatible with biodiversity protection. For example, allowing the extraction of fruits might degrade an area somewhat and decrease the amount of genetic diversity, but it may improve local human welfare and help to protect ecosystem diversity. Thus, even with an explicit goal of protecting biodiversity and ecosystem services there are important tradeoffs between the types and levels of biodiversity and services that are associated with different patterns of conservation land and with different restrictions on use of that land.

Designing effective systems of protected areas also requires recognizing the spatial aspects of the reserve site configuration. The biological conservation literature is strewn with debate concerning whether a reserve system protects more species if the system contains a few large protected areas versus many small areas. The ratio of edge to area is also important for minimizing detrimental disturbances from outside the protected area. Connectivity and interactions across sites are also important, particularly in cases where expanding the size of individual PAs would prove particularly difficult (Beier and Noss, 1998; Sutcliffe and Thomas, 1996; Bennett, 1999). In many cases, spillover benefits from contiguous land parcels and minimum area thresholds add further complications to designing effective reserves.

Economists have rarely weighed in on the debates about the spatial aspects of protected area configuration, preferring to allow biologists to make the decisions. Economists, however, have tools that could contribute to elucidating how reserves should be designed. Albers (1996), for example, created a framework for land conservation decisions in which contiguous blocks of conserved land create a bonus value. Although earlier economic analyses of protected area establishment focused on the simple question of whether to protect an entire area or not (e.g., Dixon and Sherman, 1990), more recent work has focused how much of an area to conserve and at what level of use restriction. For example, Albers (2000) takes park-level valuation data for Khao Yai National Park in Thailand, divides the values across park sub-plots based loosely on geographic information, and looks at how management decisions change when the protected area is zoned. Turner, et al. (2003) and others have begun to look at how biodiversity and ecosystem services change with different land use restrictions, which provides useful information for determining the tradeoffs between conservation goals and land uses.

Economics also has a role to play in helping biologists determine what kind of research would be most useful for making decisions. For example, few biological studies provide information about the changes in probabilities of species survival or other characteristics of biodiversity with marginal changes in the configuration or management of a protected area network, but such information, when incorporated into economic models, could be extremely useful for policymakers and practitioners.

Finally, protected area networks must adapt to changes in economic and biological circumstances over time. Thus site selection should reflect the likelihood of changes within and between the sites that may alter those natural processes. Perhaps most pressing in this regard is the impact of climate change on biodiversity. Selecting configurations of reserves that will allow species to move in response to climate change provides one example of how dynamic considerations can be brought into reserve site selection decisions. Conservation biologists have begun to create maps of potential habitats at various points in the future with climate change and some have investigated how biodiversity might be expected to move as climate changes (e.g. Thomas et al. 2004; Peterson 2002). This kind of ecological information should be brought into site selection decisions to promote long-run biodiversity conservation in the face of change. To date, however, little economic policy analysis, or even feasibility analysis, exists to determine what policies could be implemented to allow for the ecological transitions predicted by natural scientists.

After establishing their protected areas, government landowners must enforce their property rights against those who seek to use the protected biodiversity for alternative uses. As discussed in other chapters in this volume, property rights are often poorly defined and under-enforced in developing countries, and legal systems often fail to adequately support property rights even when they are well defined. For a case study in India, Robinson (1997) demonstrates that imperfect enforcement of a public property right leads, over time, to the complete encroachment of the land. In fact, much of a protected area's budget is often spent on enforcing property rights rather than on other aspects of managing the protected area.

Most protected area managers in developing countries attempt to enforce against land conversion, hunting, and resource extraction within the protected area by patrolling the area and fining (or killing) extractors. Abbott and Mace (1999) collected spatial data on where people extract from forests in Malawi and where the patrols are, and found that fines are so low that they do not deter extraction. In an empirically-inspired model, Albers (2003) finds that, for a given budget, larger areas of biodiverse land can be protected if patrols are allocated across space in a manner that reflects the distance costs faced by extractors.

Economic theory on crime (Becker, 1968) offers additional insights into improving the effectiveness of protected area enforcement. Managers can alter the penalty and the probability of detection (in areas in which managers do not have the authority to penalize, they must depend on other agents who can affect the probability of prosecution and conviction if detected). Given the limited budgets of protected area managers, an attractive approach would be one in which patrols were few but the fines of non-compliance were high. Unfortunately, empirical work on tax compliance (e.g., Alm et al. 1995) and fishery law compliance (Furlong, 1991) in developed nations suggest that increasing the probability of detection has a much stronger effect on compliance than does the expected penalty. Beyond these studies, however, relatively little economic modeling or empirical work exists to inform decisions about enforcement of property rights within protected areas to conserve biodiversity. In particular, deadly force is being increasingly applied to wildlife poachers in developing nations (Messer 2000; Mbaria and Redfern 2002), but very little analysis of its effectiveness has been conducted to date.

