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Forests and Biodiversity in Latin America:
San Jose Solution Paper

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Resources for the Future

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I. Introduction

The purpose of this report is to provide an overview of the benefits of forests and biodiversity in Latin America and to move toward the selection of cost-effective solutions for the protection of these resources. It draws from the literature to describe the situation with forests and biodiversity, both generally and within the context of Latin America. The various techniques for estimating the value of biodiversity are particularly subtle and are developed at some length. The value of the benefits of forests and biodiversity and the costs of protection are identified and drawn from the literature. Utilizing these data, a number of cost–benefit analyses are developed and some possible "solutions" are posited. It is noted that the literature related to the costs and benefits of forests and biodiversity is poorly developed for many values and much of the world. Despite these limitations, the solutions are assessed, their strengths and weaknesses noted, and a preferred solution is selected. Since some of the selections are built on contentious numbers from the literature and for cost systems that are not fully developed, the choice of solution considers questions of data reliability and completeness, as well as B/C ratio.

A rationale for the importance of Latin American biodiversity can be found in the Inter-American Development Bank's survey[1] that found "environment" one of the "challenges" facing Latin America. In the environment category, deforestation (81.5 percent) and loss of biodiversity (73.2 percent) were the top two concerns.

Forests and biodiversity often go hand-in-hand. Natural forests are the residence of much of the world's species and genetic biodiversity. The forest systems also provide a host of

[1] http://www.iadb.org/res.ConsultaSanJose

other useful outputs in the form of ecosystem services. These include erosion control, water management and purification, and wildlife and biodiversity habitats.

Biological diversity refers to all living things. Although biodiversity sometimes refers to the number of species in a geographical area, biodiversity also occurs at a number of levels of nature, including genetic variation among different individuals and populations of the same species. The range of ecosystems such as forests, agricultural areas, wetlands, mountains, lakes, and rivers and differences between and within geographical landscapes, regions, countries, and continents also are important dimensions of biodiversity.

Many of the benefits of forest ecosystems and biodiversity are viewed as global public goods. Public goods are those where the benefits are such that their consumption by one individual does not diminish the amount available for others. This aspect makes estimating the value of public goods difficult, since they are not transacted in markets. The public good aspect is one reason that decisions with respect to these outputs usually are determined using the political process. This paper discusses some methods for trying to assess the monetary value of the goods and services produced by biodiversity. These include various forms of contingent valuation. However, since estimates of value usually are for a single output, such as preventing the extinction of the spotted owl, the values usually do not represent the full array of biodiversity outputs.

Similar measurement problems hold for the goods and services produced by ecosystems. Often a host of ecosystem services are produced, many of which are public goods or for other reasons nonmarketed, such as flood and erosion control or water retention for dry periods, as with mountain snows. Again, many of these outputs might be viewed as public goods, while others involve difficulties in establishing viable markets for various reasons. Thus, although estimates of the values of these ecosystem services have been made, often they are quite crude.

Forests may be viewed as providers of ecosystem services. Traditionally, forests have been viewed as providing both private goods, such as timber and private recreation, as well as public services and externalities, such as water flow control and erosion mitigation that may be largely local. Thus, it has been argued that the decisions concerning these forests should be largely local. However, forest biodiversity often is viewed as a global public good with substantial positive externalities that generate global benefits, such as being a repository of global genetic information and of carbon, large volumes of which are captured in the cells of tree and soils of the forests and if released would contribute substantially to global warming. Therefore, forest biodiversity has been taken as of global concern.

II. Forests and Biodiversity

Forests

The U.N. Food and Agricultural Organization's (FAO) Global Forest Resources Assessment (2005) indicates that the globe's forest cover totaled 3.99 billion ha in 2000 or 29 percent of the land area (Table 1). Forests are distributed among tropical forests (47 percent); subtropical forests (9 percent); temperate forests (11 percent) and boreal forests (33 percent). Forests are abundant on all continents except Antarctica.

In Latin America, forests cover about 1064 million ha; a large percentage of this area is tropical. Recently the Global Forest Resources Assessment (FAO 2000) estimated the net loss of forest area at 9.4 million ha, with the tropics experiencing a decrease and the temperate regional experiencing a modest increase in forest area.

Table 1: Forests by Major Region: 2005 (million ha)

Region	Forest Area
Africa	655.6
Asia	566.6
Europe	988.1
North and Central America	707.5
Oceania	208.0
South America	852.8
World Total	3988.6

Source: Global Forest Resources Assessment 2005.

Biodiversity[2]

Biological diversity refers to all living things; however, often biodiversity refers to the number of species in a geographical area. One estimate is that the total number of species globally is about 14 million, while the approximate number of identified (described) species is less than 2 million. Note that only somewhat more than 10 percent of the estimated species have been described. The species described for plants are more than one-half of the estimated global totals, while those described for bacteria and fungi are only a small fraction of the estimated total species (Table 2).

Biodiversity Definitions

Biodiversity typically is considered at three levels: species diversity, genetic diversity, and ecosystem diversity. Species diversity is the variety and abundance of species in a geographical area. Species are the central unit in biodiversity studies and conservation, at least in part because ecosystems are hard to delimit and genes until recently have been difficult to count and identify (Wilson 1999). However, each species consists of subspecies (i.e., geographical races), populations, and individuals that possess their own varying levels of genetic distinctiveness. A population is a geographically distinct group of individuals of a particular species. A biological community is defined as the collection of species populations that exist and interact in a particular location. An evolutionarily significant unit (ESU) is a population or group of populations that is substantially reproductively isolated and is genetically unique from other populations, making it an important evolutionary component of the species. Since a major goal of biodiversity conservation is to maintain the evolutionary potential of unique lines of descent, practical species management and conservation efforts often target ESUs rather than entire species.[3]

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[2] This section draws heavily for its discussion of biodiversity from "Biodiversity in the United States" by Juha Siikamäki and Jeffrey Chow, forthcoming.

[3] For example, Pacific salmon have more than 50 distinct ESUs, which are basis for their management and conservation.

Table 2. Estimated Number of Species Globally

Kingdoms	Described Worldwide	Estimated Total Worldwide
Bacteria	4,000	1,000,000
Protoctista (algae, protozoa, etc.)	80,000	600,000
Animals	1,320,000	10,600,000
Fungi	72,000	1,500,000
Plants	270,000	320,000
Total	1,746,000	Ca. 14,000,000

Source: UNEP 1995, Global Biodiversity Assessment, Table 3.1–2, page 118.

Genetic diversity refers to genetic variation within species, both among distinct populations and among the individuals within a population. Genes are the chromosomal units that code for specific proteins that generate the unique morphological and biochemical characteristics of an organism and are passed down along generations of organisms. Variation arises from mutations in genes, and natural selection of these characteristics within a population is the primary mechanism of biological evolution. In sexually reproducing species, genetic diversity also comes from recombination that occurs when genes are exchanged. Genetic diversity within species populations helps to maintain reproductive vitality, disease resistance, and the ability of populations to adapt to changing environmental conditions. Biodiversity conservation methods sometimes include in situ efforts such as crossbreeding and translocation that help maintain the genetic diversity of wild populations. Germplasm repositories also have been established to store genetic diversity in the form of semen, embryos, and seeds ex situ.

Ecosystem diversity refers to the variation within and between communities and their associations with the physical environment. The richness of ecological systems within an area also is sometimes called systems diversity. Species play different functions within their communities; some species are functionally substitutable, whereas others (keystone species) play determinant roles in the food web and cannot be removed from the system without fundamentally affecting the species composition of the community. Ecosystem diversity also relates to landscape diversity, which denotes the diversity and connectivity of ecosystems within large geographical areas.

Biodiversity and Ecosystem Processes

Ecologists generally consider that species diversity increases ecosystem productivity, stability, and resiliency (McCann 2000). The relationship between diversity and stability has intrigued ecologists. In the early 1970s, rigorous mathematical analyses of community dynamics suggested that diversity may not necessarily support stability (May 1973). Instead, these analyses suggested that in artificially constructed communities with random interspecies interactions, species richness could destabilize community dynamics. During the last decade or so, experimental studies have helped to resolve the diversity–stability debate. Results from long-term field experiments, particularly those by David Tilman and his colleagues (e.g., Tilman 1996, 2004; Tilman and Downing 1994; Tilman et al. 2001) indicate that although species richness and the resulting internal competition may cause fluctuations in individual species populations, diversity tends to increase the productive stability of an ecosystem as a whole. It is now generally believed that differential responses by different species and functional groups, not simply the number of species, give rise to ecosystem stability (McCann 2000). Changes in the biomass production by some species, for example, are associated with dissimilar changes in the biomass production by other species. Consequently, aggregate variation in biomass production generally is more stable in systems with more species.

Diverse ecosystems generally have high rates of ecosystem processes, meaning that they produce more biomass than less diverse systems. However, increases in the rates of ecosystem processes seem to plateau at relatively low levels of species richness. Experimental analyses also have shown that what matters most is the diversity of functional groups, whereas species richness within functional groups may be less important (e.g., Knops et al. 1998; Holling et al. 1995).

Ecosystem resilience in ecology can be defined as the magnitude of disturbance that can be absorbed by the ecosystem before it changes to another equilibrium state. Robustness is the speed of return to equilibrium after a disturbance. Species, some of which may seem ecologically unimportant under current conditions, may play important roles to the resiliency and robustness of ecosystems to disturbances. For example, recent research suggests that diverse communities may have a greater capacity to resist invasions by exotic, nonnative species, which are major threat to biodiversity[4] (e.g., Kennedy et al. 2002; Tilman 2004).

Several components of species diversity determine its effects within actual ecosystems, including the number of species, the relative abundances of species, the particular species present, the interactions among the species present, and the spatial and temporal variations of these components. However, current knowledge about the consequences of biodiversity loss in actual ecosystems is limited, especially when considering large ecosystems and changes in biodiversity. Present information about how ecosystem functions relate to diversity comes mostly from simple ecosystems with only few species. In addition, most scientific evidence relates to only small variations in species composition and relative abundance. Critics point out that real ecosystems may be structured quite differently and operate under different processes than those in experimental studies (e.g., Grime 1997); hence, the role of biodiversity within natural ecosystems remains problematic.

Measuring Biodiversity

Biodiversity typically is characterized as the number of species, ecosystems, and genes. Ecological systems have three primary attributes—composition, structure, and function—that constitute biodiversity. Composition denotes the identity and variability of different elements such as species, genes, and ecosystems. Structure is the physical organization, pattern, and complexity of elements at different organizational scales (habitat, ecosystem,

[4] Exotic species that establish themselves in nonnative habitats may displace native species through competition for natural resources, predate upon native species to extinction, or alter habitat to the point that native species can no longer persist.

landscape). Function consists of ecological and evolutionary processes of elements, such as nutrient recycling, disturbance, or gene flow. Although each primary attribute of biodiversity potentially is important, interest in biodiversity concentrates around composition, especially species diversity (Franklin 1981; Noss 1990).

Two major approaches to the quantification of biodiversity have emerged. Economics literature has focused on measures of biodiversity that are based on joint dissimilarity among a set of species, whereas ecological literature emphasizes measures of biodiversity that are based on the relative abundance of species within ecological communities (Polasky et al. 2005).

Economics tradition of measuring biodiversity using the joint dissimilarity of species has its origins in Weitzman's (1992) work. A phylogenetic tree describes the evolutionary interrelationships among various organisms and their common ancestors. The phylogenetic tree can be used to determine joint dissimilarity of species from the branch lengths between different species on the tree. However, joint dissimilarity of species does not necessarily indicate the value of biodiversity. For example, Brock and Xepapadeas (2003) show that a slightly more diverse ecosystem can be much more valuable although the increase in dissimilarity is almost zero.

Most conservation efforts, however, deal with habitat rather than species. Habitat-based measures of biodiversity are needed for assessing and designing alternative conservation strategies. In the ecological literature, the most common characterizations of biodiversity are based on the relative abundance of species within ecological communities. Mathematical indices of biodiversity quantify species diversity at three different geographical scales. Alpha diversity is the number of species in a certain community and can be used to compare the diversity of different locales or ecosystem types. Gamma diversity is the species richness of a wide geographical area that encompasses multiple ecosystems, such as a country or continent. Beta diversity measures the variability of species composition over an environmental or geographical gradient and is sometimes calculated as the ratio of gamma diversity to alpha diversity.

Diversity indices also are based on relative abundance in order to provide information about the rarity or commonness of species in a community. The Simpson index (Simpson 1949) and the Shannon–Weaver index (Shannon and Weaver 1949) are the most common indices based on the relative abundance of species. Simpson's index represents the probability that two randomly chosen individuals in a community belong to different species.[5] The Shannon–Weaver index measures the order or disorder of species composition.[6] In a more ordered system, the abundances of different species are similar and biodiversity is lower; in a less ordered system, the opposite is true.

Practical measurements of biodiversity often are based on a collection of biodiversity indicators. Biodiversity indicators are measures of ecological endpoints that are selected based on their perceived importance to biodiversity. Examples of such endpoints include species richness and the number of extinct, endangered, and threatened species within an ecological community or geographical area. Repeated measurements of different endpoints help evaluate how biodiversity is changing over time and how this relates to human activities.

Different indicators must be specified for particular ecosystems to reflect their unique characteristics. In the United States, the National Report on Sustainable Forests (USDA Forest Service 2004) develops nine indicators for the conservation of biodiversity in forest ecosystems. Ecosystem diversity is addressed by five indicators, which measure the extent of forest fragmentation, different forest types, successional stages, and age classes in forests and protected areas. Other biodiversity indicators adopted by the U.S. Forest Service include the number of forest-dependent species, the percent of forest-dependent species at risk of not maintaining viable populations, the number of forest-dependent species that occupy a small portion of their original range, and the population levels of representative species from diverse habitats monitored across their range.

