Forest Figures: Ecosystem Services Valuation and Policy Evaluation in Developing Countries

Paul J. Ferraro*, Kathleen Lawlor†, Katrina L. Mullan‡, and Subhrendu K. Pattanayak§

Introduction

Throughout human history, the services provided by ecosystems have been critical to the functioning and growth of the world's economies. This natural capital has been particularly important to the rural economies of developing nations, which often possess relatively little physical capital. Although the term *ecosystem services* entered the scientific lexicon only in the early 1980s, interest in the topic has grown dramatically. To illustrate, a literature search in Google Scholar on the term *ecosystem services* yields fifty-six publications for the 1986–88 time period, but 1,170 and 10,200 publications, respectively, for the 1996–98 and 2006–8 time periods. Although much of this literature focuses on high-income nations, searches on the term *ecosystem services* that include "Africa," "Asia," or "Latin America" as additional modifiers yield more than 13,000 publications for the 1996–2008 time period.

The concept of ecosystem services has become an organizing principle in international conservation practice and policy. In 2001, the Millennium Ecosystem Assessment (MA 2005) pooled the talents of more than 1,300 experts to assess the consequences of ecosystem change for human well-being and the scientific basis for action to conserve and enhance the contribution of ecosystem services to human well-being. The "ecosystem approach," promoted by the MA as a framework for environmental study and action, has become the primary framework for action under the international Convention on Biological Diversity. In the last ten years, most large international conservation organizations have created initiatives, hired specialists, and embraced rhetoric around the concept of ecosystem services. In particular, emerging international programs to reduce emissions from deforestation and degradation (REDD) in developing countries are reorienting forest conservation around delivery of

*Department of Economics, Andrew Young School of Policy Studies, Georgia State University, Atlanta, GA, USA

†University of North Carolina, Chapel Hill, NC, USA

‡University of California, Berkeley, CA, USA

§Duke University, Durham, NC, USA; Center for International Forestry, Bogor, Indonesia, and SANDEE, Nepal. E-mail: subhrendu.pattanayak@duke.edu.

Review of Environmental Economics and Policy, volume 6, issue 1, winter 2012, pp. 20–44
doi:10.1093/reep/rer019
Advance Access publication on December 20, 2011

carbon storage services. Moreover, in the last two years, motivated by the perceived success of the Intergovernmental Panel on Climate Change (IPCC), ecosystem advocates have initiated efforts to create an Intergovernmental Platform on Biodiversity and Ecosystem Services (http://ipbes.net/).

Scientists and policymakers in developing countries are leading this broad international movement to use ecosystem services as an organizing framework for science and action. For example, Costa Rica pioneered a national program to pay landowners for four ecosystem services in the mid-1990s (Programa de Pagos por Servicios Ambientales). In contrast, it was not until 2009 that the U.S. Department of Agriculture created an Office of Ecosystem Services and Markets and the U.S. Environmental Protection Agency created the Ecosystem Services Research Program Partnership. Thus international flows of information about the science and policy of ecosystem services often go from developing to developed nations. For example, the World Wildlife Fund and its partners recently created an exchange program through which U.S. ranchers travel to Namibia to learn about Namibians' success in harnessing ecosystem protection to promote economic growth.

Our research and policy-advising experience in developing countries leads us to conclude that the concept of ecosystem services has gained traction in scientific and policy circles for several reasons. The most important reason reflects the connotation associated with the term *services*, which differs from the connotations associated with terms that have been emphasized historically, such as *nature, species,* and *biodiversity.* Ecosystem services connote utilitarian values of nature, which resonate in developing nations, where reducing poverty and growing the economy are paramount concerns. Using the term *ecosystem services* rather than *biodiversity* also offers conservation practitioners access to economic development funds from international donors and national governments, which is a much larger finance pool than what is available for biodiversity conservation. More recently, the concept of ecosystem services has gained traction because of a predicted relationship between such services and climate adaptation capacity.

This article examines the evidence concerning the economic values of forest ecosystem services in developing nations and the effectiveness of policies aimed at securing these services. Because of space constraints, we are not able to address other terrestrial ecosystems (but believe our conclusions also hold for them), nonterrestrial ecosystem services, some sources of ecosystem value (e.g., option values that arise because there is value to preventing irreversible species loss in the face of uncertainty about future ecosystem values), and ecosystem services in developed nations.

The next section reviews the evidence base for values of forest ecosystem services. The literature review finds that credible valuations of ecosystem services in developing countries are rare and typically disconnected from policy options. The third section describes the evidence concerning the impacts of policies and programs designed to deliver ecosystem services in developing countries. Evaluations that apply modern empirical research designs to the most popular programs and policies, such as protected areas, decentralization and community management, and incentive payments, are also rare. Thus the evidence offers few clear guidelines for designing policies. The final section argues that the most fruitful path for future inquiry concerning ecosystems services is to integrate policy and research more tightly by conducting studies that combine nonmarket valuation and impact evaluation (i.e., valuation estimates based on observed impacts in the context of real-world programs).

How Valuable are Forest Ecosystem Services in Developing Nations?

The MA's *ecosystem approach* emphasizes the value of ecosystem processes to humans. Although not denying that intrinsic ecosystem values may also be important for decision making (MA 2005), the approach places an emphasis on economic valuation. Although economic valuation alone is neither necessary nor sufficient for successful policies (Heal 2000), it can provide important inputs into the policy process (Polasky et al. 2005). Such valuation can be used to (a) estimate the relative importance of various ecosystems, (b) justify or evaluate particular conservation decisions in particular places, (c) identify how the benefits of a particular conservation decision are distributed, and (d) identify potential sources of sustainable financing.

Valuation methods are usually referred to as nonmarket valuation methods because most ecosystem goods and services are not traded in conventional markets. Therefore a link must be established between an ecosystem process and a related market commodity. Establishing this link allows us to value services using economic demand theory, even if the ecosystem services are indirect, subtle, or latent (Pattanayak 2004).

Valuation Methods and Framework

The most common ecosystem service valuation methods are hedonic pricing, travel cost, productivity analysis, and contingent valuation. Most valuation exercises are embedded in a three-stage analytical framework that links ecosystem functions directly or indirectly to the well-being of people (Freeman 1993). The first stage measures how ecosystem flows (e.g., quantity or rate of streamflows) change as a result of some public policy or private action that alters ecosystem conditions. The second stage measures how changes in ecosystem flows affect the productivity or socioeconomic welfare of an individual. For example, there might be a change in household agricultural output, health, or income as a result of the change in the provision of a particular ecosystem service.

In the third stage, the productivity or welfare changes are expressed in monetary terms. Although this step is not always necessary, it can be important for effective communication with policymakers. When changes in productivity are directly related to market commodities, the monetary values will simply be the prices of the market commodities (e.g., coffee, electricity). In other cases, although the changes in productivity will not be market goods, they will be closely related to the production of such goods (e.g., time expended on water collection, lost labor from disease incidence). The prices associated with these related activities (e.g., time, labor) can be used to develop a "monetized" measure of the ecosystem service. When ecosystems contribute directly to an individual's utility (e.g., bequest or nonuse values), stated preference methods may be used to obtain monetary values.

To implement this valuation framework, at least three types of data are needed: (a) ecological data on the specific functions, processes, and outcomes (e.g., acres of protected forests); (b) production data to link the index of ecosystem flows to economic activities (e.g., the functioning of a reservoir for electricity and crop irrigation conditional on sedimentation rates); and (c) consumer preference (or producer technology in the case of intermediate outputs) and price data to express productivity changes in monetary terms. Statistical variation in each of the data types is needed to estimate parameters of the three functions associated with each of the stages in the framework. Data on these three types of linkages are typically lacking in developing nations, which may explain, at least in part, why there are fewer studies valuing ecosystem services in developing nations compared to developed nations.

Review of the Valuation Literature

Our review of the valuation literature is based on our collective research on this topic over the last fifteen years and on cross-referencing of prominent publications, including forward citations searches.[1] This foundation was supplemented with an extensive web search for additional studies. We then used four criteria to screen and assess the initial list of studies for economic credibility and econometric reliability. First, did the study use the previously mentioned three-stage approach in some shape or form (i.e., link policies to ecological changes to economic behaviors and ultimately to welfare impacts)? Second, did the study carefully explain how changes in ecosystem quantity or quality changed ecosystem service flows? Third, did the study use well-established nonmarket valuation methods or socio-economic responses to link this ecological change to monetary outcomes? Finally, were multivariate statistical approaches used to control for covariates, especially any behavioral responses that might confound or undermine the link between the policy-induced ecological change and the economic impact?