In many areas of the world, the problem of enforcing property rights stems from the fact that establishing protected areas, like establishing any conservation policy, typically involves curtailing consumptive uses of resources. Given the traditional reliance of local people on natural resources such as fuelwood and wild foods, it is not surprising that some authors have estimated that local people incur substantial losses when protected areas are established (see Ferraro, 2002 and references therein).[2] These losses generate conflict that jeopardizes the achievement of the protected area's objectives and thus protected area managers and other conservation organizations have tried various initiatives to "bring local people on board." These initiatives may be implemented within park boundaries or in the neighboring villages and generally involve creating economic incentives to reduce extraction (Wells and Brandon, 1992). We discuss these incentives in section III.C.

We now turn to a more fundamental question: are protected areas effective in achieving conservation goals? Despite how common and long-lived the use of protected areas in biodiversity conservation has been, we have surprisingly little quantitative data on the subject of

[2] The common wisdom that local people incur substantial losses, however, is based on either *ex post* or *ex ante* extrapolations. No one has conducted a "before-and-after" impact assessment of a protected area's establishment.

whether they work or not. In the last two decades, a debate has developed with one group arguing that the "fences and fines" approach has failed in developing nations (Brechin et al., 2002) and another side arguing that protected areas remain one of the best hopes for protecting biodiversity in developing nations (Oates, 1999; Terborgh, 1999; Brandon, 12002; Bruner et al. 2001). Among the proponents of protected areas, there is a camp of those who believe current parks are "paper parks" and thus are ineffective without more money (Oates, 1999; Terborgh, 1999; van Schaik et al., 1997) and those who believe that despite funding limitations, existing protected areas have been quite effective in protecting biodiversity (Bruner et al. 2001).

There are few empirical studies to inform this debate. Bruner et al. (2001) acquired data obtained from surveys of protected area managers or researchers associated with 93 protected areas around the world. Using simple partial correlation coefficients to determine whether protected status affected the level of (self-reported) conservation outcomes within and outside the protected areas, the authors concluded that clearing, grazing and burning are lower and the abundance of game and commercial tree species higher within parks than in the adjoining 10-km wide buffer. Cropper, et al (1999) used econometric analysis of a GIS dataset for Thailand and found that land use conversion is lower in nature preserves than in parks, perhaps because of the more restrictive nature and additional enforcement of those restrictions in preserves.

The fundamental problem in evaluating the effectiveness of protected areas is selection bias: there is evidence that many protected areas are located in areas that are not at risk for large-scale ecosystem perturbation (e.g., Green and Sussman, 1990). In other words, protected areas, for political and economic reasons, are often located in areas with few profitable alternative uses of the ecosystem and thus even without protected status, one might not see much degradation in the protected area over time. Furthermore, no one has examined if the establishment of protected areas in developing nations has lead to increased pressures on other non-protected ecosystems (some empirical work on this topic has been done in the U.S.; e.g., Berck and Bentley 1997).

B. Private Provision of Biodiversity

Although establishing protected areas has been a common approach by developing nations to protect their biodiversity, another common approach has been to simply do nothing and depend on the private provision of biodiversity. Although biodiversity protection is a global public good, free riding is not complete because there can be private benefits from actions that, purposely or inadvertently, lead to biodiversity protection. Such actions are common in developed nations. For example, large landowners like Ted Turner own large areas of undisturbed land for their personal use but, in the process of maintaining the land as undisturbed, they provide the public good of biodiversity protection. Non-governmental organizations like The Nature Conservancy (TNC) depend on voluntary contributions from member and other donors to privately provide biodiversity through habitat acquisition.

Despite having to contend with poorly defined property rights and enforcement, pprivate landowners and non-profit groups also conserve habitat and species throughout the developing world. Some protect biodiversity for financial gain and others for personal satisfaction, or a mix of both (see, for example, Langholz et al.'s (2000) analysis of motivations of private protected area managers in Costa Rica).