[5] Simpson's diversity index is calculated by taking the proportion of each species relative to the total number of species and then squaring and summing the proportions for all the species.
[6] The Shannon–Weaver diversity index is calculated from the proportion of each species relative to the total number of species, multiplied by the natural logarithm of this proportion. It is also known as the Shannon index and the Shannon–Weiner index.

However, even in the United States, several of these indicators currently cannot be monitored due to insufficient data.

Status of Biodiversity

Generally, biodiversity tends to be higher in southern areas and to decrease gradually toward the north. This pattern is especially true for flowering plants but it also emerges with the diversity of vertebrates. A similar longitudinal gradient is observed in global biodiversity (Gaston 2000). However, many basic questions related to the current status of biodiversity remain unanswered. For instance, the total number of species in the world is unknown. Estimates vary from a few million species to more than 100 million species, with the current consensus around 14 million species (Table 1). The species counts and their precision vary considerably across different taxonomic groups. Plant species are among the most completed inventories. It generally is accepted that approximately 300,000 to 500,000 plant species exist (Hammond 1995). For many groups of organisms, however, the precision of the estimated species counts is considered poor or moderate. The number of actually recognized and described species is fewer than 2 million.

The taxonomic group with the largest number of known species—about 1.3 million—is animals. Out of all known animal species, the vast majority, almost 1.1 million are insects and other arthropods. In addition, animal species include about 45,000 known chordates and about 70,000 thousand mollusks. Sponges, jellyfish, corals, hydras, and other aquatic animals comprise approximately 20,000 known species. The rest of known animal species mostly are worms of different kinds, such as flatworms, roundworms, or segmented worms (Hammond 1995).

Vertebrates and plants have been catalogued quite comprehensively, and their estimated numbers are not expected to change dramatically as more information is gathered over time. Viruses, bacteria, and fungi are the major groups of organisms with the largest estimated number of nondescribed species. New species are identified in all taxonomic groups; every year, more than 10,000 completely new species are identified. This rate of more than 300 new species per day has stayed somewhat constant (Purvis and Hector 2000; UNEP 1995).[7]

Extinctions

A species becomes extinct when the last existing member of that species dies. However, extinction designations often are regionally specific; that is, applying to a state, region, or country. Global extinction, of course, refers to the species disappearance from the entire globe. When only a few individuals of species exist, a species may become functionally extinct, meaning that the reproduction and the survival of that species is not possible. A species becomes extinct in the wild when the only living individuals belonging to that species are maintained in unnatural environments, such as zoos.

Although extinctions are difficult to observe and verify, the World Conservation Union (IUCN) is widely recognized as the world's leading conservation network.[8] The IUCN provides an estimate of threatened species based on its assessment of less than three percent of the world's 1.9 million described species (IUCN Red Book 2004). The IUCN list contains 784 species worldwide that are documented to have gone extinct in the wild since 1500. Over the past 20 years, 27 documented extensions have occurred.

Extinctions can occur naturally. A key question, therefore, is how the current extinction rate compares to the natural or background extinction rate. Background extinction rates are determined by examining fossil records. Using these data, geologists have estimated that around 0.1 to 1 species per million species per year have gone extinct globally. During the last 400,000 years, approximately 400 invertebrate and 300–350 vertebrate species are known to have gone extinct globally. The number of plant extinctions is not well known, but it is believed to be several hundred. Among birds, mammals, and amphibians, the taxa for which extinction records are most reliable, the current average

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[7] Regardless of what the exact number of current species may be, scientists believe that it is more than at any other point in the Earth's history. The current species represent only a fraction of all species that have ever existed, which is estimated at around 5 billion species.

[8] IUCN stands for the International Union for the Conservation of Nature and Natural Resources, the full name of the World Conservation Union. The IUCN involves 82 States, 111 government agencies, and more than 800 nongovernmental organizations, and some 10,000 scientists and experts from 181 countries.

extinction rates are about 50 to 500 times the background extinction rate. If possible extinctions are included; the current extinction rates are about 100 to 1,000 times the geological extinction rates. These estimates generally are considered conservative. Consequently, the recent extinction rates seem to be at least one or two orders of magnitude higher than the background extinction rates (IUCN Red List 2004).

Although extinctions have become more common due to human activities, considerably fewer species have gone extinct than was predicted in some widely publicized—and criticized (e.g., Simon and Wildawsky 1984)—scenarios about 20 years ago. At the time, it was predicted that as many as 15–20 percent of all species on Earth would go extinct within the next 20 years (Global Report to the President 2000). This would have meant a loss of tens of thousands or even hundreds of thousands of species every year. As noted, now that more than 20 years have passed since these predictions, the IUCN Red List reports that worldwide 27 species are known to have gone extinct during the last 20 years. So we have not witnessed the apocalyptic extinction rates that were once predicted. Nevertheless, even though extinction rates have remained relatively low, in many species groups 10–20 percent of all known species are endangered or threatened by extinction (IUCN 2004).

Species Endangerment

The IUCN characterizes endangered species as critically endangered, endangered, or vulnerable depending on the estimated risk of extinction. Critically endangered, endangered, or vulnerable species are determined by the IUCN to be under extremely high, very high, or high risk of extinction, respectively. All species in these three threat categories depend on conservation measures for their continued existence. The degree of species endangerment is determined by using multiple criteria, including population size, population range, and the rates at which they are being decreased.

Threats to Biodiversity

Major threats to biodiversity include habitat change, invasive alien species, pollution, and climate change (CBD 2007). Almost certainly the primary cause of contemporary biodiversity decline is habitat destruction and the degradation that results from the expansion of human populations and activities. Habitat loss takes several forms: habitat can be completely lost (e.g., urban development), can be degraded (e.g., forest management, pollution of wetlands), or can become fragmented (e.g., urban sprawl).

Species loss due to habitat loss relates to the species–area relationship, which is a fundamental concept in ecology. Typically, smaller areas have fewer species than the larger ones. Ecologists view distinct areas of nature as islands. These islands come in all sizes and differ in their connectivity with other islands. These islands are not necessary literal islands of land in an ocean or a lake; they refer to any relatively disconnected and distinct areas of habitat. So for instance, parks within a city are islands, as are lakes within a continuum of land. The basic idea of island biogeography (MacArthur and Wilson 1967) is that the number of species in area balances departures (extinctions) and arrivals of species. For example, the future number of species in a certain area will be determined by how many of its current species will persist and how many new species will immigrate there. Smaller islands tend to have fewer species than the bigger ones, and the less connected these islands are, the fewer species they generally contain. Habitat loss tends both to create smaller islands and to decrease their connectivity.

After habitat loss, nonnative species are the second leading cause of endangerment. Nonnative species, which also are called invasive or alien species, are broadly distributed; their numbers vary and follow roughly the patterns of population density and transportation routes. The introduction of nonnative species has important effects. Estimates of the economic losses due to nonnative species are tentative, but Pimentel et al. (2000) suggest that the losses from invasive species may be more than $100 billion annually, although this estimate generally is considered speculative (Polasky, Costello et al. 2005).

In addition to habitat loss and nonnative species, pollution, overexploitation of species, and illnesses are among the causes of endangerment of several species.

Some researchers believe that in addition to current threats of extinction, climate change may become one of the greatest drivers of biodiversity loss in the long run. Although nature has a notable capacity to adapt to changes, the relatively rapid climatic changes that have been predicted may leave species without adequate opportunity to adjust their ranges, especially if combined with increased fragmentation and decreased connectivity of habitats that create additional barriers to adjustment (Thomas et al. 2004; Millennium Ecosystem Assessment 2005.

III. Costs and Benefits of Biodiversity: Some Estimates

Biodiversity Losses

Extinctions are irreversible events that permanently remove a unique constituent of current biodiversity. How much of the evolutionary information passed on to the future is lost through extinctions? Since much of the evolutionary information (evolutionary history) is shared by other species, there is no one-to-one relationship between extinctions and the loss of evolutionary information. For example, if one in every ten species goes extinct, typically less than one tenth of the total evolutionary information in all species is lost. Even in extreme mass extinctions, in which nearly all species would disappear, the majority of evolutionary information would be maintained by the surviving species (Nee and May 1997). This does not suggest that extinctions are somehow insignificant events; the idea is simply that species share significant amount of evolutionary information and different species substitute as carriers of this information.

Alternative conservation strategies may be evaluated not only based on the number of species they protect but also how effectively they preserve evolutionary history (e.g., Mace et al. 2003; Brooks et al. 1992). For example, each species in the phylogenetic tree has some unique and some shared evolutionary history. Extinction of any one species leads to the loss of some unique evolutionary history, but not every species has the same amount of unique history.

Using preservation of evolutionary history for assessments of alternative conservation plans is intuitively appealing but difficult in practice because it requires currently lacking information on the evolutionary history of different species. Also, human ability to determine ex ante what evolutionary information is most valuable for preservation is problematic.

Values for Biodiversity

The values of biodiversity can be divided into three types: biodiversity as a global public good (i.e., biodiversity that provides global public benefits); biodiversity that provides national or regional benefits, but involves externalities; and biodiversity that provides private goods absence externalities. The typical approach of economists is to argue that the private goods need no special policies outside that of the provision of appropriate and enforced property rights. For local and regional public goods, appropriate policies, such as tax or subsidy policies usually are recommended. For global public good, some mechanism to provide global policies and perhaps global funding usually are deemed appropriate. Specific values from biodiversity stem from different beneficial uses, functions, and purposes of biodiversity, ecosystems, and their different components.

Biological commodities, such as food, feed, wood, and other fiber materials, that are largely traded in markets provide valuable good and services. Biodiversity in breeding stocks adds to the long-term sustainability of the production of these commodities. Additionally, medicinal and pharmaceutical products and sources for biological control and remediation found in biodiversity are valuable assets to society. While the final products are traded in markets, the biological resources that are inputs are usually viewed as public goods (Sedjo 1992a).

Environmental Services

Ecosystems, and especially forest ecosystems, provide a host of valuable environmental services. Many are largely nonmarket services, such as protection of water resources, nutrient storage and cycling, pollution breakdown and absorption, soil fertility and protection, climate stability, and recovery from disturbances of natural systems. In addition, forests and forest biodiversity generates benefits that are related to recreation and tourism, research and education, and culture and tradition. Many of these benefits are local or regional. Also, ecosystems provide a residence for unique plants and other organisms that have pharmaceutical potential.

Importantly, forests ecosystems also provide the ecological service of carbon sequestration. In the process of biological growth, carbon dioxide is taken into the plant and the carbon is captured into the cells of the plant and the oxygen released back into the atmosphere. Forest, unlike many plants, accumulate large amounts of biomass over long periods of time. Within this biomass, huge volumes of carbon are held captive in forest ecosystems, including in the trees, litter, and forest soils. Unlike many ecosystem services, which are local in their benefits, the climate benefits of forests are not restricted to one locale but are truly global in nature.

Use and Existence Values of Species: Contingent Valuation

The value of biodiversity, especially endangered and threatened species, often is not related to direct uses of those species but rather to the intrinsic worth of their existence. Such non-use values are difficult to estimate because they are not captured in market transactions. This has given rise to the development of a variety of nonmarket valuation methods that use surveys to elicit preferences for public goods, such as protection of threatened and endangered species. Because these methods are based on eliciting and examining stated rather than actual preferences, they also are broadly categorized as stated preference (SP) methods (e.g., Louviere et al. 2000). The contingent valuation (CV) method is the most commonly applied SP method for valuing biodiversity. The CV method requests people to state their approval or disapproval of specific policy programs for the protection of certain species or habitat. Because the proposed scenarios have specified costs that are varied across survey respondents, researchers can examine survey responses and estimate how much people are willing to pay for the protection of certain species. For a review of CV, see Carson and Hanemann (2005).

Not all economists agree that expressions of stated preferences are useful for economic valuation. For example, Diamond et al. argue that people simply do not have preferences for the types of goods CV studies deal with, and they note that the lack of experience in markets for environmental commodities make answering CV questions difficult. Hanemann (1994) is more optimistic about CV and argues that despite its challenges, the ability to place an economic value on environmental quality is essential for environmental policy and a cornerstone of the economic approach to the environment. Stated preference methods are widely applied in other fields of research, such as marketing and transportation research and decision analysis (McFadden 1974; Ben-Akiwa 1985; Louviere et al. 2000; Train 2003; Frances and Montgomery 2002). In these fields, SP methods regularly are used for eliciting consumer preferences for new products, product alternatives, and transportation options and to help their design and marketing.

Individual Species

Although wild species are no longer essential for human survival, their direct uses continue through hunting, fishing, and gathering. Methods for the valuation of such benefits are quite well established; they are explained and reviewed by Phaneuf and Smith (2005). Values from recreational uses of nature can be substantial. For example, salmon fishing has been estimated to be worth from around $14 to more than $110 per day per angler, depending on where and which type of salmon is fished. A day of hunting has been estimated to be worth between approximately $30 and $45 per hunter, depending on what game is targeted (Phaneuf and Smith 2005.)

Willingness to Pay (WTP) estimates suggest substantial variation in the WTP of the public for protection of rare, threatened, and endangered species. Bell et al. (2003) estimate that the WTP for the protection of Coho salmon can surpass $100 per year per household. Giraud et al. (2002) and Giraud and Valdic (2004) predict that U.S. households are willing to pay $69–99 for the protection of the Steller sea lion in Alaska, whereas the WTP of Wisconsin households for the protection of striped shiner (Boyle and Bishop 1987) and that of Massachusetts households for the protection of Atlantic salmon (Stevens et al. 1991) both are roughly $10. The finding that WTP varies between different species is not new. It is widely known that both the preferences of public and public officials vary across different species, often favoring charismatic megafauna over less magnificent yet equally or more vulnerable species.