Although we sought studies that satisfied all four criteria, we included a study as long as it clearly satisfied the third criterion and marginally satisfied two other criteria. As discussed later, most studies did not clearly identify the relevant policy/action change (the first criterion), and few addressed the second criterion. Although our selection procedure was somewhat liberal, we did not, for example, include contingent valuation studies that failed to identify the ecological processes and policy context underlying the hypothetical ecological changes. We also excluded studies that report nontimber forest product (NTFP) values based on botanical inventories of commercial species (e.g., Peters et al. 1989) because they overstate values by ignoring the realities of limited demand and price elasticity (for other problems, see Chomitz and Kumari 1998 and Sheil and Wunder 2002).

The literature review is organized around six types of ecosystem services: carbon storage, ecotourism, hydrological flows, pollination, health, and NTFPs. The studies that met our criteria are presented in Tables 1–4. We have not included tables for carbon storage or pollination because, in the case of the former, there were too many studies to present in a table, and, in the case of the latter, there were too few studies to warrant a separate table. We present values exactly as they are reported in the studies and do not convert them to a common-year equivalent. All values are reported in US dollars ($). We order our presentation according to the scale at which benefits from the services are enjoyed, beginning with the most global (carbon storage) and ending with the most local (NTFPs). We acknowledge the current debate surrounding classification of ecosystem services and critiques of the MA's approach, which mixes ends (benefits) and means (services) (see Boyd and Banzhaf 2007). We also

[1]A forward citation search typically involves searching in electronic databases (e.g., Google Scholar, Web of Knowledge) for articles that cite a seminal article. These articles may in turn apply the methods or address the ecosystem services that were considered in the seminal article, thereby expanding the literature review.

recognize as helpful Fisher and Turner's (2008) distinction between "final services" (e.g., water regulation, primary productivity) and "benefits" (e.g., water for irrigation, drinking water, hydroelectricity, NTFPs). Our organization of the ecosystem services literature is not intended to be a new classification scheme, and we recognize the potential risks of double counting when health benefits that are a function of water quality and quantity are catalogued separately from other hydrological benefits. The findings of our literature review are presented here according to each type of ecosystem service.

Carbon Storage

Tropical forests play a critical role in regulating both the global and local climates by storing huge quantities of carbon and regulating localized precipitation and temperature patterns. The carbon released from the cutting and burning of forests accounts for a significant share of global greenhouse gas emissions: about 12 percent, according to the most recent study (Van der Werf et al. 2009). Many environmentalists believe that valuing forests for their carbon storage ability will send a price signal that is strong enough to protect these ecosystems (Pearce 2001). This has led to the development of REDD programs to pay for forests' carbon storage, with financing channeled through voluntary carbon markets and public funds. REDD programs may also eventually include carbon offsets for cap-and-trade systems in developed nations.

Estimates of the value of forests' carbon storage range from \$378 per hectare (ha) in Paraguay's Atlantic forests (Naidoo and Ricketts 2006) to \$1500/ha on Borneo (Naidoo, Malcolm, and Tomasek 2009). Three factors affect the carbon storage values reported in the literature. First, per hectare carbon stocks vary greatly across regions and specific sites. Biome-specific averages in the tropics range from 72 tons of carbon (tC)/ha in African dry forests to 225 tC/ha in Southeast Asian rainforests (not including soil) (Gibbs et al. 2007). Mean values, however, obscure considerable variation within biomes based on elevation, slope, and recent anthropogenic degradation. This variation has led to a preference for field measurements. Second, estimating carbon storage requires estimating the status quo deforestation levels in order to calculate avoided emissions and damages. However, estimating status quo deforestation is difficult because forest transition trajectories are uncertain. Deforestation drivers are complex and the past may not predict the future (Angelsen 2008). Third, the price used to monetize tons of carbon dioxide varies greatly. Typically, either a carbon market price or an estimate of the social damages from emissions is used (and there is a range of prices for each option). It is important to note that carbon market prices do not necessarily reflect the social value of avoiding forest carbon emissions. Rather, carbon market prices reflect what project proponents might be able to earn based on political decisions in developed nations regarding their commitments to reduce emissions and allowances for pollution (Convery and Redmond 2007). Applying estimates from climate change damage functions may be more appropriate for valuation, although some contend that this approach yields excessively high values for carbon storage (Pearce 2001).

Table 1. Empirical evidence on forest ecotourism

				Δ Welfare	
Reference	**Country**	**Δ Quantity**	**Valuation method**	**Average value per visitor**	**Aggregate value for all visitors**
Bienabe and Hearne (2006)	Costa Rica	Ecotourism (and nonuse benefits)	Choice experiment	WTP for '1 level' increase in protection of scenic beauty: Costa Rican residents: $0.25/month Foreign tourists: $3.36 (one-off)	Not calculated
Chase et al. (1998)	Costa Rica	Ecotourism	Contingent valuation	WTP: $22–$25 per person	Not calculated
Echeverría, Hanrahan, and Solórzano (1995)	Costa Rica	Ecotourism	Contingent valuation	Mean WTP: Non-Costa Ricans: $137.41Costa Ricans: $118.76	Aggregate WTP to protect park by all visitors: $37,517,374
Ellingson and Seidl (2007)	Bolivia	Ecotourism	Contingent valuation and contingent behavior	Mean WTP: $76.50 (CB); $36.73(CV)	Total expenditure: $2.2m (CB); $1.9m(CV)
Holmes et al. (1998)	Brazil	Ecotourism	Choice experiment	WTP for new parks: $22–$86/person depending on the extent of tourist facilities.	Not calculated
Menkhaus and Lober (2006)	Costa Rica	Ecotourism	Travel cost	Estimated consumer surplus per person: $1150	Total value of all ecotourist visits to Costa Rica by US residents: $68 million
Naidoo and Adamowicz (2005)	Uganda	Ecotourism	Choice experiment	Mean WTP entrance fees per visitor: $46	Maximum total revenue from park fees with 20 bird species seen: $18,032 Maximum total revenue from park fees with 80 bird species seen: $40,423
Tobias and Mendelsohn (1991)	Costa Rica	Ecotourism	Travel cost	Mean WTP per person: $35	Annual consumer surplus from all domestic visits to reserve: $97,500–$116,200

Ecotourism

The second service we consider is the opportunity for ecotourism or "responsible travel to natural areas that conserves the environment and improves the welfare of local people."[2] Table 1 presents consumer surplus estimates from forest-based ecotourism. Visitors are found to value the biodiversity and scenic beauty attributes of forested sites at $20 to $140 per person. Infrastructure such as accommodations and trails are found to enhance the benefits of ecotourism in natural forests (Holmes et al. 1998). For example, Monteverde Cloud Forest Reserve, which is the subject of three studies in Table 1, has a range of human-built features that are not typical of the average tropical forest (Plummer 2009). Thus it may be appropriate to view the forest as an input into the production of ecotourism activities.

The ecotourism literature describes how the benefits of forest ecotourism are likely to vary across different visitor types. Bienabe and Hearne (2006) and Echeverría, Hanrahan, and Solórzano (1995) survey foreign and domestic visitors, and they find that willingness to pay (WTP) for ecotourism is higher among foreign visitors because of differences in incomes and other individual characteristics. The characteristics of the country in which the forest is located will also affect the potential ecotourism benefits. Costa Rica, which is the subject of over half of the available studies, is particularly attractive for ecotourists due to its political stability and quality of infrastructure. Thus the value of ecotourism benefits may be lower in other developing nations.

Most studies in Table 1 estimate the value of ecotourism benefits from forests that have been designated as parks or reserves. These protected areas tend to be particularly diverse or unique areas of forest, or they contain additional attractions such as mountains or beaches. Chase et al. (1998) estimate the cross-price elasticity of parks with forest and volcanoes and parks with forest and beaches. Their results indicate that the nonforest attributes contribute significantly to the benefits experienced by visitors. They also suggest that tourists view similar parks as substitutes for one another, which implies that the ecotourism benefit estimates from individual parks are not necessarily additive either within or across countries.

Hydrological Services

Forests provide multiple hydrological, or watershed, services, including prevention of soil erosion, regulation of water flows, and water purification. As shown in Table 2, few studies have estimated values of hydrological services. We believe that the paucity of studies is due at least partly to the challenges of interdisciplinary collaboration. These challenges have caused some authors to focus only on the impact of changes in forest cover on hydrological outcomes (e.g., Ataroff and Rada 2000; Williams, Fisher, and Melack 1997) or the value of hypothetical outcomes for local populations (e.g., Wang et al. 2007). Our review includes only those studies that attempt to quantify the full relationship between forest protection and hydrological benefits.

[2] http://www.ecotourism.org/site/c.orLQKXPCLmF/b.4835303/k.BEB9/What_is_Ecotourism__The_International_Ecotourism_Society.htm.