In some regions, private provision of biodiversity protection occurs because firms or landowners are able to capture some of the willingness to pay for the public good by bundling it with private goods (Heal, 2002a). For example in Zimbabwe, ecotourism firms buy land or create incentives for landowners to restore and maintain habitat for the "charismatic megafauna" (e.g., lions, elephants) that many tourists pay to see on hunting and photographic safaris. By fencing off the property and providing habitat on private land, these individuals and firms provide biodiversity conservation because they capture a significant fraction of the international willingness to pay for that biodiversity. Heal (2002b) notes that southern Africa's property rights system encourages biodiversity conservation because an animal on a plot of land belongs to the landowner. Instead of pitting incentives to protect their crops against regulatory disincentives (such as fines for injuring or killing an animal), this system creates incentives for farmers or grazers to capture the animal (often paying a professional for this service) in order to sell the animal to private game reserves for quite large sums of money. In this way, private game reserves protect biodiversity both within and outside the reserve. Langholz and Lassoie (2001) document a substantial increase in the number of private protected areas worldwide.

Other privately-owned land provides biodiversity protection serendipitously because the mode of production is biodiversity-friendly. For example, shade-grown coffee, especially traditional styles of production that maintain nearly closed forest canopies, can provide habitat for a wide range of both flora and fauna. In fact, Mexico's shade coffee plantations provide critical habitat to a large number of migratory birds. Similar systems include other understory crops such as cacao and bananas or planted agroforestry systems or multiple economic species that still provide some habitat. The landowner's incentives to undertake these modes of production may be factors such as the lack of fertilizer and other expensive inputs, the lack of large initial clearing costs, the use of labor-intensive rather than capital-intensive production, credit constraints that prevent the landowners from greatly modifying their land, or simply tradition. In these cases, the farmers do not capture any of the public's willingness to pay for the biodiversity protection they provide on their land (although attempts are being made to capture part of this willingness to pay; see next section).

Other potential sources of private incentives for biodiversity protection include “bio-prospecting,” the term for the search among diverse natural organisms for commercial products of industrial, agricultural, or pharmaceutical value. A few contracts have been struck between pharmaceutical firms and the government or private agents who control biodiverse ecosystems and at least one paper claims that the value of protecting certain ecosystems for bioprospecting can be quite high (Rausser and Small, 2000). However, other analysts have concluded that the value of biodiversity for bioprospecting is quite small (Simpson et al., 1996) and, most tellingly, the large number of private partnerships originally envisioned by bio-prospecting proponents were never realized.[3]

Although the actions of private agents can contribute to the provision of biodiversity protection, these actions by themselves will not lead to the “optimal” level of biodiversity protection. When mechanisms for capturing the global willingness to pay for biodiversity are absent or incomplete, outside incentives for decision makers who have de facto control over the fate of ecosystems and species will be necessary.

C. Economic Incentives: Indirect

In our discussion of protected areas and the private provision of biodiversity, we highlighted that conservation practitioners and policymakers have turned to creating economic incentives to reduce resistance to conservation goals among residents around protected areas and to induce potential “eco-entrepreneurs” to provide more biodiversity than they provide under

[3] Costello and Ward (2003) point out that the parameter values chosen in different models of bioprospecting can make huge differences in the estimated profitability of such activities.

prevailing private incentives. When done correctly, the incentives align the public's interest in protecting biodiversity with the private interests of those who control the fate of biodiversity. When done poorly, however, such incentives either have no effect on biodiversity protection or, worse, exacerbate the threats to biodiversity (Wells and Brandon, 1992).

Perhaps the most common initiatives aimed at discouraging biodiversity depletion in developing countries are development projects—often called Integrated Conservation and Development Projects (ICDP) or Community-based Natural Resource Management (CNRM) initiatives—located at protected area boundaries or in ecologically sensitive areas. In general, these projects attempt to create a conservation incentive in an *indirect* way through three mechanisms: (1) by re-directing labor and capital away from activities that degrade ecosystems (e.g., agricultural intensification); (2) by encouraging commercial activities that supply ecosystem services as joint outputs (e.g., ecotourism); or (3) by raising incomes to "reduce dependence" on resource extraction that degrades the ecosystem (Ferraro 2001). These mechanisms, however, may not be powerful and may backfire in many settings.

To examine these mechanisms, several studies use household production function models, which were developed to examine decisions of rural households in regions where markets are thin or missing (Singh, Squire, and Strauss, 1986). With re-directing labor or "conservation by distraction", an agricultural project, for example, may not reduce the labor allocated to the degrading activity if people can be hired to take advantage of the opportunities the project provides (Muller and Albers, 2003). Agricultural intensification is likely to lead to reduced pressures on ecosystems only in the special case in which residents are subsistence agriculturalists (Angelsen, 1999; even then, one requires the assumption that markets will not develop in the presence of agricultural surpluses).