Table 3 summarizes values for 29 species for which values have been estimated by one or more CV studies. These studies estimate WTP for avoiding loss of a species studies or for an increase in population size or viability.

Four caveats should be kept in mind when interpreting these WTP results. First, they are for the United States and so are unlikely to be representative for many Latin American countries. Second, since many animals are game animals, they are not representative for rare, threatened, or endangered species in general, where recreational values would be absence or quite different. Furthermore, most valuation efforts have focused on species of significant publicity, such as the bald eagle, spotted owl, pacific salmon, or whooping crane, and many less-publicized yet susceptible species have not been valued by any studies. Third, sampled populations vary remarkably across different studies. Fourth, survey and estimation methods advance quite rapidly and not every study uses methods that are now considered state-of–the-art. While not necessarily representative, however, the table suggests significant economic value is given to many very different individual species.

Natural Habitat

Several studies, again largely in the United States, have estimated benefits from different types of natural habitat. The results of these analyses are summarized in Table 5. Most of the benefits studied relate to water, and other ecosystem benefits are not considered. Also, most of these studies have estimated the value of specific resources in a specific geographic area. Thus, although many of the benefits are external to markets, they are largely confined to certain geographic areas. Where these areas are within one country,

Table 3. Estimates of WTP Values for Rare, Threatened, or Endangered Species in the United States ($2005)

	Low	High	Avg.	Reference(s)	Sample
Studies reporting annual WPT					
Striped shiner*			$8	Boyle and Bishop, 1987	Wisconsin
Atlantic salmon*	$10	$11	$10	Stevens et al., 1991	Massachusetts
Florida manatee*			$11	Solomon et al., 2004	Citrus County, Florida
Squawfish*			$11	Cummings et al., 1994	New Mexico
Red-cockaded woodpecker	$9	$20	$13	Reaves et al., 1994; Reaves et al., 1999	Varies (South Carolina and USA)
Bighorn sheep	$17	$40	$28	Brookshire et al., 1983; King et al., 1988	Varies (Wyoming hunters, Arizona households)
Riverside fairy shrimp*	$27	$31	$29	Stanley, 2005	Orange County, California
Gray whale	$23	$42	$32	Loomis and Larson, 1994	California
Bald eagle*	$21	$44	$32	Boyle and Bishop, 1987; Stevens et al., 1991	Varies (Wisconsins, New England)
Silvery minnow*	$31	$36	$34	Berrens et al., 1996; Berrens et al., 2000	New Mexico
Sea otter*			$39	Hageman, 1985	California
Gray whale	$31	$47	$39	Larson et al. 2004	California
Grizzly bear			$49	Brookshire et al., 1983	Wyoming hunters
Mexican spotted owl*	$49	$58	$53	Loomis and Eckstrand, 1997; Giraud et al., 1999	Varies (Arizona, Colorado, Utah, Northwest)
Whooping crane*	$43	$67	$55	Bowker and Stoll, 1987	Varies (Texas, USA)
Northern spotted owl*	$30	$128	$64	Rubin et al., 1991; Hagen et al., 1992	Varies (Washington, USA)
Pacific salmon and steelhead	$42	$118	$80	Olsen et al., 1991	Pacific Northwest (anglers and households)
Steller sea lion	$69	$99	$84	Giraud et al., 2002; Giraud and Valcic, 2004	USA
Coho salmon	$23	$137	$87	Bell et al., 2003	Oregon, Washington
Studies reporting lump sum WTP					
Sea turtle*			$17	Whitehead, 1991, 1992	North Carolina
Cutthroat trout*			$17	Duffield and Patterson, 1992	Visitors
Artic grayling			$23	Duffield and Patterson, 1992	Visitors
Peregrine falcon			$35	Kotchen and Reiling, 2000	Maine
Shortnose sturgeon*			$36	Kotchen and Reiling, 2000	Maine
Timber wolf	$54	$56	$55	Heberlein et al., 2005	Minnesota
Gray wolf	$5	$157	$67	Duffield 1991, 1992; Duffield et al. 1993; USDOI 1994; Chambers and Whitehead 2003	Varies (Minnesota, USA, visitors)
Monk seal*			$160	Samples and Hollyer, 1989	Hawaii
Humpback whale*			$231	Samples and Hollyer, 1989	Hawaii
Bald eagle	$239	$341	$290	Swanson, 1993	Washington

Note: * indicates that the study estimates WTP for avoiding the loss of species. Other studies typically estimate WTP for an increase in population size or chance of survival. Samples are typically households if not otherwise indicated.

Source:

Siikamaki, J. and J. Chow (forthcoming) "Biodiversity", in *Perspectives in Sustainable Resources in America*, R.Sedjo editor, as adapted, expanded, and updated from Loomis and White (1996).

public policy of that country should be able, in principle, to capture those benefits. And, indeed, the projects undertaken reflect that objective. For example, Holmes et al. (2004) estimate the value of riparian restoration along Little Tennessee River in western North Carolina; Kealy and Turner (1993) and Banzhaf et al. (2004) value the preservation of aquatic systems in the Adirondacks region; Smith and Desvousges (1986) estimate benefits from the preservation of water quality in the Monongahela river; and Hoehn and

Loomis (1993) estimate the value of wetland and habitat in the San Joaquin Valley in California.

As an alternative to scrutinizing finely defined natural habitats, some studies have estimated values of broadly specified resources. For example, in a widely known study, Mitchell and Carson (1984) estimate the benefits from preserving water quality in all rivers and lakes of the United States. In another geographically broad study, Larson and Siikamäki (2006) estimate the WTP of California households for regional and state programs designed to improve surface water quality and remove the impairments of water quality that limit beneficial uses of surface water bodies. Although breaking down the value estimates of broadly defined resources, such as all water bodies in an entire region or state, can be difficult, using a broadly defined resource as the basis of valuation may help to estimate a WTP that better reflects actual WTP for an aggregate resource than can be estimated by summing up WTP estimates for its sub-components.

Table 4. Estimates of the Value of Natural Habitat in the United States

Author(s)	Study	Mean WTP estimates (per household)
Holmes et al. (2004)	Riparian restoration along the Little Tennessee River in western NC	$0.69-40.89 per year
Loomis (1989)	Preservation of the Mono Lake, CA	$4-11
Silberman et al. (1992)	Protection of beach systems, NJ	$9.26-15.1
Kealy and Turner (1993)	Preservation of the aquatic system in the Adirondack region, US	$12-18
Smith and Desvousges (1986)	Preservation of water quality in the Monongahela River Basin	$21-58 (for users), $14-53 (for non users)
Walsh et al. (1984)	Protection of wilderness areas in CO	$32
Diamond et al. (1993)	Protection of wilderness areas in Colorado, Idaho, Montana, and Wyoming, US	$29-66
Boyle (1990)	Preservation of the Illinois Beach State Nature Reserve	$37-41
Loomis and Gonzales-Caban (1998)	Fire management plan to reduce burning of old growth forests in CA and OR	$56 per year
Larson and Siikamaki (2006)	Removal of surface water quality impairment in California	$67-133 per year
Loomis et al. (1994)	Fire management plan to reduce burning of old growth forests in OR	$90 per year
Hoehn and Loomis (1993)	Enhancing wetlands and habitat in San Joaquin valley in California, US	$96-284 (single program)
Richer (1995)	Desert protection in CA	$101
Mitchell and Carson (1984)	Preservation of water quality for all rivers and lakes	$242

Source: Siikamaki and Chow (forthcoming) as adapted from Nunes and van den Bergh (2001).

Adding up results from separate studies, each of which is focused on valuing a single species or localized habitat, can lead to ignoring relevant substitutes, improper rescaling, and unrealistically high aggregate value estimates. For example, Brown and Shogren (1998) note that aggregating the estimates summarized in Loomis and White (1996) implies that the total WTP for less than two percent of endangered species exceeds one percent of GDP, which they consider suspiciously high.

Ecosystem Services

A literature review of valuation of ecosystem services is provided by Nunes and van den Bergh (2001). Ecosystem services are the economically valuable functions that ecosystems provide to humans. More broadly, natural capital consists not only of specific biophysical natural resources, such as minerals, energy, animals and trees, but their interaction within ecosystems (Heal et al. 2005). These functions can generate both marketed and nonmarketed benefits. As discussed earlier, examples include water purification, oxygen creation, maintenance of soil productivity, waste decomposition, nutrient cycling, pest control, flood control, climatic control (climate moderation, carbon sequestration), pollination of crops and native vegetation, and provision of recreation opportunities. Valuation of ecosystem services can view the natural environment as a type of capital asset, natural capital, which generates returns in the form of ecosystem services. Biodiversity is considered an element of the natural capital and ecosystems are productive systems in which different elements serve different functions in the production of ecosystem services.

An example of ecosystem services is wetlands, which serve as flood barriers, soaking up excess water and slowing and preventing floodwaters from spreading uncontrollably. Wetlands help replenish groundwater and improve both groundwater and surface water quality, slowing down the flow of water, and absorbing and filtering out sediment and contaminants. Wetlands also provide spawning habitat for fish, supporting the regeneration of fisheries. Also, wetlands provide habitat for many species and support fishing, hunting, and recreation.

The National Academy of Sciences established a Committee on the Valuation of Ecosystem Services, which was given the task of evaluating methods for assessing ecosystem services and their associated economic value. The committee prepared a report, "Valuation of Ecosystem Services" (Heal et al. 2005), which highlights the central issues in valuing ecosystem services, especially those relating to aquatic ecosystems. The report notes that the key challenge in the valuation of ecosystem services is the successful integration of economic and ecology. It is required for "providing an explicit description and adequate assessment of the links between the structures and functions of natural systems, the benefits (i.e., goods and services), derived by humanity, and their subsequent values." The complexity of ecosystems makes the translation of ecosystem structure and function to ecosystem goods and services and their value especially difficult.

Table 5 summarizes a number of studies that have estimated values for ecosystem services that are related to water supplies, water quality, and soil-erosion control. These analyses have used a variety of methods, including CV (McClelland et al. 1992; Heberlein et al. 2005), averting expenditures (Laughland et al. 1996; Abdalla et al. 1992; Ribaudo 1989a, 1989b), replacement cost (Holmes 1988; Huszar 1989), and production function (Walker and Young 1986; Torell et al. 1990). For different techniques of valuing the environment as a factor of production, including averting cost, replacement costs, and production function methods, see McConnell and Bockstael (2005).

Table 5. Estimates of the Value of Selected Ecosystem Services in the United States

Author(s)	Study	Measurement method	Estimates
Walker and Young (1986)	Value soil erosion on (loss) agriculture revenue in the Palouse region	Production function	$4 and 6 per acre
McClelland et al. (1992)	Protection of groundwater program, US	Contigent valuation	$7-22
Torell et al. (1990)	Water in-storage on the high plains aquifer, US	Production function	$9.5-1.09 per acre-foot
Laughland et al. (1996)	Value of a water supply in Milesburg, PA	Averting expenditures	$14 and $36 per household
Heberlein et al. (2005)	Water quality in lakes of northern WI	Contigent valuation	$107-260 per household
Abdalla et al. (1992)	Groundwater ecosystem in Perkasie, PA	Averting expenditures	$61,313-131,334
Holmes (1988)	Value of the impact of water turbidity due to soil erosion on the water treatment, US	Replacement cost	$35-661 million per year
Huszar (1989)	Value of wind erosion costs to households in NM	Replacement cost	$454 million per year
Ribaudo (1989a,b)	Water quality benefits in ten regions in US	Averting expenditures	$4.4 billion per year

Source: Siikamaki and Chow (forthcoming) as adapted from Nunes and van den Bergh (2001).

An interesting example of ecosystem services valuation, not found in Table 5, has to do with the Catskills Mountains, from where New York City obtains its water supply. For years, the Catskills watershed has provided New York City with water that is usable without additional filtering. By the end of the 1980s, changing land use patterns, urbanization, and agricultural practices in the Catskills degraded the quality of groundwater and left New York City evaluating different alternatives for securing the quality of its drinking water. Constructing and operating a filtration plant was estimated to cost approximately $8–10 billion. Rather than making this expensive investment, New York City decided to invest in the preservation of the Catskills rural environment that had for so long provided the city with a high-quality water supply. The Catskills preservation program cost the city about $1.5 billion, generating considerable cost savings relative to constructing and operating a filtration plant. In this example, the cost savings from not having to construct and operate a filtration plant can be used as a value of ecosystem services of water filtration by the Catskills (Chichilnisky and Heal 1998). Using a replacement cost approach to the measurement of ecosystem services, however, requires the following three conditions to hold: 1) the replacement service is equivalent in quality and magnitude to the ecosystem service; 2) the replacement is the least-cost approach to replacing the service; and 3) people are willing to pay the replacement cost to obtain the services (Shabman and Batie 1987).

IV. Forests and Biodiversity in Latin America

Much of the work on biodiversity has been done either in a global context or, for studies of specific species or outputs, in North America. This section focuses on the very considerable biodiversity that is found largely in the tropical forests of Latin America. Latin America biodiversity is tied closely to its vast expanses of relatively undisturbed forests. However, this forest area is among the most threatened in the world. Brazil, for example, which has the world's largest area of tropical forests, is experiencing very rapid rates and absolute levels of deforestation (Kauppi et al. 2006). Similarly, fairly high rates of deforestation are being experienced in Venezuela, Argentina, and elsewhere in Latin America.

Forest Losses

The FAO Global Forest Resource Assessment 2005 estimates total world forested area at 3.95 million ha. It gives a global deforestation estimate of 13 million ha per year on average for 2000–2005. Although the FAO does not give deforestation figures at the national level for all countries, it does note that Brazil leads the countries with a net annual loss of -3,103,000 ha and Venezuela is tenth with net losses of -288,000 ha. South America has the largest continental net loss of forests, estimated at -7.3 million ha per year, but down from -8.9 million has per years in the period 1990–2000. Central America also experienced substantial losses.