Table 2. Empirical evidence on forest hydrological services

Reference	Country	Δ Quantity	Valuation Method	Δ Welfare
Barkmann et al. (2008)	Indonesia	Availability of water for wet rice irrigation	Choice experiment	Marginal WTP for one month less of water scarcity: $4.29/household/year Aggregate WTP across project area: $96,000/year
Guo et al. (2007)	China	Avoided sedimentation and water regulation for Three Gorges Hydroelectric Power Plant	Change in productivity	NPV of water regulation benefits: $21.9 million NPV of reduced sedimentation: $15.1 million
Klemick (2011)	Brazil	On-farm and downstream hydrological regulation benefits from forest fallow	Production function	Increase in on-farm output value from 1 ha fallow: 0.5–0.7% ($17.6–$24.3/household/year) Increase in downstream farm output value from 1 ha fallow: 0.1–0.3% ($3.6–$10.3/household/year)
Lele et al. (2008)	India	Water for irrigation	Change in productivity	Reduction in expected annual income resulting from changes in forest cover: $107/household Reduction in expected value of total annual income for region: $15,360
Núñez, Nahuelhual, and Oyarzún (2006)	Chile	Streamflow as an input to production of drinkable water	Change in productivity	$15.4/household (summer); $5.8/household (rest of year) $162.4/ha of native forest (summer); $61.2/ha of native forest (rest of year)
Pattanayak and Mercer (1998)	Philippines	Agroforestry reduces soil erosion	Profit function	
Pattanayak (2004)	Indonesia	Availability of water for domestic uses	Hedonic cost function	10% increase in base flow results in reduction in collection costs of $0.003 Savings in water collection costs per household: $0.11–$0.23. Negative in some locations.

Continued

Table 2. *Continued*

Reference	Country	Δ Quantity	Valuation Method	Δ Welfare
Pattanayak and Kramer (2001a)	Indonesia	Drought mitigation	Profit function	Marginal annual profit: \$0.36/mm of base flow. Simulated incremental impacts of increased forest cover vary from \$3.5–\$35/household/year (1–10% of average annual farm profits). Negative in some locations.
Pattanayak and Kramer (2001b)	Indonesia	Drought mitigation	Contingent valuation	Mean WTP attributable to perceived increase in annual profits from drought control: \$1.97/household/year
Veloz et al. (1985)	Dominican Republic	Avoided sedimentation of hydroelectric dam	Change in productivity	NPV of additional electricity production: \$2.7 million

Even among these studies, the focus tends to be on individual hydrological services, rather than the full range. Núñez, Nahuelhual, and Oyarzún (2006) and Pattanayak (2004) estimate the value of the contribution of forests to the production of drinking water. Most other studies in Table 2 consider the benefits to farmers of drought mitigation and water regulation using farm-level profit functions, household labor allocation, or choice experiments. Positive values for these services range between \$0 and \$35 per household per year, with values concentrated between \$2 and \$10 per household per year. However, Lele et al. (2008) find that forest regeneration results in losses in expected income rather than gains. The remaining hydrological studies look at the benefits of forest cover to the operation of hydroelectric dams. Guo et al. (2007) predict that protection of existing forest cover in the watershed around the Three Gorge Hydroelectric Plant would generate net present benefits of \$22 million from water regulation and \$15 million from reduced sedimentation. Veloz et al. (1985) estimate the benefits to a single power plant in the Dominican Republic of reduced sedimentation alone at \$2.7 million over twenty-five years.

These benefits are highly location specific for two reasons. First, local geophysical and climatic conditions determine the extent to which sedimentation or water scarcity are problems. Second, the economic benefits from these hydrological changes depend on their spatial relationships to human activities, such as the presence of adjacent farming activity, downstream residential populations demanding clean drinking water, or downstream hydropower plants.

Pollination

Proximity to forests can increase the productivity of agricultural land due to the pollination services provided by wild pollinators. These benefits can apply to a wide range of crops, including nuts, fruits, flowers, and oils (see Ricketts et al. [2008] for a review). However, the studies that attempt to quantify the economic value of pollination services have focused on coffee production. Ricketts et al. (2004) find that a single large coffee farm obtains benefits of \$62,000 per year from neighboring forest fragments, which is equivalent to 7 percent of annual profits. Priess et al. (2007) consider a larger landscape and estimate that without full forest protection, local coffee farmers would experience losses of 0.3 to 13.8 percent of net revenues over twenty years, implying a mean benefit of \$63/ha. Although pollination studies show how benefits vary with distance from the forest edge, it is unclear how benefits vary with the area of protected forest.

Human Health

Forests provide numerous health services to humans, some of which we are only beginning to understand (Pongsiri et al. 2009). An emerging body of evidence shows that intact tropical forests regulate the spread of vector-borne diseases such as malaria and dengue, especially in the context of rising temperatures and rainfall changes (Patz et al. 2005; Vittor et al. 2006). Some health benefits are a function of forests' hydrological services. For example, reductions in streamflow in deforested watersheds can increase diarrhea rates in rural communities because of lack of access to other sources of water for sanitation (Pattanayak and Wendland 2007). Deforestation and degradation through large-scale fires result in haze-related air pollution that can cause respiratory illnesses (Frankenberg, McKee, and Thomas 2005).

Forests also contribute to human health through the provision of NTFPs for food and medicine (Colfer, Sheil, and Kishi 2006).

Despite the fact that much of the early work aimed at valuing nonmarket benefits of environmental policies focused on human health (Bockstael et al. 2000) and that the health benefits provided by forests may be some of the most important given the large number of people that stand to be affected (Pattanayak et al. 2006), there is a dearth of studies valuing forests' health services. As with hydrological services, we believe that this is because the underlying ecological–epidemiological links are still not well understood. For example, there is uncertainty regarding how much of the deforestation–malaria link is ecological (i.e., clearing increases mosquito breeding) and how much is behavioral (i.e., deforestation is often accompanied by an influx of migrants who may not know or be in the habit of taking precautions against malaria) (Pattanayak and Yasuoka 2008). We found ten studies valuing forests' health benefits (see Table 3). We included studies that report values in terms of health benefits, but we did not convert these benefits to monetary terms. We also include a study that reports the lifesaving value of mangroves (Das and Vincent 2009) in the context of cyclone storm surges.[3]

Nontimber Forest Products

Millions of people in the developing world live in or adjacent to forests, and they harvest NTFPs for food, fuel, construction, and medicine to satisfy subsistence and income needs. Although the NTFP valuation literature is large, we found only twelve studies that met our screening criteria (see Table 4). Most of these studies used direct-use methods, which measure households' collection of NTFPs for sale and consumption and then convert these quantities to values by applying local prices (for the NTFP itself, close substitutes, or wage rates for harvesting labor). However, direct-use methods do not necessarily reveal preferences; that is, an individual's WTP to maintain the service or that individual's willingness to accept (WTA) to forgo it (Bockstael et al. 2000).

Exceptions include Shyamsundar and Kramer (1996), who examined households' WTA (in baskets of rice) for forgoing forest access due to establishment of a national park in Madagascar. Because the study's scope encompassed both values of harvesting NTFPs and clearing forest for agriculture, the results likely overstate the value of standing forest. Pattanayak and Sills (2001) and Pattanayak et al. (2004) use travel cost logic—the choice made by households to spend time collecting NTFPs—to derive values for NTFPs.

For those studies that report valuation estimates per hectare, the range of values for NTFPs is \$6 to \$35 per hectare/year. In poor communities isolated from markets, NTFPs account for a significant share of household consumption and earnings and provide a safety net against economic shocks, such as crop failure. However, the contributions of NTFPs are minor for households in communities with access to robust commodity markets.

Do Policies Deliver Forest Ecosystem Services in Developing Nations?

The previous section suggests that the literature on rigorous valuation and measurement of ecosystem services is rather thin. Despite this paucity of empirical data on the relationship

[3]Barbier (2012) provides a more detailed discussion of ecosystem services from mangroves.