Encouraging the private provision of biodiversity through support for eco-friendly commercial activities that maintain ecosystem services is another popular form of economic incentive. In these cases, outside aid is often directed towards increasing the eco-output price or facilitating the acquisition of complementary inputs, such as tourism infrastructure, product marketing, and processing facilities. In some cases, the incentive may be successful on a limited basis, but rarely is the demand for eco-outputs, such as for ecotourism or non-timber forest products, large enough to support more than a small fraction of the local population.

Some success has been realized in efforts to market products as "green" or wildlife-friendly (e.g., shade-grown coffee) and thereby generate a price premium in international markets. If the landowner receives the price premium, it increases his incentives for production through a technology that encourages biodiversity protection. The price premium thus serves as a mechanism with which to capture the broader public's willingness to pay for biodiversity protection and thus encourage its continued protection, but some have criticized the approach for being an inefficient mechanism to transfer funds from beneficiaries to suppliers of biodiversity protection (Ferraro et al. 2003).

Raising incomes leads to conservation only if the extracted products are "inferior" goods that are replaced by other, preferable and, by serendipity less degrading, goods as incomes rise. We know of no empirical evidence that suggests that increases in income in developing nations will lead to more biodiversity protection. In fact, the empirical evidence from developing nations suggests otherwise: increased incomes, particularly when investment opportunities are limited to agriculture, leads to increased conversion of habitat and thus biodiversity loss (Foster et al. 2002; Zwane 2002).

Economic analyses of indirect incentives also reveals other problems with these policies such as implicit assumptions about local people's desire to be nature's stewards, complex issues in implementation, inefficiency and lack of conformity with the temporal and spatial dimensions of biodiversity conservation objectives (Ferraro et al. 1997; Brandon, 1998; Southgate 1998; Chomitz and Kumari 1998; Simpson 1999; Ferraro 2001; Ferraro and Simpson 2002; Terborgh and van Schaik, 2002, Muller and Albers, 2003). Despite their widespread use, many assessments of indirect conservation policies demonstrate rather limited success in achieving their conservation and development objectives (Wells and Brandon 1992; Ferraro et al. 1997; Wells, et al. 1999; Oates 1999; Ferraro, 2001; Terborgh et al., 2002). Salafsky et al. (1999) investigated three years of financial data from 37 eco-enterprises subsidized by the USAID-funded Biodiversity Support Program. They found that that the vast majority failed to cover their costs. As with the case of protected areas, however, there has been little formal empirical work in evaluating the effectiveness of indirect incentives on the achievement of biodiversity conservation objectives.

D. Economic Incentives: Direct

An alternative approach to encouraging the conservation of endangered natural ecosystems is to pay for conservation performance *directly*. In this approach, domestic and international actors make payments in cash or in kind to individuals or groups conditional on specific ecosystem conservation outcomes (Ferraro, 2001; Ferraro and Kiss, 2002).

In high-income nations, tax incentives, easements, and tradeable development permit programs are widespread and useful for inducing private agents to conservation land and biodiversity voluntarily. Despite some issues with the potential for "conservation overkill," full-interest acquisitions (or "fee-simple" acquisitions) are the most institutionally straight-forward of all the conservation payment mechanisms and the costs of monitoring and enforcing an agreement are relatively low (Boyd, Caballero, and Simpson, 1999). Conservation easements provide a payment or tax deduction to landowners who extinguish their rights to future land development. Monitored by the conservator, easements involve complex contracting issues but are a legal mechanism with a well-established legal pedigree. Tax credits or other subsidies equal to the difference in value between developed and un-developed uses can also leave land and its biodiversity protected but require monitoring and a well-developed tax administration (Boyd, Caballero, and Simpson, 1999). Tradable development rights (TDRs), which require a restriction on the amount of land that can be developed in a given area, leads to the least-cost development restrictions but are institutionally complex and, as with tax incentives, do not allow for targeting of particularly biodiverse areas.

Many low-income countries do not have the legal, property right, and tax institutions to make considerable use of these direct incentives for biodiversity conservation. Still, other methods of direct incentives for conservation, usually in the form of a payment, are underway in several developing countries. Examples include forest protection payments in Costa Rica, conservation leases for wildlife migration corridors in Kenya, conservation concessions on forest tracts in Guyana, performance payments for endangered predators and their prey in Mongolia, and "contractual national parks" in South Africa and American Samoa (Ferraro and Kiss, 2002).[4]

Proponents of the direct payment approach argue that such an approach is preferable to indirect approaches because it is likely to be more effective, cost-efficient, and equitable, as well as more flexibly targeted across space and time (Simpson and Sedjo 1996; Ferraro 2001; Ferraro and Simpson, 2002, 2003; Ferraro and Kiss, 2002). Payments can be made for protecting entire ecosystems or specific species, with diverse institutional arrangements existing among governments, firms, multilateral donors, communities, and individuals.