An important ecosystem function of forests is the holding of large volumes of carbon in the tree cells, the dead wood and litter, and in the forest soils. South American forests hold roughly 90 billion tons of carbon directly in the forest biomass (about 32 percent) and another 70 billion tons (22 percent) in the dead wood, litter and forest soils (FAO 2006).

Latin America Biodiversity

Although Latin America constitutes only 16 percent of the land area of the planet, it is home to 27 percent of the world's mammal species, 42 percent of known reptile species, 43 percent of know bird species, 47 percent of known amphibians, and 34 percent of know flowering plants (IUCN 1996, 1997).

Another measure of biodiversity and biodiversity challenges is that of biodiversity "hot spots"; IUCN estimates of conservation "hot spots" are given in Figure 1. Latin America biodiversity "hot spots" run geographically from northern Mexico along the Pacific coast of Columbia to parts of Chile and include forested areas along the Atlantic coast of Brazil. Although none of these areas are in the Amazon, parts of the Amazon often also are considered hot spots. For example, high levels of unique biodiversity are found in the foothills of the Andes, which have a combination of wet tropical climatic conditions and large variations in elevation.

A second estimate of biodiversity hot spots is that of Conservation International. Conservation International identifies seven regions in Latin America as biodiversity hot spots (Figure 2). Again, these run geographically from northern Mexico to parts of Chile and include forested areas along most of the Pacific coast, from Mexico to Chile, and many of the forested regions along the Atlantic coastal regions of southern Brazil to parts of Argentina.

Threats to Biodiversity in Latin America

The 2004 IUCN Red List contains 15,589 species threatened with extinction. Of these, some 10,823 are found in South America and 3,946 in Brazil. Roughly 60 percent of these are species that reside in forests. Mesoamerica accounts for 4,117 of the threatened species, with about one-half forest species. Of these, Mexico accounts for 2,732 threatened species, more than 40 percent of which are forest species. While these lists may not be mutually exclusive, it is clear that Latin America (South America and Mesoamerica) account for a very large portion of the world's total threatened species (Table 6).

Table 6: Number of Species Threatened with Extinction in Latin America[9]

Habitat	South America	Brazil	Mesoamerica	Mexico	Caribbean	World
Total	10,823	3,946	4,117	2,732	2072	15,589
Forest	6,065	2,157	1,913	1,151		

Source: IUCN Redlist 2004 (accessed February 6, 2007).

[9] Note that there is a great deal of overlap in species across areas. While total species threatened globally is 15,589, the sum of the threatened species by major region is almost 50,000.

Figure 1. IUCN Biodiversity Hot Spots

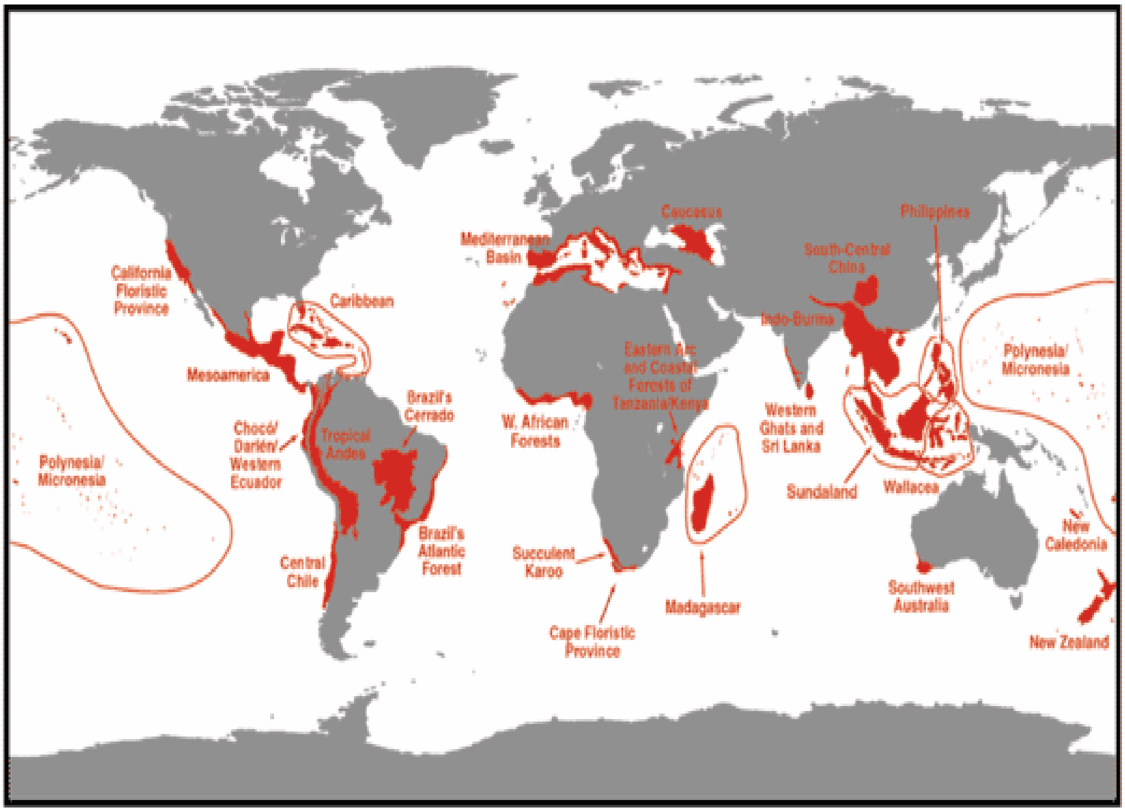
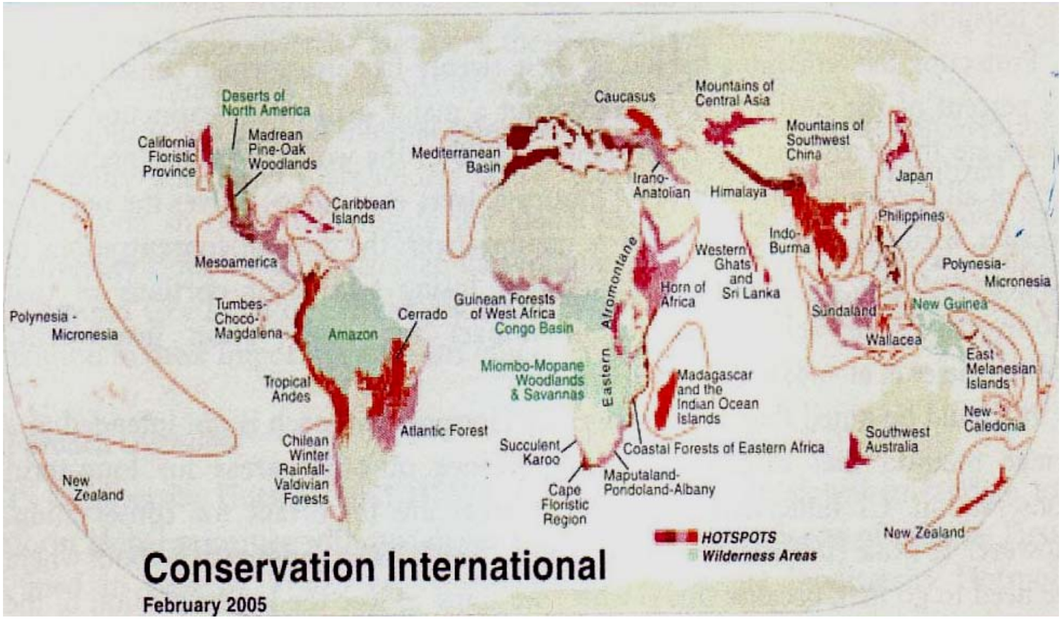


Figure 2.

Conservation International's Biodiversity Hotspots & High Biodiversity Wilderness Areas



V. Valuation of Benefits and Costs of Protection of Global Ecosystems and Biodiversity

As demonstrated above, although there are a very large number of valuations of biodiversity and ecosystem services, most are quite specific to a region and an output. Typically biodiversity has been addressed in a piecemeal and fragmented manner and there are a number of attempts to value various aspects of biodiversity and other nonmarket environmental services (Tables 4 and 5). However, it is difficult to generalize these results more broadly. In this section, we look at the few attempts to examine the benefits associated with forest ecosystems and the costs associated with protecting forests for ecosystem outputs, including biodiversity and carbon sequestration functions and services.

The conception and technical problems in estimating the benefits and costs of forest and biodiversity are quite different. In many cases, the benefits are nonmarket final consumption goods. The question of what these are worth to consumers is not revealed in markets but must be estimated by a CV approach such as a hypothetical WTP. This question becomes increasingly difficult to answer as the outputs being evaluated expand from the value of a spotted owl (species), to the value of a specific forest or habitat, to the value of the global forest system.

Costs, by contrast, typically are less difficult to quantify, as they usually involve the expenditure of resources that are valued in the market. Thus, land set-asides for ecosystem purposes have an opportunity cost that the market recognizes and is relatively easy to estimate.

Not surprisingly, then, analysts often avoid trying to quantify both benefits and costs and focus on the least-cost way to achieve a desired (perhaps politically determined) environmental goal, such as the protection of the spotted owl. This reality is reflected in this study, which finds many more estimates of the costs of preserving global ecosystems that estimates of the benefits. Nevertheless, some estimates of both global benefits and costs exist.

Global Ecosystem Services

There have been a limited number of broad ecosystem valuation attempts. Perhaps the grandest effort is that of Costanza et al. (1996) to evaluate the entire aggregate of global ecosystem services. They attempted to estimate the total value of global ecosystem services by deriving and summing up value estimates from the existing literature for a wide range of ecosystem services. The study, which estimates that the likely range of the total value of global ecosystem services as $16 to $54 trillion annually, was widely criticized, especially on the grounds that estimating the total value of global ecosystem services is not meaningful because the global ecosystem is a necessity without which life would not be sustained. This argument is summed up by Taylor (1998), who notes that the Costanza et al. estimate of the total value of global ecosystem services is "a serious underestimate of infinity." Although the Costanza et al. study has been influential and widely quoted, especially among scientists and environmentalists, it is considered by most economists fundamentally flawed both conceptually and methodologically. For a critique of the Costanza study see Toman (1998) or Bockstael et al. (2000).

Bioprospecting: Pharmaceutical Products

A major justification for preserving biodiversity has been its value as an input to medicinal and pharmaceutical products. Wilson (1992) argued that conserving species preserved an option value for the future since species might contain valuable compounds that would yield valuable pharmaceuticals in the future. The concept developed that there might be large economic returns from bioprospecting, that is, collecting various wild plants (and perhaps animals) and developing commercially valuable products from these or the constituents of these. Newman et al. (2003) analyze the origins of new drugs approved by the regulatory agencies over the period 1981–2002 and find that the majority of them have origins in nature.

Early studies estimated the value of conserving a species for pharmaceuticals use from $44 (Aylward et al. 1993) to $23.7 million (Principe 1989) per untested species. Besides the huge differences in estimates, the approach used—multiplying the probability of a successful product with its average revenue—was criticized in that it estimates the average value rather than the marginal value of an untested species. Because multiple species may contain the same compound and in this sense be redundant, marginal values are likely to be far less than average values. Simpson et al. (1996) developed a sequential search mode that takes redundancy into account. They demonstrate that when there is a large volume of biological material to search for valuable new products, the incremental value of one species likely is not very high. This is because when the total number of species is large and biological redundancy is common, different species may be close substitutes as sources of the useful biological constituent.

Nevertheless, in the 1990s bioprospecting was viewed as a vehicle by which developing countries could capture some of the rents that would accrue to their biodiversity. A famous project in the 1990s had Merck Pharmaceutical providing $1 million dollars to Costa Rico in return for 1,000 plants collected its forest. A Costa Rican organization was created to inventory and collect plants for possible use in the development of new pharmaceuticals (Sittenfeld and Gamez 1993). Although the Merck project has been successful in raising monies for Costa Rican biodiversity inventorying and research, few, if any, drugs have been developed from the plants collected and the model has not been transferred readily to other countries.

Table 7: Value of Biodiversity for Pharmaceuticals

Unit	Value of Benefits	Authors and Source
Biodiversity for pharmaceuticals	$23.7 million per untested species ($537 billion for 250,000 species)	Principe (1989)
Biodiversity for pharmaceuticals	$44.00 per untested species ($11 million)	Aylward et al. (1993)
Biodiversity for pharmaceuticals	As high as $9,431 for some species, or $20.63 per hectare, but much lower for other species and lands.	Simpson et al. (1996)
Biodiversity for pharmaceuticals	Land opportunity cost as bio-habitat as high as $9177 per hectare.	Rausser and Small (2000)

The reason for the lack of commercial success was suggested by the work of Simpson et al. (1996). They solved for the probability that generates that maximum species values using evidence to assign reasonable parameter values for revenues and costs. They found that the expected marginal value of a species may be as high as $9,431 in some locations. However, when estimating the economic returns from bioprospecting per hectare of land, Simpson et al. find that the returns are modest—from $0.20 per hectare in the California Floristic Province to $20.63 in Western Ecuador—even under assumptions likely to overstate the benefits and in regions that are known as global biodiversity hotspots and rich in endemic species. For this reason, Simpson et al. note that the potential for new product development does not provide a compelling economic argument for protecting biodiversity hot spots, and bioprospecting alone likely will not provide incentives for private landowners or companies to protect land for its pharmaceutical biodiversity values.[10]

This result was challenged. Using a similar conceptual approach as Simpson et al., Rausser and Small (2000) argued that if bioprospectors use prior information on the likelihood of product discovery, bioprospecting may in some cases result in considerable economic returns per hectare of land. They estimate that the value of richest biodiversity hotspots land in Western Ecuador may be as high as $9,177 per hectare (compared to the estimate of $20.63 per hectare by Simpson et al.).