Table 3. Empirical evidence on forest health services

Reference	Country	Δ Quantity	Valuation methods	Δ Welfare
Anaman and Ibrahim (2003)	Forests: Indonesia Health impacts: Brunei Darussalam	Avoided respiratory disease caused by air pollution from large-scale forest fires	Avoided damages	1 unit increase in Pollution Standard Index = increase in respiratory diseases, valued at $1,264/day in social costs (hospital visits and lost labor)
Das and Vincent (2009)	India	Mortality because of storm surges avoided because of mangroves	Avoided mortality	Clearing mangroves causes 1.72 additional deaths per village, valued at $318,000–$1.4 million per death (Each 1 ha of remaining mangroves saved 0.0148 lives)
Frankenberg, McKee, and Thomas (2005)	Forests: Indonesia Health Impacts: Indonesia	Avoided health problems caused by air pollution from large-scale forest fires	Avoided damages	Exposure to haze adversely affects health
Jayachandran (2009)	Indonesia	Infant mortality health problems caused by air pollution from large-scale forest fires	Avoided mortality	Exposure to air pollution causes infant mortality
Pattanayak and Wendland (2007)	Indonesia	Avoided diarrhea caused by reduced water quantity	Avoided morbidity	1 unit increase in base flow decreases diarrhea incidence by 2,600 cases/year, valued at $5,900 savings in annual medical costs
Pattanayak and Yasuoka (2008)	Indonesia	Avoided malaria correlated with extent of disturbed forest	Avoided morbidity	More disturbed forest associated with higher incidence of malaria in children under 5 years of age
Pattanayak et al. (2009)	Brazil	Dengue and malaria caused by mosquito habitat increase due to deforestation	Avoided morbidity and mortality	1 million hectare reduction in deforestation would reduce malaria rates by 2.7 per 1,000 and dengue by 0.1 per 1,000 in rural areas
Saha et al. (2011)	India	Respiratory infections caused by air pollution from mining-induced deforestation and degradation	Avoided morbidity	Living 1 km closer to mines is associated with a 2.7% increase in log-odds of respiratory infections
Sanglimsuwam et al. (forthcoming)	India	Diarrhea and typhoid caused by water pollution from mining-induced deforestation and degradation	Avoided morbidity	Distance from mines reduces incidence of diarrhea and typhoid
Sastry (2002)	Forests: Indonesia Health impacts: Malaysia	Avoided mortality caused by air pollution from large-scale forest fires	Avoided damages	100 μ/m^3 increase in PM_{10} = increase in mortality risk of 1.07 in Kuala Lumpur

between ecosystem services and human welfare in developing nations, policymakers and practitioners are continuing to design and implement programs and policies to supply ecosystem services. Thus it is also important to examine what we know about the effectiveness and cost effectiveness of these efforts to deliver ecosystem services and their impacts on human welfare. Given that ecosystem services is a relatively new concept, no studies directly address this issue. Nonetheless, there is a long history of policies and programs aimed at protecting ecosystems for their biodiversity and NTFPs. Efforts aimed at supplying ecosystem services generally comprise these same interventions, with the notable exception of the new concept of "payments for environmental services." In this section, we review the findings of studies that have evaluated the impacts of interventions designed to slow or reverse ecosystem degradation, including protected areas, decentralization of management authority, and payments for environmental services.

Protected Areas

Defining ecosystems as "protected" and restricting access to them is the most common deliberate policy used to protect ecosystems globally (MA 2005). This command-and-control strategy is now applied to more than 13 percent of the terrestrial area of developing nations (WDPA 2009). The theory underlying the use of protected areas is simple: legal restrictions prevent anthropogenic disturbance, thus contributing to the maintenance or recovery of ecosystem services. Protected area impacts are diminished by assigning protection to unthreatened areas, by noncompliance with legal restrictions, and by spillovers to unprotected areas (e.g., leakage).

Despite the dramatic growth in the extent of protected areas globally in the last three decades,[4] little empirical work has documented their effectiveness in reducing anthropogenic disturbances (see Albers and Ferraro 2006; Joppa and Pfaff 2010, for reviews). Much of the empirical evidence falls into two categories: (a) trends of indicators (e.g., forest cover) inside protected areas over time; and (b) cross-sectional comparisons of indicators inside and outside of protected areas. There is strong evidence, however, that protection is assigned conditional on baseline ecosystem and community characteristics that also affect human use of the ecosystems (Joppa and Pfaff 2009; MA 2005, p. 130). Thus there is likely to be severe selection bias in most of the literature. For example, Andam et al. (2008) show that estimates of avoided deforestation from Costa Rica's protected area network range from 20 to 50 percent of the forest protected when the analysis fails to control for baseline characteristics that affect both deforestation and where the protected areas were established. When the analysis controls for these characteristics, the estimates range from 8 to 12 percent.

There are fewer than a dozen studies in seven developing nations that attempt to control for selection bias (see reviews in Albers and Ferraro 2006; Joppa and Pfaff 2010) and two global studies that use coarse data and some postprotection covariates (Joppa and Pfaff 2011; Nelson and Chomitz 2011). With the exception of Nelson and Chomitz (2011), which measures forest fires, these studies measure the outcome of forest cover loss. They generally conclude that protection has led to reductions in ecosystem disturbance but at much lower levels than conservation scientists have claimed because protection tends to be assigned to ecosystems at

[4]http://www.unep-wcmc.org/wdpa/PA_growth_chart_2007.gif.

Table 4. Empirical evidence on nontimber forest products values

Reference	Country	Δ Quantity	Valuation method	Δ Welfare: Average annual value/ha	Δ Welfare: Average contribution to household consumption and earnings
Campbell, Luckert, and Scoones (1997)	Zimbabwe	Selected range of marketed products	Market values of direct use	\$5–\$17	Not calculated
Cavendish (2000)	Zimbabwe	Food, fuelwood, household items, fertilizers, livestock grazing	Market values of direct use	Not specified	35–37%; greater for poorest quintile (39–43%)
Godoy et al. (2000)	Honduras	Fish; game; plants for food, medicine construction, craft; fuelwood; timber	Market values of direct use	\$6–\$8 (\$18–24 PPP[a])	Not calculated
Godoy et al. (2002)	Bolivia and Honduras	Fish, game, food, fuelwood, timber	Market values of direct use	\$7–\$10 (\$18–\$47 in PPP[a])	39% of consumption; 23% of earnings; in general, contribution is greater in those villages farther from markets
Gram (2001)	Peru	Fish, game, fruits and other food, fuelwood, timber	Market values of direct use	\$9–\$17	Not calculated
Heubach et al. (2011)	Benin	Plants only	Market values of direct use	Not specified	\$380/adult equivalent unit/year 39% of household income; 49%, 44%, and 33% for low-, middle-, and high-income groups, respectively
Pattanayak and Sills (2001)	Brazil	Food and construction (fuelwood and certain fruits excluded); households make about six collection trips/year	Model of household labor demand, based on forest collection trips and risk of crop failure	Not specified	Forests provide risk mitigation to all households, especially to the poorest households
Pattanayak et al. (2004)	Indonesia	Fuelwood; households make 218 collection trips	Travel costs as input to household production model	Not specified	\$0.19/collection trip Average annual consumer surplus: \$122/household

Continued

Table 4. *Continued*

				Δ Welfare	
Reference	**Country**	**Δ Quantity**	**Valuation method**	**Average annual value/ha**	**Average contribution to household consumption and earnings**
Pattanayak et al. (2010)	India	Cash income and diet	Production function	Not specified	Forest quality and quantity increase cash income and diet (direct consumption)
Shackleton et al. (2002)	South Africa	118–208 species used for food, fuelwood, household items, medicine, construction	Market values of direct use	Not specified	$469–$1,206/household/year; majority live below poverty line of $1,476/family of five per year
Shone and Caviglia-Harris (2005)	Brazil	Honey, fish, fruits	Market values of direct use	$17–$35	Only 5% (due to high returns from cattle and agriculture)
Shyamsundar and Kramer (1996)	Madagascar	Access to land for NTFP and agriculture	Contingent valuation (WTA)	Sustained access to forest avoids welfare loss of $50/household/year	

[a]PPP, purchasing power parity.

below-average risk of disturbance. Only one study considers the potential bias (positive and negative) from spatial spillovers as a result of land use regulations (Andam et al. 2008). Little empirical research has been conducted on the effects of heterogeneous land use restrictions across a protected area system (e.g., Nelson and Chomitz 2011; Sims 2010) or the effects of different types of management of protected areas, such as comparisons of protected areas run by government, nonprofit organizations, and indigenous communities (Somanathan, Prabhakar, and Mehta 2009). No study has examined the cost effectiveness of protected areas.

The net socioeconomic impacts of protected areas in developing nations have also not been adequately assessed (Andam et al. 2010; Sims 2010). Most studies are simple case study narratives or *ex ante* projections based on extrapolations of historical economic activity. As noted by others (Coad et al. 2008; Wilkie et al. 2006), most studies prove little more than that protected areas are established near poor people and provide both opportunities and constraints to economic development.

Decentralization and Community-Based Natural Resource Management

In many developing nations in the nineteenth and twentieth centuries, forests became the property of the state and management responsibilities shifted from forest users' traditional common property regimes to centralized state authorities. Over the past two decades, however, a decentralization trend has taken hold, with states devolving forest ownership and management rights to local institutions (Sunderlin, Hatcher, and Liddle 2008). This decentralization effort is driven by theoretical work in economics and political science that argues that groups of forest users can, under certain conditions (including secure property/management rights and democratic institutions), sustainably manage forests (Ostrom 1990; Ribot 2002).