However, direct payments have also been criticized. They may transfer property right enforcement responsibilities to local participants, which can lead to inter and intra-community conflict. They, like indirect interventions, also require on-going financial commitments to maintain the link between the investment and the conservation objectives.[5] Large sums of financial transfers can exacerbate existing corruption problems and payments to individuals who are threats to biodiversity can lead to a perverse outcome in which individuals attempt to become a threat in order to receive conservation payments. Other authors (e.g., Swart 2003) worry that by tying conservation outcomes to financial transfers, one loses the moral foundation on which a sustainable conservation ethic can be built. In many settings, a combination of incentives and disincentives, and of indirect and direct mechanisms may prove best (Muller and Albers, 2003). For example, the World Bank's Conservation Trust Fund and Conservation International (CI)'s conservation concession combine direct and indirect approaches.

Although economists have begun to weigh in on the issues surrounding direct payment incentives through theory and simulations, no one has conducted a rigorous and systematic empirical evaluation to assess if an existing direct payment initiative is achieving the conservation and development objectives it purports to achieve. Carefully designed, empirical research on the use of conservation payments to achieve conservation and development goals in low-income nations is a critical next step.

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[4] For more details and examples, see (Kiss, 2003) and http://epp.gsu.edu/pferraro/special/special.htm.

[5] We note, however, that social programs in which families are paid for sending their children to school (instead of allowing them to work or skip school) has become a popular and successful program in many Latin American nations (Dugger, 2004). Thus, in a sense, the issue of "sustainability" is not one of creating self-financing conservation initiatives or waiting until one has a fully-financed trust fund capable of making payments well into the future. Instead it is more accurately characterized as an issue of a durable constituency and political will.

E. Factors that Affect Incentives

The efficacy of any incentives that are introduced, whether direct or indirect, are strongly affected by the particular market and institutional setting (Muller and Albers, 2003; Robinson, et al. 2002). As discussed elsewhere in this volume, markets are notoriously thin or missing in many developing countries. From a theoretical perspective, the impact of improved market access on forest degradation and biodiversity is ambiguous (Omamo, 1998; Key, Sadoulet, and de Janvry, 2000; Robinson et al., 2002). Without access to markets, most resource use will be for home consumption (Sierra, 1999). As market access increases, the impact on the resource base, whether positive or negative, depends on the relative strength of two effects. Some households will increasingly switch from purely subsistence extraction to commercial extraction, whereas other households, especially those with high opportunity costs of labour, may choose to purchase forest resources from the market rather than extract, using their labour for alternative activities (Robinson et al., 2002).

In addition, policies or programs that improve market access to create economic incentives will typically interact with the distribution of labor opportunity costs and so forest managers need to look beyond the pre–policy degree of market interaction to predict a policy's impact on resource extraction patterns (Robinson et al, 2002). The creation or improvement of roads allows a policy maker to reduce market access costs directly (Bluffstone 1993; Cropper, Griffiths, and Mani 1999; Imbernon 1999). Resource use incentives change because the roads both reduce the cost of accessing and removing resources from threatened ecosystems. Working in the opposite direction, the same roads also reduce the cost of accessing substitutes for forest resources (Robinson et al., 2002). The creation of roads also changes opportunities for labor, which may alter resource management decisions (Muller and Albers, 2003).

One should also recognize that the use of incentives to induce the “voluntary” provision of biodiversity does not necessarily require, or allow, the abandonment of traditional regulations and enforcement. In fact, most of the incentives discussed in the previous two sections require a strong institutional setting in which rights and responsibilities can be allocated and enforced. Restrictions on resource use may still be required given biological and economic uncertainty, asymmetric information between those who provide and receive the incentives, biological thresholds and non-linear responses to resource use, and the need to induce private agents to innovate in the biodiversity provision “market.”[6] Bowen-Jones et al (2002) argue that a combination of controls and incentives will be more cost effective than relying on one or the other, and hence sustainable in the long run.

Economic incentive-based responses alter the relative value of opportunities or constraints and thereby induce change in actions coming from a decision process. How effective a given incentive will be, whether direct or indirect, is determined by the value of the range of possible activities, the market setting, the institutional setting, and other constraints faced by the decision-maker. In some cases, a response may induce radically different reactions in two settings with dissimilar institutions. The effectiveness of economic incentives for inducing biodiversity conservation is, therefore, strongly dependent on the setting in which the decision is made.