Costello and Ward (2006) resolved the conflict by showing that the difference between the results of Simpson et al. and Rausser and Small relate mostly to different underlying assumptions regarding the relevant real-world parameter values, such as the number of relevant species, development costs, and to forth. They show that using similar parameters, the alternative approaches by Simpson et al. and Rausser and Small give

[10] Using the estimate of $122 per ha as the annual opportunity cost of the biodiversity hot spots in Latin America would cost about 7.93 billion.

similar results. For example, the Rauser and Small value of a hectare in Western Ecuador of $9,177 when conducting an optimal ordered search drops to $8,840 for a random search such as used by Simpson et al. Thus, the differences are not a result of the different search methods but reflect the choice of different parameters. Furthermore, upon investigation, Costello and Ward conclude that the underlying assumptions of Simpson et al. comport substantially more closely with the estimated empirical values than those of Rauser and Small. An implication is that the marginal value of an average species or hectare of land for bioprospecting usually is too small to cover the opportunity costs.

Regardless of the initial enthusiasm about benefit-sharing agreements and their use for biodiversity conservation, their success has been limited. One likely reason is that the economic returns from bioprospecting by themselves do not generate sufficient incentives to preserve biodiversity (Simpson et al. 1996; Polasky et al. 2005).[11] More relevant for this study is that although global, the effort focused on only one use of biodiversity: as an input for pharmaceutical products. The results suggested that the benefits rarely exceeded the opportunity costs of the lands for other uses.

Carbon Storage as an Ecosystem Service

A different approach to the evaluation of the benefits provided by the forest ecosystem has emerged from concerns about atmospheric carbon dioxide and climate change. The Intergovernmental Panel on Climate Change Third Assessment Report on Climate Change noted that forests contain huge amounts of carbon and that controlling deforestation and establishing new forests could have a significant effect on net carbon emissions (Kauppi, Sedjo et al. 2001).

It is possible to estimate the value of a forest for sequestrating carbon. Recent market transactions in the European Climate Exchange have placed the value of carbon for

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[11] Benefit-sharing agreement application has occurred in the United States in Yellowstone, the nation's oldest national park. Diversa Corporation, a biotech company, entered into a benefit-sharing contract in 1997 whereby Diversa compensates Yellowstone $175,000 for bioprospecting rights. In addition, the company pays to Yellowstone between 0.5– 10 percent in royalties for products developed from biological material obtained from the park (Siikamati and Chow forthcoming).

atmospheric carbon reduction within a wide range, from $10 to $100 per ton of carbon.[12] While volatile, at least a market price is available.

The total Latin America forest covers about 1 billion ha. The amount of carbon captured in a forest varies considerably, depending on the species, age, and density of the stand. Old primary, tropical forest may have 300 tons of carbon while young forests may have 100 tons.

An early effort to quantify the benefits and costs of using the forest ecosystem for carbon reduction was that of Pearce (1996). In quantifying the benefits, Pearce examine the relationship between land conversion in the tropics and the amount of carbon released. He related the carbon stored in the tropical forests to an estimate of the cost of carbon, which is an estimate of the damages associated with its net emission into the atmosphere, to obtain an estimate of the value of tropical forests in preventing a greater build-up of carbon in the atmosphere. Pearce used an average of about 100 tons of carbon per ha of forest and a price of $20 per ton of carbon. These figures provided estimated values for the services of ecosystems and forests as about $12 trillion and are presented in Table 8. This number is in the range estimated by Constanza.

Although these studies focus on the costs of the containment of carbon in the forest, the market approach for the cost of carbon provides some information on the benefits of containment. If a market for carbon credits is created, as in the European Union, the demand for carbon credits is determined by the severity of the constraints on emissions imposed by governments. One can infer from these constraints the value that governments place on containing carbon, which is an assessment of the benefits. Thus, the price per ton of carbon sequestered provides an estimate of the marginal damages believed to be forgone and an estimate of the market price of carbon times the quantity of carbon held captive as a result is an estimate of the benefits of this effort.

[12] The literature uses both carbon and carbon dioxide in its estimates of carbon storage and emissions. One ton of carbon dioxide is equal to 12/11 tons of carbon. The costs in this paper are for carbon and range between $10 and $100 per ton. This is equivalent to $2.73 and $27.30 per ton carbon dioxide.

Table 8: Estimates of the Value of Global Ecosystem and Biodiversity Benefits

Unit	Value of Benefits	Authors and Sources
Global Ecosystem Services	Globally $16 to $54 trillion annually. Prorated to $3.2 to $11 trillion annually for Latin America[13]	Constanza et al. (1997)
Carbon Capture Globally for 4 billion ha	$12 trillion net benefits	Extending Pearce's (1996) approach globally
Pharmaceutical Values	$0.20 to $20.63 per ha	Simpson et al. (1996)

Costs of Protecting Forests

As noted, a number of studies have looked at the costs of protecting very large areas or all tropical forests. These include Sedjo (1991, 1992); James et al. (2001); Brumer (2001), Greig-Gran (2006) and Kindermann (forthcoming) and are discussed below and their prorated results applied to Latin America.

VI. Latin America Benefit and Cost Estimates

Benefits from Latin America Forests and Biodiversity

What are the benefits of Latin American forests and biodiversity? As noted, the rationale for the importance of Latin American biodiversity can be found in existing assessments of the challenges facing Latin America. The question of the benefits that Latin America receives from forests and biodiversity, however, is complex. There often is confusion regarding biodiversity and ecosystem services and environmental damages.

[13] Latin America was assumed to be one-fifth of the world system roughly based on land area. It would be smaller based on GDP or population. Based on threatened species, it might be as large as one-third.

Environmental damages can be large without any direct threat to biodiversity,[14] while biodiversity can be threatened with minimal other environmental damages.

Table 9 provides estimates of the benefit values of forest ecosystems derived from the sources discussed above. In the absence of any Latin American-specific information, the value of forest ecosystems is simply a prorated value from the global estimates provided in Table 8. Note that there are only three estimates of the value of benefits provided in the two tables: those derived from Constanza et al. and Pearce and Simpson et al.

Table 9: Estimates of the Value of Latin American Biodiversity Benefits

Unit	Value of Benefits	Authors and Source
Latin American Ecosystem Services	Prorated from global estimate to \$3.2 to \$11 trillion annually for Latin America [15]	Constanza et al. (1997)
Carbon Captured in the Latin American Forest	\$3.4 trillion net benefits (carbon valued at \$20/ton.)	Derived from Pearce 1996
Latin American Ecosystem	Pharmaceutical value \$20.63 per ha in western Ecuador	Simpson et al. 1996

As noted, Constanza estimates the total value of all of the earth's ecosystem services. When prorated to Latin America, this gives an estimate of the value of ecosystem services as \$3.2–11.0 trillion annually. This study has been heavily criticized for various reasons. A different approach to estimating the value of forest/biodiversity systems in

[14] A recent article discussed environmental damages in "some of the world's most biodiverse rain forest." Although the article describes various environmental damages, it provides no discussion about threatened species or genetic resources nor makes any mention of threats to biodiversity as defined in this paper. It does, however, imply concerns over future ecosystem services. See http://www.csmonitor.com/2007/0507/p04s01-woam.html.
[15] Latin America was assumed to be one-fifth of the world system roughly based on roughly on land area. It would be smaller based on GDP or population. Based on threatened species, it might be as large as one-third.

Latin America is to consider its value for carbon sequestration services, as done by Pearce (1994). This approach is straightforward in that the estimate is made from components that, at least in concept, are easily measurable—that is the value of carbon sequestered in forest systems. We have estimates of the value of sequestered carbon as transacted in markets. However, the prices are quite variable.

Pearce examined the various regions separately and thus provides some information specific to Latin America. He concludes that "avoiding deforestation becomes a legitimate and potentially important means of reducing global warming rates" (1994, 31).

Using Pearce's average of 100 tons of carbon per ha for 1 billion ha of forest in Latin America provides an estimate that about 100 billion tons of carbon would be sequestered by the forest. This estimate is consistent with that of the FAO (2006) for Latin America forest carbon. Using a value of $20 per ton for sequestered carbon provides the estimate that the value of the ecosystem services provided by the Latin American is about $2 trillion. If the value of the 70 billion tons of carbon in the dead wood, litter, and soils noted by the FAO is included (FAO 2006), the additional value is $1.4 trillion for a total value of $3.4 trillion.

Costs of Protecting Latin America Forests and Biodiversity

This section draws from the earlier sections estimating global or near-global benefits and costs of forest ecosystems and adapting these to Latin America. The approach is to prorate the global estimates to Latin America using a one-fifth factor, since Latin America has roughly one-fifth the earth's forest area

An early study proposed a tradable systems approach for international forest protection (Sedjo 1991, 1992), with a major objective being to maintain biodiversity by preserving and protecting forest habitat. The study estimated the opportunity costs of closed-canopy forest land in other uses, that is the costs of preventing forest conversion, and used this as an estimate of the cost necessary to keep the land in natural forest. The total opportunity cost of the world's 2.655 billion ha of closed-canopy forest was estimated at about $26.4 billion annually.[16] For South America, the average annual opportunity costs used were those estimated by Browder (1988), which are $183/ha in 2005 dollars. At this land price, South and Central America's entire 670 million ha of closed forest was estimated to have a total opportunity cost of $12.3 billion annually (adjusted to 2005 dollars).[17]

It is sometimes suggested that most biodiversity could be preserved in a protected area of about 10 percent of the total ecosystem. In this case, the forest area to be preserved would be reduced to 67 million ha at an annual cost of around $1.23 billion. This estimate of land area necessary to preserve most biodiversity comports closely with Conservation International's estimate of Latin American biodiversity hot spots, where Conservation International[18] identifies seven regions and about 65 million hectares in Latin America.

In another recent global study of the costs of protecting global biodiversity, James et al. (2001) estimate that between ten and fifteen percent of global biodiversity could be protected for about $18 to $27.5 billion per year. If one-fifth of the global total is prorated to Latin America, the amount would be roughly $3.5 to 5.5 billion annually.

Pearce also estimates the cost of preventing the forests from being converted to other uses in a similar manner to the studies above. He asks what level of compensation would be necessary to bid the land away from alternative uses and keep it in its current forested state. Using Schneider's (1992) estimate of the value of the cleared land at about $300 per hectare, Pierce suggests that an average, one-time payment of $500 per ha could keep the land in forest. This provides a cost estimate of about $500 billion to provide for the permanent protection of the entire 1 billion ha of Latin America forests with their carbon sequestration services. Indeed, if Pearce had applied his approach to estimate the value of the entire global forest of 4 billion ha, Table 8 summaries this discussion of the costs and

[16] With the use of a 10% real discount rate.
[17] Browder's estimate, while dated, is similar to the much more recent estimates land opportunity cost estimates for Central America of $127 per ha and South America of $147 (cited in Chatham House Workshop, April 16-17, 2007 http://www.chathamhouse.org.uk/pdf/research/sdp/160407workshop.pdf
[18] http://www.biodiversityhotspots.org/xp/Hotspots/

presents four very broad estimates of the costs of maintaining forest ecosystems to provide biodiversity and carbon sequestration services.

In another paper, Brumer et al. (2001) argued that 70 percent of global biodiversity (about two percent of the earth's terrestrial surface) could be protected at an additional cost of $19 billion above current expenditures, or about $29 billion annually. Prorating one-fifth to Latin America provides an estimate of about $5.8 billion annually to protect 70 percent of Latin America's biodiversity. For comparison, Simpson (2004) estimates current worldwide expenditures on biodiversity conservation at about $10 billion annually.

A recent study by Grieg-Gran (2006) focuses on avoiding deforestation. The approach uses financial incentives to compensate owners for lost market value. The approach focused on eight countries that accounted for 6.2 million ha or 46 percent of global net deforestation from 2000–2005. The Latin America countries included are Brazil and Bolivia. The goal of this study was not to protect the entire forest or even a fixed percentage of it. Rather, the study was designed to estimate the cost of preventing deforestation by compensating owners for keeping the land, which would have been deforested, in forests. The target was to offset the expected 6.2 million ha of deforestation each year, which has been the rate of deforestation in recent years. The causes of deforestation varied by region, with most deforestation in Brazil and Bolivia generated by conversion to pasturelands, while in Indonesia much of the conversion is driven by land conversion to palm oil production. The study estimates the compensation costs at roughly $5 billion to permanently secure the 6.2 million ha in forest. Unlike most other studies, the study explicitly included administrative and monitoring costs, which are estimated at $4–$15/ha per year and, thus, an additional $25–$93 million. However, this payment would persist indefinitely and increase as additional lands were protected from deforestation.

In concept, the program would add an additional 6.2 million ha in protected forest in year two and subsequent years. If the program continued for 10 years, the administrative and monitoring costs would be running at between \$250 million and almost \$1 billion per year. Also, since the concept involves monitoring and protection indefinitely, those costs of \$250 million to \$1 billion would continue indefinitely even if the basic program of adding new areas of forest were discontinued. However, the study notes that the actual costs are likely to be higher than programmed because of leakage[19] and administrative expenses.