However, there are few credible empirical tests of these theories. Much of the existing evidence comes from qualitative case studies. The handful of empirical analyses that do exist fail to control for selection bias, that is, confounding factors that affect forest management and where, and to whom, management authority is devolved (e.g., Chhatre and Agrawal 2008, 2009; Persha, Agrawal, and Chhatre 2011). Two recent reviews find that despite hundreds of studies on community-based forest management, there is little credible evidence that local institutions have had a more positive environmental impact than government management (Bowler et al. 2009; Lund, Balooni, and Casse 2009). After an exhaustive review, Bowler et al. (2009) found only eight studies that made any attempt to control for selection bias. These studies (e.g., Baland et al. 2010; Edmonds 2002; Somanathan, Prabhakar, and Mehta 2009) find zero or small impacts (not always positive) on indicators such as basal stem area or forest cover. The evidence base is even thinner regarding decentralization's socioeconomic impacts. Bowler et al. (2009) and Lund, Balooni, and Casse (2009) identify only one study that examines the topic and addresses confounding variables, Jumbe and Angelsen (2006), which finds that the impact of decentralization on earnings in two community reserves is heterogeneous.

Payments and Markets for Ecosystem Services

Economic theory suggests that some form of transfer between the beneficiaries and the providers of ecosystem services, whether through government subsidies or voluntary buyer-seller arrangements (i.e., co-Asian contracting), could move the quantity of ecosystem services supplied closer to the social optimum. This theory has been put into practice in the form of "payments for environmental services" (PES) or, more broadly, "markets for environmental services" (MES). Although the details vary in practice, all forms of these incentive-based approaches depend on a financial transfer to suppliers conditional on the supply of ecosystem services (or on actions that are believed to generate services).

Using PES to protect ecosystems has been particularly popular in developing nations for four reasons: (a) the political economy and weak institutional environment in these nations makes the use of regulations or incentive-based quantity instruments (e.g., tradable development rights) difficult and leads to a preference for subsidies to achieve environmental outcomes; (b) it is becoming increasingly acceptable to use conditionality and performance measures to distribute aid and subsidies (e.g., conditional cash transfers); (c) there is a belief that PES can achieve both poverty alleviation and ecosystem protection (and attract donor funds), thereby providing win-win opportunities for a nation; and, perhaps most importantly, (c) there is a belief, particularly among international aid donors, that imperfect information about ecosystem values and high start-up costs for PES schemes are preventing voluntary contracting between ecosystem service beneficiaries and suppliers, and that thus with short-term investments in information by outsiders and fixed start-up costs, PES could become self-financing.[5]

Theory also indicates, however, that PES may not be effective in achieving either environmental protection (Ferraro 2008) or poverty alleviation (Wunder 2008). The effectiveness of PES depends on the impact of the program design on where and to whom the payments go and the degree of compliance and spatial spillovers.[6] For example, asymmetric information and poor administrative targeting can result in direct payments to unthreatened lands of little environmental value. Moreover, administrative targeting and variability in property rights, human capital, and political power can affect the impact of PES on poverty and economic growth.

In practice, issues related to adverse selection and moral hazard are often neglected in the design of PES programs.[7] In addition, PES programs and their monitoring systems are often designed in ways that make estimating causal impacts difficult. Pattanayak, Wunder, and Ferraro (2010) review the evidence concerning the effectiveness of PES in achieving environmental and socioeconomic policy goals. They find that few studies have credible empirical designs (e.g., baselines and valid control groups) to evaluate the impact of PES, and most focus on one country (Costa Rica; e.g., Arriagada et al. forthcoming). They observe that government-financed PES have resulted in modest or no reversal of deforestation (likely

[5]However, it is never clearly explained how short-term financing can resolve the public good nature of the services.

[6]Spillovers can be negative, such as agricultural displacement. They may also be positive, such as through the creation of option value on nonenrolled lands when the decision to convert the ecosystem to alternative uses is irreversible, a budget-constrained PES program has excess demand for contracts, and landowners expect that the budget may increase in the future.

[7]Adverse self-selection arises because PES is voluntary and suppliers naturally volunteer their lowest value resources for PES, which implies that payments may go to unthreatened forests or forests of low environmental value. Moral hazard arises because contract compliance is costly to enforce and contract recipients can take private unobservable actions to shirk their contract obligations.

because of adverse selection). Case studies of user-financed, smaller scale PES schemes claim more substantial impacts, but these schemes have not been evaluated using rigorous empirical frameworks. Pattanayak, Wunder, and Ferraro (2010) also identify many unresolved issues, such as the importance of preconditions (e.g., land tenure, full information) and "crowding out" of intrinsic incentives. There is even less empirical evidence from developing nations concerning MES interventions more broadly (e.g., eco-certification; tradable development rights).[8]

Economic and Planning Policies

Although the three types of policies and programs just discussed are among the most popular approaches to ecosystem protection, there are many others. Space constraints prevent us from describing these other approaches in detail, but many of them are based on affecting economic growth paths and thus indirectly affecting ecosystem management. Examples include road building decisions, the elimination of so-called perverse subsidies (e.g., fuel subsidies), the provision of alternative livelihoods or substitute goods through development programs (Ferraro and Simpson 2002), conservation education based on moral and economic values, public works programs (e.g., South Africa's Working for Water program), and international efforts to affect investment and production activities in developing nations (e.g., debt-for-nature swaps; product boycotts). With the exception of some econometric work that found a positive but heterogeneous impact of roads on deforestation (see Pfaff et al. 2010 and references therein), we found no empirical evidence concerning the impacts of these efforts.

Conclusions

This article has examined what is known about the economic values of forest ecosystem services in developing nations and the effectiveness of policies implemented to secure these services. Despite a plethora of publications in the last ten years, we found few well-designed studies that provide a coherent picture of either ecosystem values or policy effectiveness. Our review leads us to draw three main conclusions: two that characterize the status quo and one that describes a potentially more fruitful path for future inquiry in this area.

1. Valuation of Ecosystem Services is Rare and Disconnected From Policy

Our review of ecosystem service valuation studies generates five observations. First, despite the apparent richness of the literature suggested by simple searches in Google Scholar, few studies rigorously measure and value ecosystem services. We reach this conclusion despite including papers that only tangentially satisfy our screening criteria. Second, even the more "rigorous" valuation studies use multivariate regression models and essentially rely on structural form econometrics to derive welfare estimates that reflect ecosystem values. Because most of these studies use cross-sectional data sets, the estimated parameters and the associate values are subject to the usual set of biases. Third, there have been few efforts

[8]See Blackman et al. (2009) for a review on eco-certification.

to compare ecosystem service benefits with costs of service delivery (Naidoo and Ricketts 2006). Thus it is difficult to determine whether the available data (Tables 1–4) indicate that the value of ecosystem services is economically significant and if the provision of ecosystem services is optimal, on average.

Fourth, high-quality interdisciplinary research, which is a precondition for ecosystem service measurement and valuation, continues to be exceedingly rare. As a consequence, there are very few studies in which all three stages of valuation (ecological changes, economic behaviors, and welfare impacts) are studied using appropriate methods. In general, the more complex the delivery of the service (e.g., hydrological or health benefits), the fewer the number of published comprehensive studies. Because many of the valuation studies are led by teams of ecologists, the final valuation or monetization exercise usually relies on some form of benefits transfer. Unfortunately, the typical transfer of benefits or values fails to satisfy basic theoretical requirements concerning income constraints and substitution effects. The proposal by Smith et al. (2006) for "structural meta-analysis" presents one approach for estimating or calibrating a value function of ecosystem services in a manner consistent with economic theory.

Finally, and most importantly, few valuation studies link a specific policy (e.g., protected area, decentralization) to changes in ecological conditions and resulting welfare impacts. For example, many studies look at flows of benefits from a protected area without considering how those flows would change in the absence of protection or with a change in management status. The absence of a policy context implies that many of the values presented in the studies reviewed (and summarized in Tables 1–4) are not clearly linked to the key policy decisions that are needed. This absence also implies that many of the forest protection policy initiatives reviewed are generally not based on information from efforts to quantify and value the very services that these initiatives aim to deliver.