[6] In an empirical analysis of voluntary pollution abatement, Uchida and Ferraro (2003) note that the regulatory pressure in an industry was a strong factor influencing the comprehensive of “voluntary” overcompliance by firms.

F. Invasive Species and Biodiversity Protection

Invasive species have been identified as one of the main proximate causes of extinctions (Glowka et al. 1994). Invasive species disrupt important ecological functions and such disruption has substantial implications for economic activities (Heywood 1995). The growth in the global frequency and abundance of invasive species mirrors the growth in global trade, transport and travel. By the end of the 20th century, most ecosystems had been affected by invasions (Williamson 1996, Parker et al. 1999). The economic implications of these ecosystem invasions, however, have yet to be identified (as opposed to the direct costs to agriculture, transportation and recreation; Perrings et al. 2000).

The major approaches to dealing with invasive species and disease are the same within and outside of protected areas: prevent invasion; manual, chemical, and biological agents to eradicate invasive species; and containing the invasive species at some predetermined level. Macro-level initiatives such as trade and transportation regulations have also been applied (Costello and McAusland 2002). Although there are studies that compare the cost-effectiveness of invasive species policies in high-income nations (e.g. Leung, et al. 2002), we know of no such studies for low-income nations.

Among the more important problems for addressing the invasive species problem is that invaders are usually not identified until after they have become well established and thus costly to eradicate or control. Moreover, invasions are associated with a high degree of uncertainty both because they involve novel interactions, and because invasion risks are endogenous. Actual risks depend on how people react to the possibility of invasions (Perrings et al. 2002). Finally, preventing and controlling invasions is problematic because such prevention has characteristics of a "weakest-link" public good (Perrings et al. 2002). The provision of such a public good is largely a function of the actions of the least effective provider. Given that developing nations are likely to be the weakest link and they are typically spatially concentrated, solutions to the biological invasion problem in developing nations will be difficult without international institutions that support research and provide incentives for governments and citizens to prevent and control invasive species in developing nations.

Efforts to protect biodiversity from invasive species requires coordination between ecologist and land managers but this type of communication is notoriously lacking even in high income countries (Eiswerth and Johnson 2002). Most economic policy analysts focus on the dynamic aspects of invasion and policy but the spatial aspects of invasion are increasingly recognized (Leung, et al. 2002; Kaiser and Roumasset, 2002). To protect ecosystems from disruptive invasive species, policy must be based on the dynamic and spatial aspects of invasion but such analyses are in the nascent stages. For the protection of biodiversity, appropriate siting and sizing of protected areas paired with restrictions on land use in neighboring areas may eliminate or limit invasion pathways.

G. Wildlife Damage and Conflict

Human conflict with wildlife is a significant conservation problem around the world (Thirgood and Woodruff, forthcoming). The cost of conserving large and sometimes dangerous animals is often borne disproportionately by rural residents who live closest to wildlife. The risk of wildlife damage to crops, livestock, and human lives provides incentives for rural residents to kill wildlife and to reduce the quantity and quality of habitat on private and communal lands. Conservationists have attempted to reduce these incentives by spreading the economic burden of wildlife damage and moderating the financial risks to people who co-exist with wildlife.

Popular initiatives include compensation to rural residents for the costs of wildlife damage and the introduction of private and public insurance programs. For example, a non-governmental organization in the United States has a program of compensating ranchers for wolf attacks on livestock. According to government agencies who are in charge of wolf recovery efforts, the livestock compensation program has made wolf recovery more tolerable to livestock producers and has made wolf recovery more easily attainable (Nyhus et al. forthcoming).

Few systematic efforts, however, have been made to evaluate the efficacy of these programs or the best way to implement and manage these schemes for endangered species (Sillero-Zubiri and Laurenson; Nyhus et al., 2003). A recent theoretical and empirical review of these compensation and insurance initiatives (Nyhus et al. forthcoming) points to difficulties in implementing them in any nation, but particularly in developing nations. These difficulties include potential perverse incentives that could lead to greater losses of biodiversity than observed under status quo conditions and obstacles to creating targeted insurance schemes in low-income nations. The authors note that alternative approaches may be more effective in many areas: promoting trophy hunting, building wildlife barriers, providing additional habitat, moving people away from wildlife, or making explicit payment to rural residents that are conditional on wildlife abundance. None of these alternative methods, however, have been empirically evaluated in the field.