Kindermann et al. (forthcoming) use three economic models of global land use and management to analyze the potential contribution and cost of carbon credits to provide incentives for avoided deforestation activities reduce greenhouse gas emissions. These models estimate the costs associated with a reduction of deforestation below the trend for 2030. The study finds that an average of about \$3 per ton CO2, or about \$0.4– \$1.2 billion per year in 2030, would generate a 10 percent reduction in deforestation rates. The Present Value (PV) of the total costs using a 10 percent discount rate is roughly \$4–\$12 billion for the 10 percent reduction case. They estimate that a \$20 per ton CO2 credit would reduce CO2 deforestation emissions by 50 percent, or 1.5–3.4 Gt CO2/year by 2030. This program would require \$1.72 to \$2.80 billion per year through 2030 or a PV at 10 percent of \$17.2 to \$28.0 billion for the 50 percent reduction case. However, the information cannot be used for the *Solutions* paper since the study does not provide sufficient information to calculate the full costs of the project nor does it report on deforestation avoided in the transitional 30 years.

Finally, Simpson et al., while not estimating costs specifically, find biodiversity values that are considerably below the opportunity costs of the land as found in numerous other studies (e.g., Browder 1988; Schneider 1992)

[19] Leakage refers to the shifting of deforestation within the country; that is, the situation where preventing deforestation in one location only deflects the deforestation to another forest.

These models are applied to the various tropical regions separately, including Latin America. South and Central America have marginal costs similar to those for the rest of the globe, although the volume of carbon captured by avoided deforestation is a relatively large proportion: about three-eights of the total, depending upon the price.

Table 10: Estimates of the Costs of Protection for Latin America's Biodiversity

Estimate (year)	Percent Protected	Cost
Sedjo (1992) Chathamhouse (2007)	10 percent of closed forest (67 million ha) area, e.g., hot spots	$1.23 billion annually (rental)
James (2001)*	10–15 percent forest land	$3.5-5.5 billion annually (rental)
Brumer (2001)*	2 percent terrestrial area.	$5.8 billion annually
Pearce (1996)	100 percent forest are Latin America	$500 billion one time, or $25 billion annually discounted at 5%.
Grieg-Gran (2006)	6.2 million ha/yr (1.5%) each year for indefinite period or an accumulation of about 1.5% of the global forest annually	$5 billion purchase payment and $25–$100 million each year for administration
Kindermann et al. (unpublished)	Reduce rate of deforestation 10% to 50%	Cost is PV $4.0– $12.0 billion for the period through 2030; for a 10% reduction case PV of costs $17.2–$28.0 billion for the period through 2030 for the 50% reduction case
Simpson et al. (1996)	100%	Costs equal opportunity costs of the land at roughly $150–200/ha

* Prorated from global estimates assuming Latin America has one-fifth of the area and costs.

Note that the first three studies focus on the costs of establishing permanent forest reserves adequate to protect core biodiversity. The Grieg-Gran (2006) study, by contrast, estimates the cost of halting all deforestation in high-incidence countries with targeted (purchased) compensation each year for several years and with continuing monitoring. This study assumes that the remainder of the forest will provide its values without compensation. The Kindermann et al. study focuses on the costs of undertaking substantial investments gradually to reduce the rate of deforestation compared to the trend. The Pearce study focuses on the value of sequestered carbon and asks how much compensation would be require to all forest owners for not releasing the carbon through deforestation. Simpson et al. do not estimate explicitly the costs of protection. Rather they implicitly use the market land price as an estimate of the cost. Table 10 summarizes these cost estimates for Latin American biodiversity.

VII. Proposed Solutions: Benefits and Costs

Benefits and Costs

Four benefit-cost calculations are reported in Table 11. The first two use Constanza's high-benefit estimate with the low- and high-cost estimates from the literature. Although Constanza's estimates are criticized, they are prominent in the literature. Also, they do provide the high boundary for benefits. The third benefit-cost calculation is that implicated in Pearce's work, where the benefits are all derived from sequestered carbon conservatively valued. Finally, the fourth benefit-cost calculation uses Pearce's benefits approach, calculating the carbon-sequestration values of the forest protected in the Grieg-Gran study and matching it with the cost estimates of the Grieg-Gran study.

The solutions using the Constanza benefits tend to generate massive benefit-cost ratios, especially since these are placed against fairly modest cost estimates. The Pearce study has the advantage that both benefits and costs were addressed by the same researcher in a consistent manner. The Pearce estimate of values is large but so are his costs. The benefit – cost ratio of calculation four appears reasonable, suggesting that forests provide a useful and relatively low-cost mechanism for addressing the damages associated with carbon emissions.

The final estimate focuses on the carbon benefits of preventing deforestation—a timely issue. The emissions avoided by avoiding deforestation are substantial, as are the associated damages that are avoided. The benefit - cost ratio of 2.4 is favorable.

However, the research acknowledges that the costs of controlling leakage and monitoring are substantial and continue indefinitely.

Table 11. Benefit-Cost Ratios for Saving Latin America Forest/Biodiversity

Constanza's benefits (all ecosystem services) and Sedjo's (forest land costs)	Benefit - cost = 32,000–110,000/1.2 = 2,666 to 9,166 (Sedjo/ Chathamhouse's low-cost estimate)
Constanza's benefits (all ecosystem services) and Brumer's (forest costs)	Benefit - cost = 32,000–110,000/5.8 = 572 to 1,896. (Brumer's high-cost estimate)
Pearce benefits (carbon storage) and costs (forest land values)	benefit – cost = $2 trillion/$0.5 trillion = 4.0 (Pearce's cost and benefit estimates)
Carbon storage benefits derived from Pearce and Grieg-Gran avoided deforestation costs	benefit – cost = $12.4 billion/$5.2 billion = 2.4 (carbon benefits and Grieg-Gran costs)
Benefits per hectare (biodiversity for drugs) Simpson et al.	benefit – cost per ha = $20.63/$150 = 0.134 Costs per ha from selected land value estimates (Browder, Schneider)

The Results

These results indicate a range of benefit - cost ratios from 0.134 to 9,166 using cost and benefit estimates from the literature. Benefit estimates are substantially more difficult to undertake and were done by only three researchers: Constanza, Pearce, and Simpson et al. Few find Constanza's estimates creditable, both due to the nature of the methodology and the advocacy nature of the estimates. By contrast, Pearce's methodology is clear and sensible. Respectable estimates of benefits are obtained by looking at carbon capture—a service where market prices exist and where estimates of physical values are fairly straightforward. Although he only estimated the value for one output, carbon storage services, this value appears to be large and important. Simpson et al. also provided useful estimates, also for only for one output —biodiversity for pharmaceuticals use. This value, while potentially large in the aggregate, is small compared to its cost, which is the opportunity cost of the land.

On the cost side Grieg-Gran does the most comprehensive job of assessing costs. Details are included that examine the alternative uses of the forestland and develop opportunity costs accordingly. However, because the approach attempts to differentiate between lands that will be compensated and those that will not be, substantial monitoring costs would be necessary and leakage is likely to be great and difficult to control.

A weakness of these aggregate results is that in all of these cases but one the costs and benefits were estimated independently by different groups of researchers. By far the most creditable estimates are those of Pearce since one researcher estimated both the costs and benefits in a consistent and creditable manner. However, combining Pearce's benefits approach with the cost estimates of Grieg-Gran provides both a sensible combination of literature approaches and sensible results.

Proposed Solutions: Overview of Approach

The approach of this study is to examine the broad literature and several estimates of benefits and costs from these various sources and determine which of the studies have a scope and information that can be useful in the solutions analysis.

The benefit side typically is more difficult to measure and the estimates more suspect. This is because benefits typically are not transacted in markets and, hence, difficult to quantify. Approaches often use various CV techniques, typically survey approaches such as WTP. These approaches and their application with specific examples are discussed in some detail in this study. The benefit estimates of this approach remain somewhat controversial. Where the benefit outputs can be transacted in markets, however, as in the carbon-sequestration services of forests, the benefit estimates may be less controversial.

It should be noted that much of the existing literature deals with estimating the value of specific species, such as Alaskan salmon or the northern spotted owl, found in specific ecosystems or at specific locations. Typically, these studies are done on small and unique areas. For species, the focus usually is on a single activity, such as hunting or bird watching. For ecosystem outputs, the focus also usually is on one output, such as an aspect of water for a limited region. Upon investigation, these studies have been determined to be of limited usefulness for the current project of identifying solutions because of their limited focus.

After reviewing the literature, this investigation moves to the identification of the few larger studies that take a continental or global perspective. First, regarding the value of biodiversity, only two studies were found that estimate the benefits value specifically of species biodiversity at the continental or global level (Simpson et al. 1996 and Rausser and Small 2000). These studies, which look at bioprospecting and estimate the value of the benefits of biodiversity as an input into pharmaceutical product production, take a global perspective. However, they are limited to examining only one output, that of the value of biodiversity in the production of pharmaceuticals. Other values are not addressed. The Simpson et al. study is used for some of the cost–benefit ratios.

Two other studies, Constanza et al. 1997 and Pearce 1994, estimate the values of the benefits of ecosystems and are used in our analysis. Constanza develops an estimate of the aggregate value of global ecosystem services. Pearce derives an estimate of the value of the services of the forest in sequestering carbon. Although Pearce's study does not include all the outputs in its evaluation of benefits, it does demonstrate that the value of ecosystem services is quite large. In fact, Pearce's paper argues that the other values of ecosystem services are fairly modest. In the solutions, we adapt his carbon-sequestration estimate, although admittedly conservative due to the absence of other outputs, as a component of our solutions analysis. These two estimates, those of Constanza and Pearce, are utilized in our solutions-assessment process.

The cost-side estimates generally are less difficult to obtain and are viewed as more reliable. The approach often involves estimates of the opportunity costs of maintaining the land in habitat for environmental and ecosystem uses rather than converting the land to development or agricultural uses. Markets tend to provide information on opportunity costs of the land in the form of land market prices or land rents.

The costs are examined in seven cited studies. Three studies (Sedjo, James, and Brumer) examine the costs associated with preserving existing ecosystems and forests globally and provide information at the global or continental level. These studies are oriented to estimating the costs of protecting the ecosystem, and they provide estimates of the portion of the global ecosystem that needs to be protected, such as 10 percent of the forests. Pearce's study is the only one that provides estimates of benefits and costs. His cost estimates are derived in a manner similar to Sedjo in that he estimates the opportunity costs of the land by reference to average markets prices. However, since his focus is on sequestration and not on simply protecting a representative sample of species biodiversity, he incurred the costs of protecting the entire forest, not just some fraction of it. Additionally, the studies of Grieg-Gran and Kindermann et al. provide estimates of the costs of preventing deforestation in large areas of the world that currently are experiencing large-scale deforestation. However, Kindermann's estimates are incomplete and not used in the *Solutions* analysis. In these, as with the aforementioned studies, costs are derived using land prices as a measure of opportunity costs. Although none of these studies provide estimates of the values of the benefits, their estimates of the costs are viewed as sensible, and these estimates are used in our solutions-assessment process. Finally, the Simpson et al. study provides estimates of the values of biodiversity for drug production as well as the basis for estimating the costs.

Some Solutions

Four possible solutions are developed using the aforementioned set of three benefit estimates and six cost estimates. The solutions use three of the benefit estimates and relate these to an appropriate cost estimate. Four possible solutions are proposed using several estimates of benefits and costs chosen selectively from these various sources. Benefit - cost ratios for each solution are developed. A discussion of each is presented and a preferred solution is chosen based on the benefit – cost ratio and other considerations.

Solution One: Protecting Biodiversity for Its Value in Drugs

For solution one, the value of the land as a repository for biodiversity was estimated by Simpson et al. The costs of protection are the opportunity costs of the land for other uses. The stated benefit - cost ratio of 0.134 suggests that even the lands most rich in biodiversity are unlikely to justify the repository status solely on the basis of the probability that the biodiversity may someday be useful for drugs and medicines.

Table 12: Solution One: Benefit -Cost Ratio for Saving Latin America Forest/Biodiversity for the Biodiversity Values for Drugs

Benefits per ha (biodiversity for drugs) Simpson et al.	Benefit – cost ratio per ha = \$20.63/\$150 = **0.134** Costs per ha from selected land value estimates (Browder, Schneider)

Solution Two: Protecting Forests to Prevent Carbon Emissions

Solution two combines Pearce's estimates of benefits with the cost approach of Grieg-Gran for restraining deforestation in regions with high recent rates of deforestation. This approach has the advantage of using the strongest estimates of benefits and costs available in the literature for large-scale projects of this type. The benefits side uses the estimated values of the ecosystem for keeping forest carbon from being released into the atmosphere─estimates that are based on the actual price experience in the European Union carbon market. The cost approach is comprehensive with the objective to avoid deforestation. The costs, as with most of the other studies, are based upon compensation based on the opportunity costs of t keeping the land in forest. This cost approach also included estimates of administration and monitoring costs, something that is absent from all of the other cost estimates. A unique feature of this approach is that compensation is not provided for all forestlands but only for those determined to be likely candidates for deforestation. This reduces overall costs but is susceptible to leakage problems, which the author acknowledges. The Benefit – Cost ratio of this solution is about 2.4.

For this estimate, the benefits are likely to be somewhat higher than listed because only carbon-sequestration benefits were assessed; however, the carbon benefits may be somewhat lower because complete success in avoiding all deforestation is unlikely. However, the costs also are likely to be some higher because some additional compensation may be required to prevent leakage. In addition, the likely administrative and monitoring costs have only crudely entered the benefit – cost calculation.

Table 13: Solution Two: Benefit - Cost Ratio for Saving Latin America Forest/Biodiversity Through Payments for Avoided Deforestation

Carbon benefits derived from Pearce and Grieg-Gran costs	Benefit- Cost ratio = $12.4 billion/$5.2 billion = **2.4** (carbon benefits and Grieg-Gran costs)

Solution Three: Protecting Ecosystems for Ecosystem Services

Solution three justifies protecting forest and biodiversity on the basis of ecosystem services provided. The benefits are drawn from those estimated by Constanza for the globe. Although these have been highly criticized, they are the only ecosystem benefit estimates available for very large, continental-size regions and are estimated at $11 trillion for Latin America. Note that this estimate has been criticized both for being too high and for being to low (Toman 1999). However, the value of the ecosystem for carbon storage is $3.4 trillion, so even without other values, Constanza's estimate is within an order of magnitude. Furthermore, if carbon prices escalate as expected, thereby raising Pearce's estimate, and other ecosystem service values are large, which seems reasonable, Constanza's estimate becomes even more within the range of being comparable. It may be that Constanza is right for the wrong reason.

