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Ecosystem Services and Green Growth

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Abstract

"Ecosystem services" has become a catch-phrase for the complex connections between the natural environment and human well-being. This paper considers the impact of changes in the supply of ecosystem services, and programs to increase their supply, on near-term growth of gross domestic product. It focuses on the relationship between locally generated versus transboundary services and growth in developing countries, where the highest rates of ecosystem degradation tend to be found. There is a common perception that there is a tradeoff between environmental protection and economic growth, especially in the near term. This perception can make policymakers reluctant to support environmental protection. Where the environment is a source of economically important services, then environmental protection may stimulate growth of gross domestic product instead of reducing it.

The paper considers evidence on the economic value of regulating services; the degree to which ecosystems actually supply some of the services they are commonly assumed to supply; and the near-term growth implications of restoring ecosystems, and reducing their loss. This leads to a discussion on the effectiveness of programs intended to reduce ecosystem loss, with a focus on protected areas and payments for ecosystem services, and the effects of these programs on poverty alleviation.

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1 Introduction

Like "sustainable development," "ecosystem services" has become a catch-phrase for the complex connections between the natural environment and human well-being. The Millennium Ecosystem Assessment (2005) argues that ecosystems supply services that contribute to human well-being in four ways. First and most obviously, they provide tangible goods, such as food, fiber, and fuel, which are processed by firms and consumed by households (*provisioning services*). Second, they enhance production processes in various ways, including by pollinating crops, purifying water, and stabilizing climate (*regulating services*). Third, they furnish amenities, like the beauty of national parks, which humans value for aesthetic, ethical, or spiritual reasons (*cultural services*). Finally, they support all these services through nutrient cycling, soil formation, and other ecological processes (*supporting services*).

Prior research has examined a variety of economic issues related to ecosystem services (TEEB 2010). This paper considers one that has evidently been neglected, although the existing literature contains much useful information for investigating it: the effect of changes in the supply of ecosystem services, and programs to increase their supply, on near-term economic growth. By "economic growth," we mean a conventional monetary measure of the change in economic output, such as GDP. By "near-term," we mean the usual macroeconomic definition of a few years to no more than a decade or so (Cavallo and Noy 2010, p. 8).

The paper examines this issue without implying that near-term economic growth should replace a broader measure of social welfare, such as intergenerational well-being, as the preferred metric for evaluating policy interventions. Instead, the paper examines this issue because of the common perception that there is a tradeoff between environmental protection and economic growth, especially in the near term. This perception can make policymakers reluctant to support environmental protection. If the environment is a source of economically important services, however, then perhaps environmental protection can stimulate growth instead of reducing it and thus avoid, or at reduce, an environment-growth tradeoff.

The paper focuses on regulating services, given their connection to production processes. It focuses further on regulating services that primarily benefit the country where the ecosystems supplying them are located, so that any growth stimulus from changes in ecosystem management practices are reaped by the country making those changes. Hence, the paper largely ignores transboundary services, such as climate stabilization. Unless otherwise noted, the word "services" refers to this particular category of ecosystem services throughout the rest of the paper. The paper is primarily concerned with the relationship between these services and growth in

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developing countries, where the highest rates of ecosystem degradation tend to be found (e.g., deforestation).

The sustainable development literature has led to the development of improved methods for incorporating the environment into national accounts (World Bank 2010), but this paper is not about "green GDP." The reason is simple: given that the value of a regulating service derives from its use in production, GDP already reflects its current contribution to production (Mäler 1991). For example, if deforestation reduces natural pollination services that are important inputs for some crops, then the portion of GDP associated with those crops will automatically fall as deforestation reduces the supply of the service. Better accounting of regulating services could, however, reveal that value added as conventionally measured might not reflect the actual relative contributions of different sectors to GDP (Vincent 1999). In the example just given, better accounting would reveal that part of the operating surplus of agriculture is actually the value of an unpaid service provided by forests. Consequently, conventional income accounts understate true value added in the forest sector and overstate it in the agricultural sector, and sectoral GDP accounts could be adjusted by reallocating the corresponding portion of operating surplus from the latter to the former. Because this is just a reallocation, overall GDP would not change.