H. Paucity of Empirical Work

A common refrain in the previous sections was that little is known about the effectiveness of many of the policies we examined. The lack of clear results demonstrating success or failure of a given initiative (e.g., price premiums for eco-friendly products) is not unique to developing nations, but it is particularly glaring in such nations. A recent workshop on International

Conservation Finance (UC-San Diego, Dec 2-4) noted that conservation practitioners and donors lag behind their peers in other policy fields (e.g., poverty reduction, job training, criminal rehabilitation, public health) in terms of having well-designed empirical analyses of program effectiveness. The workshop concluded with a consensus agreement among participants that a critical conservation need in developing nations was a substantial increase in well-designed efforts to determine what works and when.

Given that hundreds of millions, if not billions, of dollars have been spent on conservation activities in developing nations over the last two decades and that international research support has been substantial in these efforts, one may wonder how it is that empirical results are lacking. We do not claim to have conducted a formal study on this topic, but our joint experience in the field leads us to several conclusions.

The first and most obvious constraint to well-designed empirical analyses is the lack of conservation researchers who are trained in state-of-the-art empirical program evaluation techniques. Such techniques include randomized field experiments, matching methods and sophisticated econometric analyses. In fact, we know of only one published paper that used such methods to assess a conservation policy in a developing nation. Edmunds (2000) used instrumental variable regressions and propensity score matching methods to assess the effectiveness of devolving control over forest management to local community groups in Nepal. He found that such devolution increased the average availability of fuelwood for local communities.

Moreover, much of conservation investments in developing nations are framed as projects that "test" an idea in one or several locations. Data collection in these locations is often poor or non-existent and control locations, in which no intervention is attempted, are never formally selected. Without adequate data and controls, one is left with only guesses and vague anecdotes about the effects of the program intervention on the conservation outcome of interest. Donors who fund these projects typically know little about program evaluation methods, and the practitioners who implement the projects typically have few incentives for careful analysis and falsification of hypotheses. Thus there is rarely funding available for more careful policy interventions and analysis.

Furthermore, for empirical analysis to bear fruit, policy interventions cannot be varied in complex ways across space and time. If every village or household is exposed to a different intervention (one gets direct payments, one gets nothing, another gets fish farms, a fourth gets agricultural assistance, etc.) then an analyst is left with few observations for every intervention and thus cannot make any inferences about their effectiveness. We are not proposing that all policy interventions be uniformly applied across space and time, but we are arguing that "experimental" introductions of policy interventions should be conducted in a way that allows practitioners and decision makers to make inferences about their effectiveness.

IV. Conclusion/Summary

Scientists report that biodiversity supports critical ecosystem functions and the provision of ecosystem services in a fundamental way. Many governments and policy communities recognize the importance of biodiversity and its value to society. Given that value, the unprecedented loss of biodiversity in this century signals a serious failure of policy to protect biodiversity. This chapter has focused on the using economic analysis to to develop better policy portfolios for biodiversity conservation.[7]

[7] Our analysis concentrated on in situ biodiversity protection, but ex situ conservation through gene banks and captive breeding programs also have a role to play in protecting biodiversity. Economics has not yet been brought to bear on contrasting ex situ versus in situ conservation, not has it yet helped to inform ex situ programs in allocating their limited budgets.

From an economist's perspective, the undersupply of biodiversity conservation globally results from the public good nature of biodiversity and the fact that many of its benefits accrue to communities far from the places in which biodiversity is located. Biodiverse countries rarely capture the full international social value of the biodiversity they contain (see Bulte's chapter, this volume). In turn, the governments of these countries often fail to create policies that allow the people who bear the costs of biodiversity conservation to capture much of the non-local social value of the local biodiversity. In addition, when the benefits of conservation are not captured as cash, it may be difficult for low-income countries to incur the costs of that conservation. The public good character of biodiversity thus encourages the underprovision of biodiversity conservation at the international, national, and local level.

In addition to this public good problem, several other characteristics of biodiversity complicate the formation of policy. First, the value of biodiversity is largely unknown; most species have not yet been identified or studied, the relationship between biodiversity and ecosystem services is not well understood, and little is known about how biodiversity responds at the margin to various policies. Decisions about conserving biodiversity are made under a high degree of uncertainty. Further complicating policy analysis, losses of biodiversity are likely to be irreversible. Because information about particular species and about biodiversity's role in general is forthcoming, this combination of irreversible decisions under uncertainty calls for performing more biodiversity conservation to prevent irreversible losses (Arrow and Fisher, 1974; Albers, 1996).