2. Careful Evaluation of Programs and Policies Aimed at Delivering Ecosystem Services is Rare

Despite decades of efforts to protect ecosystems in developing nations from anthropogenic disturbances, little is known about the effectiveness of the most popular efforts (and even less is known about their cost effectiveness). Protected areas appear to have some positive environmental impacts, although their placement in less threatened (and less controversial) lands has limited their additionality. Other approaches, such as decentralization and PES, have had ambiguous impacts. This ambiguity is due partly to adverse self-selection (e.g., communities already managing their forests well are selected for devolution of management authority; unthreatened forests are volunteered for incentive payments) and noncompliance with management rules. However, these findings must remain tentative until there has been more well-designed empirical research.

To date, the majority of the well-designed empirical evaluations have been conducted by economists, but these studies have not had much influence on the scientific and policy discourse. As with valuation studies, biologists and other social scientists have tended to lead the way. For example, the economic study by Cropper et al. in 2001 of the impact of protected areas in Thailand, which uses satellite forest cover images and controls for confounding factors, has 79 citations, whereas the 2001 study by Bruner et al. that compares survey measures of human disturbance inside and immediately adjacent to protected areas in twenty-two nations, but does not control for any baseline differences outside and inside protected areas, has 481 citations (Google Scholar, November 11, 2009).

The lack of involvement and influence of economists is unfortunate because the bulk of the existing literature tends to be weak on both theory and empirical design (e.g., no clear concept of potential trade-offs; no clear identification strategies for estimating causal effects). The conservation science community could benefit greatly from the advances in empirical research that have been led by economists in other policy fields, including recent advances in addressing spatial dependence and in creating quasi-experimental and experimental study designs to measure average, marginal, and continuous causal program effects. Furthermore, unlike empiricists working in public health, labor, education, and development economics, environmental scientists and practitioners have no culture of using experimental designs to implement policies and programs in the environmental field with the intent of estimating causal impacts.

These observations concerning the state of the art in ecosystem services valuation and evaluation lead us to our third and final conclusion:

3. Future Studies Must More Tightly Integrate Policy and Research by Combining Nonmarket Valuation and Program Evaluation

We believe that decision makers would prefer to know the expected economic impact of policies and programs being implemented to supply ecosystem services rather than the citizen WTP for a hypothetical change in some poorly understood ecosystem service. Thus it is imperative that future studies combine valuation and program evaluation, so that valuation estimates are based on the observed impacts of real-world programs and policies, thereby effectively embedding valuation into evaluations. Using scarce research funds to support such integrated valuation-evaluation studies will help ensure that future research on ecosystem services yields higher quality results that are of greater use to policymakers.

In practice, such an approach essentially comprises doing impact evaluations of ecosystem conservation policies and programs on social welfare outcomes. This is not necessarily more complex than current research designs. For example, consider the context of local welfare impacts of ecosystem services. Rather than examining the current use of the ecosystem, researchers could study the poverty and social welfare impacts of government conservation policies (Andam et al. 2010) or community forest management regimes (Weber et al. 2011). An alternative strategy is some form of creative benefits transfer analysis using, for example, previous single-service studies to measure the multiple benefits of conservation areas (Kremen et al. 2000; Naidoo and Ricketts 2006). However, as discussed earlier, simply transferring the benefits from numerous single-service studies will not satisfy the basic theoretical requirements concerning income constraints and substitution effects. Thus use of this alternative strategy requires a "structural meta-analysis" approach (Smith et al. 2006).

Economists clearly have much to offer to the still emerging literature on valuation and policy effectiveness. Collaborations among economists and natural scientists to design studies that uncover the economic value of ecosystem services are already underway (e.g., The Natural Capital Project). With the rise in popularity of performance payments for ecosystem service supply and the potential for billions of dollars to soon begin flowing from developed to developing nations through REDD mechanisms, it is urgent that economists and natural scientists continue to work together to build a high-quality evidence base concerning both the values of ecosystem services and the impacts and effectiveness of policies and programs designed to deliver those services.

References

Albers, Heidi J., and Paul J. Ferraro. 2006. Economics of biodiversity conservation in developing countries. In *Economic development and environmental sustainability: New policy options*. ed. M. Toman and R. Lopez. New York: Oxford University Press.

Anaman, Kwabena A., and Norsinah Ibrahim. 2003. Statistical estimation of dose-response functions of respiratory diseases and societal costs of haze-related air pollution in Brunei Darussalem. *Pure and Applied Geophysics* 160 (1–2): 279–93.

Andam, Kwaw S., Paul J. Ferraro, Alexander Pfaff, G. Arturo Sanchez-Azofeifa, and Juan Robalino. 2008. Measuring the effectiveness of protected area networks in reducing deforestation. *Proceedings of the National Academy of Sciences* 105 (42): 16089–94.

Andam, Kwaw, Paul J. Ferraro, Katharine R. E. Sims, Andrew Healy, and Margaret Holland. 2010. Protected areas reduced poverty in Costa Rica and Thailand. *Proceedings of the National Academy of Sciences* 107 (22): 9996–10001.

Angelsen, Arild. 2008. REDD models and baselines. *International Forestry Review* 10 (3): 465–75.

Arriagada, Paul, J. Ferraro, Sills Erin, Subhrendu K. Pattanayak, and Sylvia Cordero. Forthcoming. Do payments for environmental services reduce deforestation? A farm-level evaluation from Costa Rica. *Land Economics 88*.

Ataroff, Michele, and Fermín Rada. 2000. Deforestation impact on water dynamics in a Venezuelan Andean cloud forest. *Ambio* 29 (7): 440–44.

Baland, J. M., Pranab Bardhan, Sanghamitra Das, and Dilip Mookherjee. 2010. Forests to the People: Decentralization and forest degradation in the Indian Himalayas. *World Development* 38 (11): 1642–56.

Barbier, Edward B. 2012. Progress and challenges in valuing coastal and marine ecosystem services. *Review of Environmental Economics and Policy.*

Barkmann, J., K. Glenk, A. Keil, C. Leemhuis, N. Dietrich, G. Gerold, and R. Marggraf. 2008. Confronting unfamiliarity with ecosystem functions: The case for an ecosystem service approach to environmental valuation with stated preference methods. *Ecological Economics* 65 (1): 48–62.

Bienabe, Estelle, and Robert R. Hearne. 2006. Public preferences for biodiversity conservation and scenic beauty within a framework of environmental services payments. *Forest Policy and Economics* 9 (4): 335–48.

Blackman, Allen, Adriana Chacon, Jeffrey Ferris, and Jorge Rivera. 2009. *Environmental and socioeconomic impacts of 'sustainable' certification: The empirical evidence base*. Washington, DC: Scientific and Technical Advisory Panel, Global Environment Facility.

Bockstael, Nancy E., A. Myrick Freeman III, Raymond J. Kopp, Paul R. Portney, and V. Kerry Smith. 2000. On measuring economic values for nature. *Environmental Science and Technology* 34 (8): 1384–89.

Bowler, Diana, L. Buyung-Ali, J. Healey, Julia P. G. Jones, Teri Knight, and Andrew Pullin. 2009. *The evidence base for community forest management as a mechanism for supplying global environmental benefits and improving local welfare*. Systematic Review No. 48, Collaboration for Environmental Evidence.

Boyd, James, and Spencer Banzhaf. 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics* 63 (2–3): 616–26.

Bruner, Aaron G., Raymond E. Gullison, Richard E. Rice, and Gustavo A. B. da Fonseca. 2001. Effectiveness of parks in protecting tropical biodiversity. *Science* 291 (5501): 125–28.

Campbell, B. M., M. Luckert, and I. Scoones. 1997. Local-level valuation of savanna resources: a case study from Zimbabwe. *Economic Botany* 51 (1): 59–77.

Cavendish, William A. 2000. Empirical regularities in the poverty-environment relationship of rural households: evidence from Zimbabwe. *World Development* 28 (11): 1979–2003.

Chase, Lisa C., David R. Lee, William D. Schulze, and Deborah J. Anderson. 1998. Ecotourism demand and differential pricing of national park access in Costa Rica. *Land Economics* 74 (4): 466–82.

Chhatre, Ashwini, and Arun Agrawal. 2008. Forest commons and local enforcement. *Proceedings of the National Academy of Sciences* 105 (36): 13286–91.

———. 2009. Trade-offs and synergies between carbon storage and livelihood benefits from forest commons. *Proceedings of the National Academy of Sciences* 106 (42): 17667–70.

Chomitz, Kenneth M., and Kanta Kumari. 1998. The domestic benefits of tropical forests: a critical review. *The World Bank Research Observer* 13 (1): 13–35.

Coad, Lauren, Alison L. Campbell, Lera Miles, and Katherine Humphries. 2008. *The costs and benefits of protected areas for local livelihoods: A review of the current literature*. Cambridge, UK: United Nations Environment Programme, World Conservation Monitoring Centre.