A bit of reflection on this point leads to the identification of four conditions that must hold for ecosystem services to provide a near-term stimulus to economic growth:

1. A conservation program must either restore a degraded ecosystem or reduce the loss of an intact one.

2. This positive ecosystem change must increase the supply of an ecosystem service.

3. The increased supply must occur within the near term.

4. The service must be economically valuable.

This paper considers these four conditions in roughly reverse order. It begins by reviewing evidence on the economic value of regulating services (condition 4). This leads into a discussion of whether ecosystems actually supply some of the services they are commonly assumed to supply (consideration 2). The paper then considers the near-term growth implications of the two basic ways of changing the supply of regulating services: restoring ecosystems, and reducing their loss (condition 3). This leads into a discussion on the effectiveness of programs intended to reduce ecosystem loss, with a focus on protected areas and payments for ecosystem services (condition 1). The penultimate section of the paper examines the effects of these programs on poverty alleviation, which is a primary development objective that is difficult to attain without economic growth. The final section summarizes the main findings of the paper and comments on their implications for future research.

2 The value of regulating services

2.1 Cross-country estimates of service values for forests

All else being equal, regulating services are more likely to provide a significant boost to economic growth if they have a large impact on production. Methods for valuing ecosystems as production inputs are well-established (Freeman 2003, McConnell and Bockstael 2005, Vincent 2011). More valuation studies have been conducted on forests than on any other ecosystem. They suggest that, on average, regulating services comprise a small portion of the annual value of goods and services from forests.

Lampietti and Dixon (1995) conducted an early review of nearly 50 forest valuation studies from about 20 developing and developed countries. Watershed services were the only regulating service included in their estimates, and they accounted for about 5% of forest value. Provisioning services, mainly timber and fuelwood, accounted for about 60-80%. An updated review by Pearce (2001) added information on the value of carbon sequestration, which it found to be large, but otherwise it reached similar conclusions: on average, regulating services besides carbon sequestration are small both in absolute terms and relative to provisioning services, especially timber. A study by Croitoru (2007) had a more limited geographical scope, just 18 countries in the Mediterranean region, but it incorporated estimates of other regulating services besides watershed protection and carbon sequestration. Despite the wider coverage, these services collectively accounted for only about 15% of forest value, compared to about half for provisioning services.

Conventional estimates of forest sector GDP include only the value of wood harvests, which was on the order of $100 billion, or about 0.2% of global GDP, during 2003-7 (FAO 2010, p. 8). According to the studies just reviewed, wood harvests account for about a third to a half of total forest value on average. If this share holds globally—a big "if," obviously—then the value of all forest ecosystem services, not just regulating services, is well below 1% of global GDP. This is a far cry from the global value of forest ecosystem services reported in the heavily cited study by Costanza et al. (1997), approximately 25% of GDP. This large discrepancy supports previous arguments by economists that Costanza et al. dramatically overstated ecosystem service values (e.g., Bockstael et al. 2000).

Croitoru drew attention to two reasons why existing estimates of the value of forest ecosystem services are best interpreted as lower bounds, however. First, the estimates are partial: they exclude the values of some services. Despite increased valuation effort, information on the value of ecosystem services remains far from complete. A recent review sponsored by UNEP on valuation issues related to biodiversity and ecosystem services, The Economics of Ecosystems and Biodiversity (TEEB), stated that "Some … ecosystem benefits, especially regulating services … have only recently begun to be assigned an economic value" (TEEB 2010, p. 8). An even more recent review on coastal ecosystems, which include mangroves, saltmarshes, seagrass beds, and coral reefs, reached a similar conclusion: "Although reliable valuation estimates are beginning to emerge for the key services of some ECEs [estuarine and coastal ecosystems], such as coral reefs, salt marshes, and mangroves, many of the important benefits of seagrass beds and sand dunes and beaches have not been assessed properly. Even for coral reefs, marshes, and mangroves, important ecological services have yet to be valued reliably" (Barbier et al. 2011, p. 169). Estimates of the mean value of regulating services can be expected to rise as valuation studies encompass more services, but it seems unlikely that they will amount to more than a few percentage points of GDP, at most, for most countries.

Second, some services are being supplied at suboptimal levels because historical management decisions have ignored them. Service values depend on the way ecosystems are managed. For example, water quality in a river flowing out of a forest will be reduced if forest management decisions focus only on timber and ignore the effects of logging on soil erosion and siltation. As this example indicates, ecosystem services are not necessarily complements (Jackson et al. 2005; Kinzig et al. 2011. Actions that increase the supply of one (in this case, a provisioning service, timber) can decrease the supply of another (in this case, a regulating service, water purification).