Although many millions of dollars are spent by governments and other institutions to protect biodiversity each year, it appears that the current levels of spending fall far short of levels that would provide the socially optimal level of biodiversity conservation from a global perspective. In addition, decisions about what land to target for conservation and what tools to employ to achieve conservation are rarely made in a cost-effective manner to achieve the most biodiversity conservation per dollar spent. In fact, almost no rigorous social science analysis of biodiversity conservation policy exists to determine which policies are cost-effect in which settings.

Protected areas, or a "fence and fine" approach are the most common policy to promote biodiversity conservation. The policy issues surrounding protected areas fall into two categories: siting decisions and management. Historically, protected area siting decisions have not reflected biodiversity protection itself. In many cases, countries establish targets, such as a percentage of remaining forest land, and set out to meet that target with whatever lands, biodiverse or not, are easily attainable. Protected areas would be a more effective tool in biodiversity conservation if siting decisions better reflected natural science and the goal of maintaining biodiversity rather than the goal of a number of acres of protected land.

In terms of management, protected areas may allow park managers to capture some of the non-local values of biodiversity but that money rarely accrues to the local people who incur the costs of restricted access to natural resources within the protected area. Because local people still have incentives to use, and potentially degrade, the resources within the protected area, protected area management involves patrolling and fining people for illegal extraction. This enforcement of the government property right is quite costly, especially in remote areas of low-income countries where property rights and legal institutions are not always well-developed. Many people argue that the amount spent on enforcement is far too low to deter local resource degradation and biodiversity loss, and that these areas are "paper parks." Other work suggests that protected areas are successful in conserving biodiversity. The argument rages on because no systematic analysis of the effectiveness of protected area management has ever been conducted despite the widespread use of protected areas worldwide. Properly sited protected areas that receive adequate funding for enforcement and for compensating rural people for the costs they bear are likely to be effective in protecting biodiversity but are rare in practice.

Private provision of biodiversity conservation occurs in many areas but is not a large force in developing countries. Still, many environmental organizations, such as The Nature Conservancy, have begun to buy land in developing countries with the express purpose of protecting biodiversity. Private conservation actions face many of the same issues as government policies but may be better at siting protected areas for biodiversity conservation, have access to more funding, and be removed from political pressures.

Policies that create incentives for biodiversity conservation can be used alone or in combination with protected area policies. Many of the policies employed in developing countries attempt to address poverty or other issues and to use those efforts to create indirect incentives for conservation. Projects that create incentives in an indirect manner, however, are not likely to be as efficient in protecting biodiversity as direct incentive policies. In addition, assessments of many indirect incentive projects reveal widespread failure of these policies. Direct incentive programs are less well-tested in developing countries but provide an underutilized option for biodiversity conservation. As with protected area enforcement policy, neither direct nor indirect incentives programs have been subjected to analysis to ascertain when and where they are likely to be cost-effective. Some theoretical analysis suggests that the types of local institutions, the opportunities for labor, and the characteristics of the natural environment all contribute to the effectiveness of incentive programs but no empirical analysis exists to characterize settings in which any particular incentive program is likely to cost-effectively conserve biodiversity.

This chapter focuses on biodiversity as a whole rather than on species in particular and emphasizes land management and habitat protection to achieve biodiversity conservation.[8] Two major threats to biodiversity, invasive species and climate change, are not well-addressed by static land use restrictions. Protecting biodiversity from invasive species will require both monitoring and eradication activities in existing reserves. Some aspects of the siting of protected areas and of the management of buffer or transition zones between protected areas and their surroundings can reduce the opportunities for invasive species to take hold. Similarly, the impact of climate change may be mitigated to some degree if the siting of protected areas and the management of nearby land and wildlife corridors allows species to move and adapt gradually to shifts in climate.

The market failure discussed in Erwin Bulte's chapter (this volume) identifies some of the reasons why biodiversity is undersupplied by biodiverse countries and calls for increased spending by the international community to protect biodiversity at globally optimal levels. With any level of spending, forming better policy to conserve biodiversity will require using information from natural scientists, to understand the impact of policy choices on natural systems, and from economists, to understand the differences in cost effectiveness of different policy choices. Theoretical analyses and some empirical analyses suggest that current policy is not cost-effective in conserving biodiversity: more biodiversity could be protected for the level of current spending. Only significant efforts to evaluate policies from a natural science *and* an

[8] Our analysis concentrated on in situ biodiversity protection, but ex situ conservation through gene banks and captive breeding programs also have a role to play in protecting biodiversity. Economics has not yet been brought to bear on contrasting ex situ versus in situ conservation, nor has it yet helped to inform ex situ programs in allocating their limited budgets.

economic efficiency perspective will insure that monies spent on biodiversity conservation are well spent.

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