Colfer, Carol J. Pierce, Douglas Sheil, and Misa Kishi. 2006. *Forests and human health: assessing the evidence*. Center for International Forestry Research (CIFOR) Occasional Paper No. 45. Bogor, Indonesia: CIFOR.

Convery, Frank J., and Luke Redmond. 2007. Market and price developments in the European Union emissions trading scheme. *Review of Environmental Economics and Policy* 1 (1): 88–111.

Cropper, Maureen, Jyotsna Puri, and Charles Griffiths. 2001. Predicting the location of deforestation: The role of roads and protected areas in North Thailand. *Land Economics* 77 (2): 172.

Das, Saudamini, and Jeffrey R. Vincent. 2009. Mangroves protected villages and reduced death toll during Indian super cyclone. *Proceedings of the National Academy of Sciences* 106 (18): 7357–60.

Echeverría, Jaime, Michael Hanrahan, and Raúl Solórzano. 1995. Valuation of non-priced amenities provided by the biological resources within the Monteverde Cloud Forest Preserve, Costa Rica. *Ecological Economics* 13 (1): 43–52.

Edmonds, Eric E. V. 2002. Government-initiated community resource management and local resource extraction from Nepal's forests. *Journal of Development Economics* 68 (1): 89–115.

Ellingson, Lindsey, and Andrew Seidl. 2007. Comparative analysis of non-market valuation techniques for the Eduardo Avaroa Reserve, Bolivia. *Ecological Economics* 60 (3): 517–25.

Ferraro, Paul J. 2008. Asymmetric information and contract design for payments for environmental services. *Ecological Economics* 65 (4): 810–821.

Ferraro, Paul J., and R. David Simpson. 2002. The cost-effectiveness of conservation performance payments. *Land Economics* 78 (3): 339–53.

Fisher, Brendan, and R. Kerry Turner. 2008. Letter to the editor: Ecosystem services: classification for valuation. *Biological Conservation* 141 (5): 1167–69.

Frankenberg, Elizabeth, Douglas McKee, and Duncan Thomas. 2005. Health consequences of forest fires in Indonesia. *Demography* 42 (1): 109–29.

Freeman, A. Myrick. 1993. *The measurement of environmental and resource Values: Theory and methods*. Washington, DC: Resources for the Future.

Gibbs, Holly K., Sandra Brown, John O. Niles, and Jonathan A. Foley. 2007. Monitoring and estimating tropical forest carbon stocks: making REDD a reality. *Environmental Research Letters* 2 (4): 045023.

Godoy, R., H. Overman, J. Demmer, L. Apaza, E. Byron, T. Huanca, W. Leonard, E. Perez, V. Reyes-Garcia, V. Vadez, D. Wilkie, A. Cubas, K. McSweeney, and N. Brokaw. 2002. Local financial benefits of rain forests: comparative evidence from Amerindian societies in Bolivia and Honduras. *Ecological Economics* 40 (3): 397–409.

Godoy, Ricardo, David Wilkie, Han Overman, Adoni Cubas, Glenda Cubas, Josefien Demmer, Kendra McSweeney, and Nicholas Brokaw. 2000. Valuation of consumption and sale of forest goods from a Central American rain forest. *Nature* 406 (6791): 62–63.

Gram, Soren. 2001. Economic valuation of special forest products: an assessment of methodological shortcomings. *Ecological Economics* 36 (1): 109–17.

Guo, Zhongwei, Yiming Li, Xiangming Xiao, Lin Zhang, and Yaling Gan. 2007. Hydroelectricity production and forest conservation in watersheds. *Ecological Applications* 17 (6): 1557–62.

Heal, Geoffrey. 2000. *Nature and the marketplace: Capturing the value of ecosystem services.* Washington, DC: Island Press.

Heubach, Katja, Rudiger Wittig, Ernst-August Nuppenau, and Karen Hahn. 2011. The economic importance of non-timber forest products (NTFPs) for livelihood maintenance of rural west African communities: A case study from northern Benin. *Ecological Economics* 70 (11): 1991–2001.

Holmes, Thomas, Keith Alger, Chris Zinkhan, and Evan Mercer. 1998. The effect of response time on conjoint analysis estimates of rainforest protection values. *Journal of Forest Economics* 4 (1): 7–28.

Jayachandran, S. 2009. Air quality and early-life mortality: Evidence from Indonesia's wildfires. *Journal of Human Resources* 44 (4): 916–54.

Joppa, Lucas, and Alexander Pfaff. 2009. High and Far: Biases in the Location of Protected Areas. *PLoS ONE* 4 (12): e8273.

———. 2010. Re-assessing the forest impacts of protection: The challenge of non-random location & a corrective method. *Annual Review of Ecological Economics* 1185: 135–49.

———. 2011. Global protected area impacts. *Proceedings of the Royal Society B* 278 (1712): 1633–38.

Jumbe, Charles B. L, and Arild Angelsen. 2006. Do the poor benefit from devolution policies? Evidence from Malawi's forest co-management program. *Land Economics* 82 (4): 562–81.

Klemick, Heather. 2011. Shifting cultivation, forest fallow and externalities in ecosystem services: Evidence from the Eastern Amazon. *Journal of Environmental Economics and Management* 61 (1): 95–106.

Kremen, C., J. Niles, M. Dalton, G. Daily, P. Ehrlich, P. Fay, D. Grewal, and R. P. Guillery. 2000. Economic incentives for rain forest conservation across scales. *Science* 288 (5472): 1828–32.

Lele, Sharachchandra, Iswar Patil, Shrinivas Badiger, Ajit Menon, and Rajeev Kumar. *The economic impact of forest hydrological services on local communities: A case study from the western ghats of India.* SANDEE Working Paper 36–08.

Lund, Jens F., Kulbhushan Balooni, and Thorkil Casse. 2009. Change we can believe in? Reviewing studies on the conservation impact of popular participation in forest management. *Conservation and Society* 7 (2): 1–13.

Menkhaus, Susan, and Douglas J. Lober. 1996. International ecotourism and the valuation of tropical rainforests in Costa Rica. *Journal of Environmental Management* 47 (1): 1–10.

Millennium Ecosystem Assessment. 2005. *Ecosystems and human well-being: synthesis.* Washington, DC: Island Press.

Naidoo, Robin, and Wictor L. Adamowicz. 2005. Biodiversity and nature-based tourism at forest reserves in Uganda. *Environment and Development Economics* 10 (2): 159–78.

Naidoo, Robin, Trent Malcolm, and Adam Tomasek. 2009. Economic benefits of standing forests in highland areas of Borneo: quantification and policy impacts. *Conservation Letters* 2 (1): 35–44.

Naidoo, Robin, and Taylor H. Ricketts. 2006. Mapping the economic costs and benefits of conservation. *PLoS Biology* 4 (11): 2153–63.

Nelson, Andrew, and Kenneth Chomitz. 2011. Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: A global analysis using matching methods. *PLoS ONE* 6 (8): e22722.

Núñez, Daisy, Laura Nahuelhual, and Carlos Oyarzún. 2006. Forests and water: The value of native temperate forests in supplying water for human consumption. *Ecological Economics* 58 (3): 606–16.

Ostrom, Elinor. 1990. *Governing the commons: The evolution of institutions for collective action.* New York: Cambridge University Press.

Pattanayak, S. K., M. T. Ross, B. M. Depro, S. C. Bauch, C. Timmins, K. Jones, and K. Alger. 2009. Climate change and conservation in Brazil: CGE evaluation of health and wealth impacts. *B.E. Journal of Economic Analysis & Policy* 9 (2): Article 6.

Pattanayak, S. K., S. Saha, P. Sahu, E. O. Sills, A. Singha, and J.-C. Yang. 2010. Mine over matter: Health, wealth and forests in a mining area of Orissa. *Indian Growth and Development Review* 3 (2): 166–85.

Pattanayak, S. K., S. Wunder, and P. J. Ferraro. 2010. Show me the money: Do payments supply ecosystem services in developing countries? *Review of Environmental Economics and Policy* 4 (2): 254–274.

Pattanayak, Subhrendu K., Katherine Dickinson, Catherine Corey, Erin O. Sills, Brian C. Murray, and Randall Kramer. 2006. "Deforestation, malaria, and poverty: A call for transdisciplinary research to design cross-sectoral policies". *Sustainability: Science, Practice and Policy* 2 (2): 1–12.

Pattanayak, Subhrendu K. 2004. Valuing watershed services: concepts and empirics from southeast Asia. *Agriculture, Ecosystems & Environment* 104 (1): 171–84.

Pattanayak, Subhrendu K., and Randall A. Kramer. 2001a. Worth of watersheds: a producer surplus approach for valuing drought mitigation in Eastern Indonesia. *Environment and Development Economics* 6 (1): 123–46.