One reason why the costs tend to be low in the avoided deforestation at the marginal solutions is that there are some lands in which the forests would be the highest value use and thus no payments would be necessary. Thus the average opportunity cost per hectare would be drawn down by lands submarginal for nonforest use.

Table 14: Solution Three: Benefit- Cost Ratio for Ecosystem Services in Latin America

Constanza's benefits and Sedjo's costs	Benefit- Cost ratio = 32,000–110,000/1.2 = 2,666 to 9,166 (Sedjo/ Chathamhouse's low-cost estimate)
Constanza's benefits and Brumer's costs	Benefit- Cost ratio = 32,000–110,000/5.8 = 572 to 1,896 (Brumer's high-cost estimate)

Solution Four: Protecting Forests for Carbon Values

Solution four is a different approach to estimating the value of forest/biodiversity systems in Latin America: considering its value for carbon-sequestration services, as was done by Pearce (1994). This approach is the most straightforward in that the estimate is made from components that, at least in concept, are easily measurable and include the carbon sequestered in forest systems and the value of sequestered carbon as transacted in markets.

With the advent of concern about global warming and the role of carbon and carbon dioxide in that warming, it is possible to estimate the value of a forest for sequestrating carbon. Recent market transactions in the European Climate Exchange have placed the value of carbon for atmospheric carbon reduction within a wide range: from $10 to $100 per ton of carbon.[20] While volatile, at least a market price is available. The total Latin America forest covers about 1 billion ha. The amount of carbon captured in a forest varies considerably, depending on the species, age, and density of the stand. Old, primary tropical forest may have 300 tons of carbon, while younger forests may have 100 tons. Using Pearce's average of 100 tons of carbon per ha for 1 billion ha of forest in Latin America provide an estimate that about 100 billion tons of carbon would be sequestered by the forest. This estimate is consistent with that of the FAO (2006) for Latin America forest carbon. Using a value of $20 per ton for sequester carbon provides the estimate that the value of the ecosystem services provided by the Latin American forest is about $2

[20] This is equivalent to $2.73 and $27.30 per ton carbon dioxide

trillion. If the value of the 70 billion tons of carbon in the dead wood, litter, and soils of the forest noted by the FAO (FAO 2006) are included, the additional value is $1.4 trillion, for a total value of $3.4 trillion. Estimated global values for the services of ecosystems and forests are presented in Table 7.

Extending Pearce's approach globally would result in global benefits valued at about $12 billion. This number is not vastly different than that of Constanza. Pearce also estimates the costs of preventing the forests from being converted to other uses in a similar manner to the above studies. He asks what level of compensation would be necessary to bid the land away from alternative uses and keep it in its current forested state. Using Schneider's (1992) estimate of the value of the cleared land at about $300 per ha, Pierce suggests that an average, one-time payment of $500 per ha could keep the land in forest. This provides a cost estimate of about $500 billion to provide for the permanent protection of the entire 1 billion ha of Latin America forests with their carbon sequestration services. Indeed, if Pearce had applied his approach to estimate the value of the entire global forest of 4 billion ha, Table 8 summaries this discussion of the costs and presents four very broad estimates of the costs of maintaining forest ecosystems to provide biodiversity and carbon-sequestration services.

Table 15: Solution Four: Benefit- Cost Ratio for Saving Latin America Forest/Biodiversity Through Payments for Carbon Sequestration

Pearce benefits and costs	Benefit-Cost ratio = $2 trillion/$0.5 trillion = 4.0 (Pearce's cost and benefit estimates)

Choice of Solution

Solution one, which addresses the protection of biodiversity for the purpose of maintaining biodiversity values for drugs and medicines, does not give a favorable Benefit- Cost ratio (only 0.134). A problem with this solution is that the benefits capture only the value of biodiversity as inputs to drugs and ignores other values.

Solution three has the highest apparent Benefit-Cost ratio at the mid-hundreds to close to 10,000. However, as noted, the estimate of benefits has been highly criticized. Also, while all of the cost estimates cover forests, none of them appear to cover completely the full costs of maintaining the entire range of ecosystems. Instead, they are focused only on forest systems.

Solutions two and four seen the most reasonable and provide sensible and acceptable Benefit- Cost ratios. Solution two has a Benefit- Cost ratio of 2.4, while solution four has a Benefit- Cost ratio of 4.0. These two solutions focus on the benefits from carbon storage associated with avoiding deforestation. Solution two focuses on avoiding deforestation at the margin, while solution four examines avoiding deforestation in the entire system. Solution two is more realistic in that it includes monitoring costs and recognizes that some benefits will be provided without costs to the project.

I maintain that solution two is the desired solution. That project has the advantage of working on the margin, where deforestation is high. Forests that are not about to be harvested need not receive compensation. Although in concept this is desirable, leakage will be a problem. However, the benefits are sufficiently large that some slippage can occur and the project still will be economically viable by the cost–benefit calculus.

In the context of Latin America, areas of high deforestation would be identifiable given the high rates of deforestation in several Latin American countries and the compensation package would be applied there The above discussion provides a range of general of average cost estimates that might be needed to protect Latin America biodiversity. The costs are generic and are unrelated to any particular approach to protection. Rather, the costs reflect the market opportunity costs of the habitat to be protected and do not account for the costs of administering such a program. Of course, if the biodiversity values were private goods transacted in markets, a nonmarket administered program would not be necessary. However, it is the externality aspect of biodiversity that makes an essentially nonmarket–oriented approach necessary.

VIII. Conclusions

The empirical evidence on the costs and benefits of protecting forest ecosystems and biodiversity is limited, particularly for large global or continental systems. Most of the considerable research on biodiversity benefits has focused on the value of individual species for a specific purpose or a confined ecosystem, often for only one of its multiple outputs.

There is more evidence for the costs of protecting species and ecosystems than for the benefits. However, the data on the costs of protecting individual species is very specific and narrow in focus. The information on protection of individual ecosystems is slightly more robust. However, most studies apply to the United States and deal with only one aspect of ecosystem services. Even in the United States, there are a few studies that look at forest ecosystems or biodiversity broadly and most of those focus on a signal aspect, such as water or recreation.

Globally, there are a very few studies that look at the benefits of global or large regional ecosystems or that look at the costs. Only two studies, Pearce and Simpson et al., have looked at both benefits and costs. Additionally, although no studies have been directed specifically at Latin America, several of the studies have looked at this issue from a global perspective and have some regionally specific information or estimates.

This paper estimated forest or ecosystem service benefits using information from Constanza et al. (1997), Pearce (1994), and Simpson et al. (1996). The range of costs used was found in six studies: Sedjo (1992), James (2001), Brumer (2001), Pearce (1994), Kindermann (forthcoming), and Grieg-Gram (2006). A preferred "solution" is suggested based on an estimate of the value of avoided deforestation for carbon sequestration derived from Pearce and a program to retard deforestation through compensation for retaining forests. A cost component of this approach involved that of monitoring the condition of the forest.

References

Aylward, B.A. 1993. A case study of pharmaceutical prospecting. In *The Economic Value of Species Information and Its Role in Biodiversity Conservation: Case Studies of Costa Rica's National Biodiversity Institute and Pharmaceutical Prospecting*, edited by B.A. Alyward, J. Echeverria, L. Fendt, and E.B. Barbier. London: Environmental Economics Centre,

Abdalla, C.A., B.A. Roach, and D.J. Epp. 1992. Valuing environmental groundwater changes using averting expenditures: An application to groundwater contamination. *Land Economics* 68:163–169.

Baillie, J.E.M., C. Hilton-Taylor, and S.N. Stuart (eds.). 2004. 2004 IUCN Red List of Threatened Species. A Global Species Assessment. Gland, Switzerland and Cambridge, UK: IUCN.

Balmford, A., K. Gaston, S. Blyth, A. James and V. Kapos. 2003. Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. *Proceedings of the National Academy of Sciences* 100: 1046–1050.

Bockstael, N., A.M. Freeman, R.J. Kopp, P.R. Portney and V.K. Smith. 2000. On measuring economic values for nature. *Environmental Science and Technology* 34: 1384–1389.

Bell, K. P., D. Huppert, and R. L. Johnson. 2003. Willingness to pay for local Coho Salmon enhancement in coastal communities. *Marine Resource Economics* 18: 15–31.

Boyle, K.J., and R.C. Bishop. 1987. Valuing wildlife in benefit-cost analyses: A case study involving endangered species. *Water Resources Research* (23): 943–950.

Brock W.A., and A. Xepapadeas. 2003. Valuing Biodiversity from an Economic Perspective: A Unified Economic, Ecological, and Genetic Approach. *The American Economic Review* 93(5): 1597–1614.

Brooks, D. R., R.L. Mayden, and D.A. McLennan. 1992. Phylogeny and biodiversity: Conserving our evolutionary legacy. *Trends in Ecology & Evolution* 7(2): 55–59.

Brookshire, D., L. Eubanks, and A. Randall. 1983. Estimating option prices and existence values for wildlife resources. *Land Economics* 59: 1–15.

Brown, G. M., Jr., and J. F. Shogren. 1998 "The Economics of the Endangered Species Act." *Journal of Economic Perspectives* 12(3): 3–20.

Browder, John O. 1988. Public Policy and Deforestation in the Brazilian Amazon. In *Public Policies and the Misuse of Forest Resources*, edited by R. Repetto and M. Gillis. New York: Cambridge University Press, Brumer, Aaron, Richard E. Rice, and Gustavo da Fonseca. 2001. Beyond Fences: Seeking Social Sustainability in Conservation. Gland, Switzerland: IUCN.

Carson, R. T., and Hanemann, M. W. 2005. Contingent valuation. In *Handbook of Environmental Economics* (vol. 2), edited by K.-G. Mäler and J. R. Vincent. Amsterdam: Elsevier,

Chichilnisky, G., and G. Heal. 1998. Economic returns from the biosphere. *Nature* 391: 629–630.

Convention on Biodiversity. 2007. www.biodiv.org/2010-target/focal.shtml (accessed February 2, 2007).

Costanza, Robert, Ralph d'Arge, Rudolf de Groot, Stephen Farber, Monica Grasso, Bruce Hannon, Karin Limburg, Shahid Naeem*, Robert V. O'Neill, Jose Paruelo, Robert G. Raskin, Paul Sutton & Marjan van den Belt 1997. The Value of the World's Ecosystem Services and Natural Capital. *Nature* 387: 253–260.

Costello, C., and M. Ward. 2006. Search, bioprospecting, and biodiversity conservation. *Journal of Environmental Economics and Management* Vol 52, Issue 3: 615-626.

Council on Environmental Quality. 1980. Global 2000 Report to the President (vol. 2). Washington, DC: Council on Environmental Quality.

Daily, G.C. (ed.). 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington, DC: Island Press.

Dalton, R. 2004. Bioprospects less than golden. *Nature* 249:598–600.

Diamond, P.A., J. A. Hausman, G. L. Leonard, and M. A. Denning. 1993. Does contingent valuation measure preferences? Experimental evidence. In *Contingent valuation: a critical assessment, contributions to economic analysis*, edited by J.A. Hausman. New York: North-Holland, XX–XX.

Gaston, K. G. 2000. Global patterns in biodiversity. *Nature* 405: 220–227.

Giraud, K., B. Turcin, J. Loomis, and J. Cooper. 2002. Economic benefit of the protection program for the Steller sea lion. *Marine Policy* 26: 451–458.

Giraud, K., and B. Valcic. 2004. Willingness-to-pay estimates and geographic embedded samples: case study of Alaskan Steller sea lion. *Journal of International Wildlife Law and Policy* 7: 57–72.

Global Crop Diversity Trust. 2007. Regional Strategies: Americas: 2007. http://www.croptrust.org/main/regional.php?itemid=83(accessed March 2007

Grime, J. P. 1997. Biodiversity and ecosystem function: the debate deepens. *Science* 277:1260–1261.

Hammond, P. 1995. The current magnitude of biodiversity. In *Global Biodiversity Assessment*, edited by V.H. Heywood and R.T. Watson. Cambridge, U.K: Cambridge University Press, 113–138.

Hahnemann, W.M. 1994. Valuing the Environment Through Contingent Valuation. *Journal of Economics Perspectives* 8(4) 19–43.

Hanley, N., A. Konstantinos, D. Tinch, A. Davies, F. Watson, and E.B. Barbier. 2007. What Drives Long-term Biodiversity Change. Discussion paper 07-23. Washington, DC: Resources for the Future.

Heal, G.M., E.B. Barbier, K.J. Boyle, A.P. Covich, S.P. Gloss, C.H. Hershner, J.P. Hoehn, C.M. Pringle, S. Polasky, K. Segerson, and K. Shrader-Frechette. 2005. Valuing Ecosystem Services: Toward Better Environmental Decision Making. Washington, DC: The National Academies Press.

Heberlein, T.A., M.A. Wilson, R.C. Bishop, and N.C. Schaeffer. 2005. Rethinking the scope test as a criterion for validity in contingent valuation. *Journal of Environmental Economics and Management* 50: 1–22.

Hoehn, J. P., and J. B. Loomis. 1993. Substitution effects in the valuation of multiple environmental programs. *Journal of Environmental Economics and Management* 25: 56–75.