In this situation, optimal management decisions typically must be made at a landscape level, not at the level of individual properties, with different parts of the landscape being managed in different ways. Said another way, the aggregate value of ecosystem services from a landscape will be higher if the supply of particular services is concentrated more in some locations than in others, instead of combining ("stacking") all services in all locations. This is obvious when ecosystem characteristics vary substantially within the landscape: for example, the combined value of timber and water quality will likely be maximized if logging is allowed only in forests on gentler slopes, with forests on steeper slopes being protected in order to safeguard water quality.

What is less obvious is that the incompatibility of services can result in this kind of spatially specialized management regime being optimal even when ecosystems are homogeneous. In technical terms, this happens when the ecosystem production set is nonconvex, which causes a diseconomy of scope (Bowes and Krutilla 1989, Vincent and Binkley 1993). For example, managing part of a given type of forest intensively for timber, while managing the remainder intensively for biodiversity by prohibiting timber harvesting in it, could yield a higher aggregate value than managing all parts of the forest for both timber production and biodiversity conservation (Potts and Vincent 2008a,b). For forests at least, there is abundant evidence that production sets are nonconvex (Calish et al. 1978, Bowes and Krutilla 1989, Swallow et al. 1990, Boscolo and Vincent 2003). This complicates the interpretation of valuation studies: values are contingent on management regimes, yet optimal management regimes can vary spatially even within a given type of ecosystem. Hence, a lower value in some parts of an ecosystem is not necessarily a sign that those parts are being managed suboptimally.

2.2 Quality of estimates

Two research priorities are therefore to expand the scope of valuation studies on ecosystem services, and to conduct studies that go beyond the estimation of single numbers ("the value is …") and provide information on the impact of management decisions on values at different scales. An even higher priority is to upgrade the quality of valuation studies. Ferraro et al. (in press) reviewed valuation studies on four regulating services of forests: watershed services, pollination, carbon sequestration, and human health. They identified a large number of studies, but they judged that few valued ecosystem services in a rigorous manner.

One problem is a lack of clarity about the change being valued: for example, service losses if forests are converted to other land uses (i.e., deforestation), vs. losses if they are degraded but not converted (e.g., logging). This raises doubts about the validity of the estimates, as the theory of environmental valuation is based on comparison of utility levels in the presence and absence of well-defined environmental changes (Bockstael and Freeman 2005).

Another common problem is inadequate control for factors that confound estimates of the physical supply or value of services. This can lead to exaggerated estimates of the importance of services. This is a pervasive and underappreciated problem, so in the remainder of this section we consider three forest-related examples in detail: flood mitigation, pollination, and coastal protection.

2.2.1 Flood mitigation by forests

The belief that forests reduce floods was a driving force behind laws that established national forests in United States more than a century ago (Williams 2003). Similarly, Malaysia's 1978 national forest policy called for the creation of "protective forests," "in order to ensure … the sound climatic and physical condition of the country, the safe-guarding of water supplies, soil fertility and environmental quality and the minimization of damage by floods and erosion to rivers and agricultural land" (Vincent and Rozali 2005).

Despite this long history, the first cross-country statistical analysis on forests and floods was published only recently. Bradshaw et al. (2007) compiled panel data on flood events in 56 developing countries during 1990-2000. They found that natural forest area had a significant, negative effect on flood frequency. Their regression models included several controls, but all were environmental variables, such as rainfall and slope. van Dijk et al. (2009) argued that the effect of forests in these models was confounded by the exclusion of population density, which is negatively correlated with forests and positively correlated with flood reporting, because floods are more likely to be reported if they affect human populations.

Reanalyzing the Bradshaw et al. data, van Dijk et al. found that population density explained 83% of the variation in reported flood frequency, with forest cover explaining less than 1% of the remaining 17%. Although reduced flooding might boost value-added in agriculture and other flood-sensitive sectors, it is thus not clear that forests actually supply a flood-reduction service. The forest-flood link has also been questioned by previous studies (Bruijnzeel 2004, FAO and CIFOR 2005).

2.2.2 Natural pollination services

Dozens of studies have investigated the effect of distance from natural habitat on pollination of commercial crops. Many of these studies simply examine the relationship between crop yield and distance and assume that a negative relationship, if detected, is due to reduced pollination. Typical of this approach is a study on coffee at 24 sites in Indonesia, which found that both fruit set and berry weight declined significantly with distance from remaining natural forests (Olschewski et al. 2006).