———. 2001b. Pricing ecological services: Willingness to pay for drought mitigation from watershed protection in eastern Indonesia. *Water Resources Research* 37 (3): 771–78.

Pattanayak, Subhrendu K., and E. Mercer. 1998. Valuing soil conservation benefits of agroforestry practices: Contour hedgerows in the eastern Visayas, Philippines. *Agricultural Economics* 18 (1): 31–46.

Pattanayak, Subhrendu K., and Erin O. Sills. 2001. Do tropical forests provide natural insurance? The microeconomics of non-timber forest product collection in the Brazilian Amazon. *Land Economics* 77 (4): 595–612.

Pattanayak, Subhrendu K., Erin O. Sills, and Randall A. Kramer. 2004. Seeing the forest for the fuel. *Environment and Development Economics* 9 (2): 155–79.

Pattanayak, Subhrendu K., and Kelly J. Wendland. 2007. Nature's care: diarrhea, watershed protection, and biodiversity conservation in Flores, Indonesia. *Biodiversity Conservation* 16 (10): 2801–19.

Pattanayak, Subhrendu K., and Junko Yasuoka. 2008. Deforestation and malaria: Revisiting the human ecology perspective. In *Human health and forests: A global overview of issues, practice, and policy*. ed. Carol J. Pierce Colfer, 197–220. Sterling, VA: Earthscan.

Patz, Jonathan A., Diarmid Campbell-Lendrum, Tracey Holloway, and Jonathan A. Foley. 2005. Impact of regional climate change on human health. *Nature* 438 (17): 310–17.

Pearce, David W. 2001. The economic value of forest ecosystems. *Ecosystem Health* 7 (4): 284–96.

Persha, L., A. Agrawal, and A. Chhatre. 2011. Social and ecological synergy: Local rulemaking, forest livelihoods, and biodiversity conservation. *Science* 331 (6024): 1606–8.

Peters, Charles M., Alwyn H. Gentry, and Robert O. Mendelsohn. 1989. Valuation of an Amazonian rainforest. *Nature* 339 (6227): 655–56.

Pfaff, A., E. O. Sills, G. S. Amacher, M. J. Coren, K. Lawlor, and C. Streck. 2010. Policy impacts on deforestation: Lessons learned from past experiences to inform new initiatives. Nicholas Institute Report 10-02. Durham, NC.

Plummer, Mark L. 2009. Assessing benefit transfer for the valuation of ecosystem services. *Frontiers in Ecology and the Environment* 7 (1): 38–45.

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.

Ponsgiri, Montira J., Joe Roman, Vanessa O. Ezenwa, Tony L. Goldberg, Hillel S. Koren, Stephan C. Newbold, Richard S. Ostfeld, Subhrendu K. Pattanayak, and Daniel J. Salkeld. 2009. Biodiversity loss affects global disease ecology. *Bioscience* 59 (11): 945–54.

Priess, J. A., M. Mimler, A. M Klein, S. Schwarze, T. Tscharntke, and I. Steffan-Dewenter. 2007. Linking deforestation scenarios to pollination services and economic returns in coffee agroforestry systems. *Ecological Applications* 17 (2): 407–17.

Ribot, Jesse. 2002. *Democratic decentralization of natural resources: Institutionalizing popular participation*. Washington, DC: World Resources Institute.

Ricketts, Taylor H., Gretchen C. Daily, Paul R. Ehrlich, and Charles D. Michener. 2004. Economic value of tropical forest to coffee production. *Proceedings of the National Academy of Sciences of the United States of America* 101 (34): 12579.

Ricketts, Taylor H., James Regetz, Steffan-Dewenter Ingolf, Saul A. Cunningham, Claire Kremen, Anne Bogdanski, Barbara Gemmill-Herren, Sarah S. Greenleaf, Alexandra M. Klein, Margaret M. Mayfield, Lora A. Morandin, Alfred Ochieng, Simon G. Potts, and Blande F. Viana. 2008. Landscape effects on crop pollination services: are there general patterns? *Ecology Letters* 11 (5): 499–515.

Saha, S., S. K. Pattanayak, E. O. Sills, and A. Singha. 2011. Under-mining health: Environmental justice and mining in India. *Health & Place* 17 (1): 140–48.

Sanglimsuwam, K., E. O. Sills, S. K. Pattanayak, S. Saha, A. Singha, and B. Sahoo. Forthcoming.

Occupational and environmental health impacts from mining in Orissa, India. In *Environment and Development Economics: Essays in Honor of Sir Partha Dasgupta*. ed. S. Barrett. Oxford University Press.

Sastry, Narayan. 2002. Forest fires, air pollution, and mortality in Southeast Asia. *Demography* 39 (1): 1–23.

Shackleton, S. E., C. M. Shackleton, T. R. Netshiluvhi, B. S. Geach, A. Balance, and D. H. K. Fairbanks. 2002. Use patterns and value of savanna resources in three rural villages in South Africa. *Economic Botany* 56 (2): 130–46.

Sheil, Douglas, and Sven Wunder. 2002. The value of tropical forest to local communities: complications, caveats, and cautions. *Conservation Ecology* 6 (2): Article 9.

Shone, Bryan M., and Jill L. Caviglia-Harris. 2005. Quantifying and comparing the value of non-timber forest products in the Amazon. *Ecological Economics* 58 (2): 249–67.

Shyamsundar, Priya, and Randall A. Kramer. 1996. Tropical forest protection: An empirical analysis of the costs borne by local people. *Journal of Environmental Economics and Management* 31 (2): 129–44.

Sims, Katharine. 2010. Conservation and development: Evidence from Thai protected areas. *Journal of Environmental Economics and Management* 60 (2): 94–114.

Smith, V. Kerry, Subhrendu K. Pattanayak, and George L. Van Houtven. 2006. Structural benefits transfer: An example using VSL estimation. *Ecological Economics* 60 (2): 361–71.

Somanathan, E., R. Prabhakar, and Bhupendra Singh Mehta. 2009. Decentralization for cost-effective conservation. *Proceedings of the National Academy of Sciences* 106 (11): 4143–47.

Sunderlin, William W. D, Jeffrey J. Hatcher, and Megan M. Liddle. 2008. *From exclusion to ownership? Challenges and opportunities in advancing forest tenure reform*. Washington, DC: Rights and Resources Initiative.

Tobias, Dave, and Robert Mendelsohn. 1991. Valuing ecotourism in a tropical rain-forest reserve. *Ambio* 20 (2): 91–93.

Van der Werf, G. R., D. C. Morton, R. S. DeFries, J. G. J. Olivier, P. S. Kasibhatla, R. B. Jackson, G. J. Collatz, and J. T. Rnaderson. 2009. CO_2 emissions from forest loss. *Nature Geoscience* 2 (12): 737–38.

Veloz, Alberto, Douglas Southgate, Fred Hitzhusen, and Robert Macgregor. 1985. The economics of erosion control in a subtropical watershed: A Dominican case. *Land Economics* 61 (2): 145–55.

Vittor, Amy Yomiko, Robert H. Gilman, James Tielsch, Gregory Glass, Tim Shields, Wagner Sanchez Lozano, Viviana Pinedo-Cancino, and Jonathan A. Patz. 2006. The effect of deforestation on the human-biting rate of *Anopheles Darlingi*, the primary vector of falciparum malaria in the Peruvian Amazon. *American Journal of Tropical Medicine and Hygiene* 74 (1): 3–11.

Wang, Xuehong, Jeff Bennett, Chen Xie, Zhitao Zhang, and Dan Liang. 2007. Estimating non-market environmental benefits of the Conversion of Cropland to Forest and Grassland Program: A choice modeling approach. *Ecological Economics* 63 (1): 114–25.

Weber, J., E. Sills, S. C. Bauch, and S. K. Pattanayak. 2011. Do ICDPs work? An empirical evaluation of forest-based microenterprises in the Brazilian Amazon. *Land Economics* 87 (4): 645–81.

Wilkie, David S., Gilda A. Morelli, Josefien Demmer, Malcolm Starkey, Paul Telfer, and Matthew Steil. 2006. Parks and people: Assessing the human welfare effects of establishing protected areas for biodiversity conservation. *Conservation Biology* 20 (1): 247–49.

Williams, Michael R., Thomas R. Fisher, and John M. Melack. 1997. Solute dynamics in soil water and groundwater in a central Amazon catchment undergoing deforestation. *Biogeochemistry* 38 (3): 303–35.

World Database on Protected Areas (WDPA). 2009. http://www.wdpa.org/.

Wunder, Sven. 2008. Payments for environmental services and the poor: concepts and preliminary evidence. *Environment and Development Economics* 13 (3): 279–97.