Holling, C.S., D.W. Shindler, B.W. Walker, and J. Roughgarden. 1995. Biodiversity in the functioning of ecosystems: an ecological synthesis. *Biodiversity Loss: Economic and Ecological Issues*, edited by C. Perrings, K.-G. Mäler, C. Folke, C. S. Holing, and B.-O. Jansson. Cambridge: Cambridge University Press,

International Union for the Conservation of Nature (IUCN), U.N. Environmental Program, and World Wildlife Fund.1991. *World Conservation Strategy: Living Resource Conservation for Sustainable Development*. Gland, Switzerland: IUCN.

IUCN. 2004. 2004 IUCN Red List of Threatened Species. www.iucn.org/themes/ssc/red_list_2004/gsaEXECSUMM_en.HTM (accessed February 7, 2007).

James, Alexander N., Kevin J. Gaston, and Andrew Balmford. 2001. Can We Afford to Conserve Biodiversity? *BioScience* 51: 43–52.

Kauppi, P.E., J.H. Assubel, J. Fang, A. Mather, R.A. Sedjo, and P.E. Waggoner. 2006. Returning Forests Analyzed with the Forest Identity. *Proceedings of the National Academy of Science* 103 (46): 17574–17579.

Kauppi, P., Sedjo, R., et al. 2001. Technological and economic potential of options to enhance, maintain, and manage biological carbon reservoirs and geo-engineering. In *Climate Change 2001: Mitigation*, edited by B. Metz et al. New York: Cambridge University Press, 303–353.

Kealy, M.J., and R.W. Turner. 1993. A test of the equality of the close-ended and the open-ended contingent valuation. *American Journal of Agricultural Economics* 75: 311–331.

Kennedy, T.A., S. Naeem, K.M. Howe, J.M.H. Knops, D. Tilman, and P. Reich. 2002. Biodiversity as a barrier to ecological invasion. *Nature* 417: 636–638.

Kindermann, G., M. Obersteiner, B. Sohngen, J.Sathaye, K. Andrasko, E.Rametsteiner, B.Schlamadinger, S. Wunder, R. Beach.. "On the Economics of Avoiding Deforestation." Unpublished paper.

Knops, J.M.H et al. Effects of plant species richness on invasion dynamics, disease outbreaks, insect abundance and diversity. *Ecological Letters* 2: 286–294.

Larson, D., and Siikamäki, J. 2006. Valuing Surface Water Quality in California: A Mixed Panel Logit Model Using a 2½-Bounded CVM. Paper contributed to the 3rd World Congress of Environmental and Resource Economics, July 3–7, Kyoto, Japan.

Laughland, A.S., W.N. Musser, J.S. Shortle, and L.M. Musser. 1996. Construct validity of averting cost measures of environmental benefits. *Land Economics* 72: 100–112.

Loomis, J.B., A. Gonzales-Caban, and R. Gregory. 1994. Do reminders of substitutes and budget constraints influence contingent valuation estimates? *Land Economics* 70: 499–506.

Loomis, J.B., and D.S. White. 1996. Economic benefits of rare and endangered species: summary and meta-analysis. *Ecological Economics* 18: 197–206.

MacArthur, R.H., and Wilson, E.O. 1967. *The Theory of Island Biogeography*. Princeton, NJ: Princeton University Press.

Mace, G.M., J.L. Gittleman, and A. Purvis. 2003. Preserving the Tree of Life. *Science* 300:1707–1709.

May, R.M. 1972. Will a large complex system be stable? *Nature* 238: 413–414.

May, R.M., J.H. Lawton, and N.E. Stork. 1995. Assessing extinction rates. In *Extinction Rates*, edited by J.H. Lawton and R.M. May. Oxford: Oxford University Press, 1–24.

McCann, K.S. 2000. The diversity-stability debate. *Nature* 405: 228–233.

McConnell, K.E., and Bockstael, N.E. 2005. Valuing the Environment as a Factor of Production. In *Handbook of Environmental Economics* (vol. 2), edited by K.-G. Mäler and J.R. Vincent. Amsterdam Elsevier,

McClelland, G.H., W.D. Schultze, J.K. Lazo, D.M. Waldman, J.K. Doyle, S.R. Elliot, and J.R. Irwin. 1992. Methods for measuring the non-use values: a contingent valuation study of the groundwater cleanup. Boulder, CO: Center for Economic Analysis, University of Colorado.

Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Synthesis. Washington, DC: Island Press.

Mitchell, R.C. and R.T. Carson. 1984. Willingness to pay for national freshwater improvements. Washington, DC: Resources for the Future.

National Oceanic and Atmospheric Administration. 2005. Endangered and Threatened Species; Designation of Critical Habitat for Seven Evolutionarily Significant Units of Pacific Salmon and Steelhead in California; Final Rule. *Federal Register* 70(170): 52488–52586.

NatureServe. 2003. Ecological Systems Database, version 1.02. Arlington, VA: NatureServe.

Nee, S., and May, R.M. 1997. Extinction and the Loss of Evolutionary History. *Science* 278: 692–694.

Newman, D.J., G.M. Cragg, and K.M. Snader. 2003. Natural Products as Sources of New Drugs Over the Period 1981–2002. *Journal of Natural Products* 66(7): 1022–1037.

Noss, R. F. 1990. Indicators for Monitoring Biodiversity: A Hierarchical Approach. *Conservation Biology* 4: 355–364.

Nunes, P. A. L. D., & van den Bergh, J. C. J. M. 2001. Economic valuation of biodiversity:. Sense or nonsense? *Ecological Economics*, 39: 203-222.

Phaneuf, D.J., and V.K. Smith. 2005. Recreation demand models. In *Handbook of Environmental Economics* (vol. 2), edited by K.-G. Mäler and J.R. Vincent. Amsterdam: Elsevier,

Pimentel, D., C. Wilson, C. McCullum, R. Huang, P. Dwen, J. Flack, Q. Tran, T. Saltman, and B. Cliff 1997. Economic and Environmental Benefits of Biodiversity. *BioScience* 47(11): 747–757.

Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000. Environmental and economic costs of nonindigenous species in the United States. *BioScience* 50(1): 53–67.

Polasky, S., C. Costello, and A. Solow. 2005. The Economics of Biodiversity. In *Handbook of Environmental Economics* (vol. 3), edited by K.-G. Mäler and J.R. Vincent. Amsterdam, Elsevier,

Polasky, S., E. Nelson, E. Lonsdorf, P. Fackler, and A. Starfield. 2005. Conserving species in a working landscape: land use with biological and economic objectives. *Ecological Applications* 15(4): 1387–1401.

Principe, P. 1989. *The Economic Value of Biodiversity Among Medicinal Plants*. Paris: OECD.

Rausser, G., and A. Small. 2000. Valuing research leads: bioprospecting and the conservation of genetic resources. *Journal of Political Economy* 108(1): 173–206.

Redwood III, John. 2002. World Bank Approaches to the Brazilian Amazon. LCR Sustainable Development working paper #13. Washington World Bank.

Ribaudo, M.O. 1989a. Water quality benefits from the conservation reserve program, agricultural economics report. Washington, DC: Economic Research Service.

Ribaudo, M.O. 1989b. Targeting the conservation reserve program to maximize water quality benefits. *Land Economics* 65: 320–332.

Sedjo, R.A. 1991. Toward a Worldwide System of Tradable Forest Protection and Management Obligations. Discussion paper ENR91-16. Washington, DC: Resources for the Future.

Sedjo, R.A. 1992a. Implications of a Tradable Obligations Approach to International Forest Protection (revised February 10, 1992). Washington. Report, Resources for the Future.

Sedjo, Roger A. 1992b. Property Rights, Genetic Resources, and Biotechnological Change. *Journal of Law and Economics* 35(1):

Sedjo, Roger A. (n.d.). An Emerging Market for Forest Ecosystem Services in the U.S. *Management of Environmental Quality* forthcoming.

Shabman, L., and S. Batie. 1978. The Economic Value of Natural Coastal Wetlands: A Critique. *Coastal Zone Management Journal* 4(3): 231–247.

Shannon C.E., and W. Weaver. 1949. *The Mathematical Theory of Communication*. Urbana, IL: The University of Illinois Press.

Siikamaki. J., and J. Chow. (n.d.). “Biodiversity”, in *Perspectives in Sustainable Resources in America*, R.Sedjo editor, forthcoming. Washington, DC: Resources for the Future.

Siikamäki, Juha, and David F. Layton. 2007. Potential Cost-Effectiveness of Incentive Payment Programs for the Protection of Non-Industrial Private Forests. *Land Economics* forthcoming.

Simon, J.L., and A. Wildawsky. 1984. On Species Loss, the Absence of Data, and Risks to Humanity. In *The Resourceful Earth. A Response to "Global 2000*," edited by J.L. Simon and H. Kahn. NY: Blackwell, 171 – 183.

Simpson, R. D. 2000. Economic perspectives on preservation of biodiversity. In *Conserving Nature's Diversity. Insights from biology, ethics, and economics*, Edited by G.C. van Kooten, E.H. Bulte, and A.R.E. Sinclair. Aldershot, UK.: Ashgate, 88-105.

Simpson, R. David. 2004. Conserving Biodiversity through Markets: A Better Approach. *PERC Special Issue* 32, July.

Simpson, R.D., and R.A. Sedjo. 1996. Commercialization of Indigenous Genetic Resources. *Contemporary Economic Policy* 12(4): 34–44.

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

Simpson E.H. 1949. Measurement of diversity. *Nature* 163: 688.

Smith, V.K. and W.H. Desvousges. 1986. *Measuring Water Quality Benefits*. Dordrecht, Netherlands: Kluwer Nijhoff Publishing.

Sittenfeld, Ana, and Rodrigo Gamez. 1993. Biodiversity Prospecting by INbio. In *Biodiversity prospecting: Using Genetic Resources for Sustainable Development,* edited by Walter V. Reid et al. Washington, DC: World Resource Institute.

Stevens, T., J. Echeverria, R. Glass, T. Hager, and T. More. 1991. Measuring the existence value of wildlife: what do CVM estimates really show? *Land Economics* 67: 390–400.

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

Svalbard International Seed Vault. 2007, http://www.croptrust.org/main/articles.php (accessed February 2007).

Terborgh, John. 2002. Overcoming Impediments to Conservation. In *Making Parks Work: Strategies for Preserving Tropical Nature*, edited by John Terborgh, Carel van Schaik, Lisa Davenport, and Madhu Rao. Washington, DC: Island Press, 243–249.

Thomas, C.D., A. Cameron, R.E. Green, et al. 2004. Extinction risk from climate change. *Nature* 427: 145–148.

Tilman, D. 1996. Biodiversity: Population versus ecosystem stability. *Ecology* 77(3):350–363.

Tilman, D. 2004. Niche tradeoffs, neutrality, and community structure: A stochastic theory of resource competition, invasion, and community assembly. *Proceedings of the National Academy of Sciences* 101:10854–10861.

Tilman, D., and J.A. Downing. 1994. Biodiversity and stability in grasslands. *Nature* 367:363–365.

Tilman, D., P. B. Reich, J. Knops, D. Wedin, T. Mielke, and C. Lehman. 2001. Diversity and productivity in a long-term grassland experiment. *Science* 294:843–845.

Toman, Michael. 1998. Why Not to Calculate the Value of the World's Ecosystem Services and Natural Capital. *Ecological Economics* 25: 61–65.

Torell, L. A., J. D. Libbin, and M. D. Miller. 1990. The market value of water in the Ogalla Aquifer. Land Economics 66: 163-175.

U.S. Department of Agriculture Forest Service. 2004. National Report on Sustainable Forests–2003. Washington DC: United States Department of Agriculture, Forest Service.

U.S. Department of Energy. 2007. Natural and Accelerated Bioremedation Research, http://www.lbl.gov/NABIR/index.html (accessed March 2007) U.S. Fish and Wildlife Service. 2002. Candidate Conservation Agreements With Assurances For Non-Federal Property Owners. Washington, DC: Department of the Interior, U.S. Fish and Wildlife Service.

U.N. Environmental Program. 1995. Global Biodiversity Assessment UNEP.

U.N. Food and Agriculture Organization. 2006. Global Forest Resources Assessment 2005, FAO Forestry Paper 147. Rome. FAO.

Walker, D.J., and D.L. Young. 1986. The effect of technical progress erosion damage and economic incentives for soil conservation. *Land Economics* 62: 83–93.

Weitzman, M. 1992. On diversity. *Quarterly Journal of Economics* 107(2): 363–405.

Weitzman, M. 2000. Economic profit versus ecological entropy. *Quarterly Journal of Economics* 115(1): 237–263.

Wells, M., and K.E. Brandon. 1992. People and Parks: Linking Protected Area Management With Local Communities. Washington, DC: The World Bank.

Western, D., and R.M. Wright (eds). 1994. *Natural Connections: Perspectives in Community-based Conservation*. Washington, DC: Island Press.

Wilcove, D., D. Rothstein, J. Dubow, A. Phillips and E. Losos. 1998. Quantifying threats to imperiled species in the United States. *BioScience* 48: 607–615.

Wilson, C., and C. Graham. 1983. *Exotic Plant Pests and North American Agriculture*.: Academic Press.

Wilson, E.O. (ed.). 1988. *BioDiversity*. Washington, DC: National Academy Press.

Wilson, E.O. 1992. *The Diversity of Life*. New York: W.W. Norton & Co.

World Conservation Monitoring Centre. 1992. *Global Biodiversity: Status of the Earth's Living Resources*. London: Chapman & Hall.

World Resources Institute, United Nations Development Programme, U.N. Environment Program, and the World Bank. 1998. *World Resources 1998-99*. New York: Oxford University Press.

Wu, J., and W.G. Boggess. 1999. The optimal allocation of conservation funds. *Journal of Environmental Economics and Management* 38: 320–321.