In the most comprehensive review to date, Ricketts et al. (2008) reviewed 23 pollination studies on 16 crops in 10 countries. They found that 12 studies provided information on the relationship between fruit set and distance from natural habitat, with 8 using this uncontrolled approach to infer the presence of a pollination service. They pointed out that factors besides declining pollination services, such as soil quality or moisture, could explain observed declines of fruit set with distance. Newly cleared land might be more fertile, with land farther from the forest boundary being less productive due to reduced fertility, not reduced pollination.

Ricketts et al. also conducted a meta-analysis of results from the studies, and they found that the relationship between fruit set and distance was not statistically significant, even though the number and diversity of pollinators did indeed decline with distance. This suggests that pollinators were not scarce, which in turn implies that the marginal value of natural pollination services was zero.[2] Ricketts et al. cautioned, however, that the lack of significance could also be due to low power caused by the small number of observations in the meta-analysis.

The role of natural habitat in supplying economically important pollination services is therefore not clear. Even if this service is present, results in the literature indicate that it does not necessarily justify protecting habitat. Ricketts et al. (2004) found that the value of pollination services provided by forests to coffee was comparable to the value of alternative land uses to which forests could be converted in Costa Rica, but Olschewski et al. (2006) found that alternative uses of forestland were more valuable in Indonesia and Ecuador.

Winfree et al. (2011) highlight an additional problem, and it suggests that published estimates of the value of pollination services tend to be overstated. They report that most studies have valued pollination services either by calculating the gross revenue that is lost if pollination declines or by calculating the cost of restoring production by replacing natural pollination with commercial bee hives. These methods are known to exaggerate economic losses, as they do not account adequately for adjustments that farmers make in response to negative environmental shocks (McConnell and Bockstael 2005). Winfree et al. find that the degree of exaggeration is very large: for watermelons in New Jersey and Pennsylvania, the estimated value of pollination services falls by more than 50% when such adjustments are accounted for.

2.2.3 Coastal protection

The third example, coastal protection, illustrates that controlling for potentially confounding factors does not necessarily undermine evidence of significant service values. The role of mangrove forests in protecting against coastal disasters has been debated since at least the devastating 1970 cyclone that struck then-East Pakistan (Chapman 1971, Fosberg 1971). It attracted renewed attention after the 2004 Indian Ocean tsunami. One post-tsunami study,

[2] The literature on bioprospecting values—genetic resources in tropical forests as a source of leads for new pharmaceuticals—provides a second example of the very richness of nature reducing the value of an ecosystem service. The large number of species present in biodiversity "hotspots" reduces the scarcity of leads and results in a low bioprospecting value for forestland: a mean net present value of just $14 per hectare (Costello and Ward 2006).

published in *Science*, concluded that mangroves had protected coastal Indian villages on the basis of evidence from just five villages; it found that three villages located behind mangroves had suffered less damage than two villages located directly on the coast (Danielsen et al. 2008, p. 643). This comparison is flawed by the confounding of the presence or absence of mangroves with distance from coast: all villages with mangroves were farther from the coast, while all those without mangroves were closer to it. The tiny sample precluded the use of regression methods to control for any other differences between the villages.

A more recent study on damage from the 2004 tsunami in Indonesia provides stronger evidence of protection by mangroves (Laso Bayas et al. 2011). It analyzed a larger sample of locations—180 transects along over 100 km of coastline—and found that a significant effect of mangroves remained after controlling for various environmental factors, including distance from coast. It did not control for socioeconomic factors, however. A study on another type of coastal disaster, tropical storms, controlled for both the environmental and socioeconomic characteristics of a large sample of Indian villages (409; Das and Vincent 2009). It found that mangroves significantly reduced the death toll of a 1999 storm and that the value of this protective service exceeded the value of converting remaining mangroves to agriculture. An analysis of more highly aggregated data from Thailand has also reported statistically significant evidence of coastal ecosystems protecting against coastal disasters (Barbier 2007).

As these three examples illustrate, the information base for valuing regulating services is both thin and, in some important cases, of dubious reliability. The more reliable studies are mainly ones on small, subnational regions, such as a sample of villages or farms in a particular state. This poses a significant obstacle for aggregating results to the national level (Barbier 2011).

2.3 Spatial heterogeneity

Aggregation to the national level is also impeded by substantial spatial heterogeneity in service values, which was alluded to earlier and is an active area of research (Kareiva et al. 2011). Estimates of mean values, such as those cited for forests at the start of this section, conceal substantial variation across locations, a fact that the authors of the cited studies acknowledged. Even coarse aggregates, such as groups of countries, can show large variation: the estimated mean value of ecosystem services from forests varies by a factor of more than three between northern and eastern Mediterranean countries (Croitoru 2007).

The variation in services at a smaller scale is even more striking. A study of 37 watersheds on the island of Flores in Indonesia found that increased forest area was associated with higher dry-season flows in about half of the watersheds but lower flows in the other half (Pattanayak and Kramer 2001). Hence, forests provided a positive drought-mitigation service in some locations but a negative one in others,[3] and this dramatic difference occurred within a small area, just 50 km from east to west and 20 km from north to south.

[3] The effect can go in either direction because it depends on the relative importance of two opposing processes, both of which forests tend to enhance: infiltration of precipitation into the soil, which tends to raise dry-season flows, and evapotranspiration, which tends to reduce them (Bruijnzeel 2004).

2.4 Postscript: Forests and water quality

The lack of consistent support for the common belief that forests reduce floods and droughts does not mean that forests do not provide any important watershed services. There is little dispute that conversion of forests to agricultural, industrial, urban, or residential uses reduces the quality of water flowing out of an area (Brauman et al. 2007). Soil erosion rates typically rise, causing water to contain more sediment (Bruijnzeel 2004). This can create several adverse economic impacts downstream: increased costs of dredging harbors, increased costs of water treatment, reduced reservoir storage capacity, and decreased fish catch and recreation values.

The most famous example of a watershed service is probably water purification by the Catskill Mountains, which reportedly enabled New York City to avoid the high cost of a new filtration plant (Chichilnisky and Heal 1998).[4] As yet, it is not clear if this is the first of many cases of highly valuable water-quality related services waiting to be documented, or the exception that proves the rule.

3 Restoring ecosystem services vs. reducing their loss

The supply of ecosystem services can be increased relative to a status quo scenario in two ways: restoring a degraded ecosystem and the services it formerly provided, or reducing the degradation of an intact ecosystem and the services it is currently providing. Assuming the services are economically valuable, actions of both types can boost economic output relative to its level in the absence of restorative or protective action. A positive effect on *near-term* growth depends on the ecosystem changes —restoration or degradation—affecting the supply of services quickly. If the supply of services rises or falls only gradually, then beneficial effects on growth will occur mostly in the long term. Actions to restore or protect ecosystems might still be justified, but not because they boost near-term growth. This section reviews evidence on the dynamics of changes in ecosystems and ecosystem services, starting with restoration of degraded ecosystems.

3.1 Restoring services

Restoration ecology is a relatively new field within ecology, with the leading professional organization, the Society for Ecological Restoration, being just over two decades old. A 2009 special issue of *Science* provides a good overview of the field. An article by Rey Benayas et al. in that issue presented a meta-analysis of 89 assessments of restoration projects in a variety of ecosystems around the world. It compared physical measures of regulating, provisioning, and supporting services across three types of ecosystems: reference (i.e., relatively intact), degraded, and restored. Compared to degraded ecosystems, restored ecosystems provided "substantially higher" services than degraded systems, but the services remained much lower than in reference systems. Restoration time scales ranged from less than 5 years to 300 years, with regulating and provisioning services being restored less rapidly than supporting services. These findings suggest that restoration projects are likely to have a small effect, at best, on near-term growth.

[4] The details of this case have been debated, however. See Sagoff (2002) and Kenny (2006).

A study published earlier in 2009 under the optimistic title "Rapid recovery of damaged ecosystems" bolsters this pessimistic conclusion (Jones and Schmitz 2009). Based on a review of recovery times for degraded terrestrial and aquatic ecosystems from 240 studies, it reported "startling evidence that most ecosystems globally can, given human will, recover from very major perturbations on timescales of *decades to half-centuries*" (p. 1; emphasis added). These timescales are well outside the range of near-term growth. From an economic standpoint, additional reasons for pessimism are that the study focused on supporting services, not services more closely connected to human use, and that 67 of the 240 studies did not record recovery for any variable considered.

Adding to the pessimism, a more recent review by Bullock et al. commented that "many of the ecosystems considered [by Jones and Schmitz 2009] were relatively undegraded at the outset," adding that "Other studies have suggested that recovery occurs over longer periods of time" (p. 544). Although it also noted that "Trajectories of … ecosystem services towards the desired reference exhibit great variation in the pattern and rate of change," the evidence from it and the other studies indicates that, on average, restoration of ecosystem services occurs on time scales that economists would consider long-term, not near-term.

The review by Bullock et al. provided some information on restoration economics. It noted that data on restoration costs are rare, with TEEB finding that only 96 of the more than 20,000 restoration case studies that it reviewed contained "meaningful cost data" (p. 543). More hopefully, it observed that "new methods of ecosystem service valuation are suggesting that the economic benefits of restoration can outweigh costs" (p. 541). It cited an analysis of forest restoration in four dryland areas of Latin America, which found that restoration of forests over a *20-year time horizon* generated a positive NPV as long as the restoration approach was a passive, low-cost one: no tree planting, and no protection measures such as fencing or fire protection (Birch et al. 2010). This is hardly a robust endorsement of ecosystem restoration projects providing net economic benefits in the near term.

3.2 Reducing the loss of services

The restoration of regulating services therefore appears to be a gradual process that is unlikely to boost near-term economic growth.[5] Reducing the loss of regulating services is more likely to do this: strictly speaking, not in terms of increasing the growth rate, but rather in terms of sustaining it by avoiding economic losses. Ecosystems are increasingly understood to be highly nonlinear systems, which can "flip" to severely degraded, irreversible or near-irreversible states if levels of disturbance exceed threshold values (Steffen et al. 2004, Dasgupta and Mäler 2004). Examples include eutrophication of shallow lakes, desertification, and catastrophic fires and pest outbreaks. The evidence of ongoing ecosystem degradation presented in the Millennium Ecosystem Assessment (2005) implies that the probability of such flips, and the losses of ecosystem services that accompany them, is rising. By the same token, the benefits of reducing ecosystem degradation—the averted economic losses—are also rising.

[5] The restoration of provisioning services—specifically, marine fisheries—might be more likely to do this: Heal and Schlenker (2008) analyzed 121 fisheries that had introduced individual tradable quotas by 2003, and they found that catch increased by an average factor of three times within five years of the introduction of the quotas.

The information on service values in the previous section provides an indication of how large these benefits could be. Based on the estimates for forests, the amounts are very small compared to the overall economy. This does not mean, however, that they are necessarily small for particular economic sectors within particular parts of a country. They can be expected to be larger in locations where, in the absence of policy interventions, large-scale losses of ecosystems will occur within a matter of months or a few years, and where the threatened ecosystems provide valuable regulating services to sectors that are important to the local economy. One thinks of locations where forests are rapidly being lost to agriculture, with detrimental effects on water quality to downstream communities that are dependent on industries that rely on clean water (e.g., food processing, fishing, water-based recreation). Losses might be averted across multiple sectors in locations where ecosystems provide protection against natural disasters, whose short-run effect on economic growth tends to be negative even after accounting for the stimulus of reconstruction spending (Cavallo and Noy 2010).

The potentially most promising opportunity for developing countries to obtain large economic benefits from avoided ecosystem losses is through compensation from developed countries for reduced emissions of greenhouse gases from deforestation and forest degradation (REDD). Murray et al. (2009) estimate that a REDD program could result in purchases of carbon credits by the U.S. from developing countries valued at $32-52 billion per year during 2013-2020 if the program included only avoided deforestation, and $36-58 billion if it included other forest carbon activities. These amounts are equivalent to 2-4% of the 2009 GDP of Brazil, which accounted for more than 40% of global deforestation during 2000-2010. A review by Coren et al. (2011) indicates that the estimates by Murray et al. are on the lower end of those in the literature, and so actual payments might be even larger.

The net impact of REDD payments on growth would depend on the size of the payments relative to the value of the economic activities that countries forgo in return for the payments, which is typically some form of agriculture. The net impact would be nil if countries were compensated exactly for this opportunity cost. If the price of REDD credits is a market-wide price determined on the margin, then the largest net impacts would occur in countries with the lowest opportunity costs, which Murray et al. estimate tend to be in Africa, followed by South and Central America and then Southeast Asia.

4 Impact of programs intended to reduce loss of ecosystems and ecosystem services

4.1 Additionality, selection bias, and impact evaluation

On the surface, efforts to protect the world's ecosystems made strong progress in recent decades. The percentage of global forests under some form of protection grew strongly during 1990-2010, by a third in the case of forests designated for conservation of biodiversity and a fourth in the case of forests designated for protection of soil and water (FAO 2010). Payments for ecosystem services (PES)—especially watershed services and carbon sequestration—emerged as a means of creating financial incentives for landowners to maintain their land in a natural state (http://www.ecosystemmarketplace.com). As of 2008, there were 113 active watershed service payment programs in the world, with all but 11 located in developing countries. Control over a substantial portion of forests in some developing countries was transferred from governments to local communities, to provide the latter with a stronger stake in their management (White and Martin 2002, Sunderlin et al. 2008). Data were available on forest tenure changes during 2002-8 in 25 of the 30 most forest countries in the world, and they show that the area designated for use by communities and indigenous peoples rose in 10.[6]

At the same time, concern has been expressed that these conservation programs might not have achieved much *additional* protection. This is a different issue than the purely ecological issue of whether ecosystems actually supply the services they are assumed to, which was discussed in section 2.[7] The root of the concern is selection bias: land parcels included in protected areas or PES programs might have characteristics that would have made them unlikely to be converted even if they had not been included in the programs.[8] For example, a ministry of environment that is politically less powerful than the ministry of agriculture might be able to protect only land that is less valuable for agriculture, and thus at a lower risk of deforestation. If this is the case, then these programs have probably contributed little towards securing the supply of ecosystem services and sustaining near-term economic growth in the manner discussed at the end of the last section. Tallis et al. (2008, p. 9464) warn that "conservation groups are risking damaged reputations because they have largely failed to deliver data that provide evidence of a link between their actions and any improvement in the status of biodiversity or ecosystem services."

Economists have responded to this concern by calling for, and increasingly implementing, empirical studies that apply impact evaluation methods developed by labor and development economists to conservation programs (Ferraro and Pattanayak 2006). These methods include differences-in-differences models, regression discontinuity models, instrumental variables models, and matching methods (Ravallion 2008). Joppa and Pfaff (2010a) and Pattanayak et al. (2010) review applications of these methods to conservation programs, with an emphasis on developing countries.

Though rising, the number of applications remains small. Pattanayak et al. (2010) identify only 8 rigorous impact evaluations of PES programs, with 7 being in Latin America and 5 analyzing the same PES program in Costa Rica. A similarly small number of studies have evaluated community-based natural resource management programs, but they have a relatively even distribution across countries in Africa, Asia, and Latin America (Miteva et al., in review). Geographical coverage is broadest for evaluations of protected areas, as at least one study has analyzed a global dataset (Joppa and Pfaff 2010b).

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[6] The area decreased in one country.

[7] Wunder et al. (2008, p. 846) state that "it is fair to say that many PES programs are based on a shaky scientific foundation." Kinzig et al. (2011, p. 603), similarly observe, "Often the science is uncertain or ignored," adding that "PES schemes for water supply through afforestation face uncertainty about the net effects of changing forest cover."

[8] There is also concern about spillovers, which refer to conservation programs inadvertently damaging ecosystems in areas not included in the programs. An example would a protection program that increases timber scarcity by reducing the area of forests where logging is allowed. The resulting timber price increase would be expected to increase logging in unprotected forests. A recent study of a PES program in Mexico reported significant evidence of this sort of spillover (Alix-Garcia et al. 2011b).

4.2 Evaluating impacts on deforestation

Nearly all of these studies have focused on changes in forest cover as the measure of conservation impact. This is understandable: deforestation is a major conservation policy issue, and data from satellites and other sources are available for measuring it (and are much more available than data on specific ecosystem services). The studies find that conservation programs have had only small effects on deforestation, in the sense that only a small portion of the forest area included in the programs would have been deforested if it had not been included. In some cases, the net effect is only a few percentage points of the area included in the program.

This dismal finding suggests that the possibility mentioned above, that conservation programs have contributed little towards sustaining ecosystem services and near-term economic growth, is real. This conclusion might be too pessimistic, however, for two reasons. The first is that reduced deforestation is not necessarily the most appropriate measure of conservation impact. Protected areas are created for various reasons, not just to combat deforestation. Protection of forests on steeper slopes is often intended to reduce logging, which degrades forests but is a less common cause of deforestation than agricultural expansion (Geist and Lambin 2001, Chomitz 2006). Protected areas might also be established to protect flora and fauna against excessive hunting. Impact evaluations that focus on deforestation have not given protected areas created for these reasons a fair chance.

The second reason is that protecting forests that are at a lower risk of deforestation is not necessarily economically inefficient. Impact evaluations have provided high-quality information on the extent to which conservation programs have reduced this risk, but this is only one of three essential pieces of information for evaluating conservation programs. The other two are the benefits and costs of protection. Impact evaluations provide little guidance for conservation decisions without this additional information. A simple two-period model of conservation decision-making illustrates these points.

Suppose that a conservation agency has a budget of B, which it can use to reduce deforestation by acquiring and protecting forestland. The country's forests are of two types (i = 1,2), with acquisition prices of P_i per parcel, conservation values of V_i per parcel, and deforestation probabilities of d_i between the first and second periods. The agency's budget enables it to purchase B/P_i parcels of forest type i. The agency is economically rational, and so its objective is to maximize the expected net present value of avoided deforestation. For forest type i, the expected net present value is given by

$$[d_iV_i(B/P_i)]/(1+r) - B,$$

where r is the discount rate between the two periods. Suppose that the expected present value is higher for type 1 forests. Eliminating identical terms, this implies:

$$d_1V_1/P_1 > d_2V_2/P_2,$$

which is the expected benefit/cost ratio per parcel.

Impact evaluations provide information on d_i, the avoided deforestation rate, but this does not provide sufficient information for identifying type 1 forests as the best ones to protect. Type 1 forests can be the best ones to protect even if $d_1 < d_2$, as long as V_1/P_1 is enough larger than

V_2/P_2. In some situations, estimates of d_i might not even be necessary for conservation decisions. It is reasonable to assume that P_i is positively correlated with d_i: higher demand for land implies both a higher deforestation risk and a higher land price (Costello and Polasky 2004, p. 160). If these two variables are perfectly correlated, then both are irrelevant to the conservation decision, which instead depends entirely on relative conservation values, V_1 vs. V_2. In this special case, valuation is needed to make the economically efficient conservation decision, but impact evaluation is not.

Knowledge of d_i is sufficient for making the correct conservation decision in a different special case, where $d_2 = 0$ and $d_1 > 0$: forest acquisition provides no additional protection in the case of type 2 forests but some additional protection in the case of type 1 forests.[9] Though intuitive, this result does not necessarily hold if the model is slightly reformulated. Suppose it is known that type 2 forests will be deforested in the current period, while type 1 forests will not be deforested for n years. Protecting the less threatened forest type, i.e. type 1, is still the economically optimal decision if

$$V_1/(P_1(1+r)^n) > V_2/P_2.$$

Again, benefits and costs need to be considered, not just the risk of deforestation, which is here captured by the time until deforestation occurs. The implication of this example is that an impact evaluation that determines a zero immediate probability of deforestation for parcels included in a conservation program cannot be assumed to have identified a bad conservation decision. The decision could instead be a smart, forward-looking one, akin to buying land in the desert before it becomes Las Vegas.

The impact evaluation literature sometimes holds up medical trials as the gold standard for measuring program effectiveness (Deaton 2010). This analogy is potentially misleading for conservation programs. In the case of a medical trial, researchers are appropriately interested only in determining whether the treatment works, because the cost of administering the treatment is typically the same across patients and because ethical considerations typically preclude valuing successful treatment differently for patients of different genders or races. In conservation decisions, both costs and benefits can vary across “patients.” Costs and benefits, and not just treatment effectiveness, must be considered if conservation decisions are to be economically efficient.

The need to consider the benefits and costs of conservation decisions, and not just the level of threat to different ecosystems that could be protected, is well understood in the conservation economics literature (Polasky et al. 2005) and has been acknowledged by conservation impact studies (e.g., Joppa and Pfaff 2010a). Impact evaluations have greatly improved the quality of estimates of conservation programs’ effectiveness in protecting ecosystems, but they would be more useful for conservation decisions if they were better integrated with the literatures on conservation costs and benefits (e.g., Polasky et al. 2001, 2008;

[9] Strictly speaking, knowledge of d_i is not sufficient in this case, as one also needs to know whether the expected net present value is positive for type 1 forests: $[d_1V_1(B/P_1)]/(1+r) - B > 0$, which requires for information on benefits and costs. Knowledge of d_i is also “sufficient” in this sense if $V_1/P_1 = V_2/P_2$, but land prices are less likely to be positively correlated with conservation benefits than with deforestation risk.

Naidoo and Ricketts 2006). A study on community-managed forests by Somanathan et al. (2009) provides a rare example of combining impact evaluation results with information on conservation costs. The review by Ferraro et al. (in press) indicates that no impact evaluation study has yet been similarly combined with information on conservation benefits.

5 Conservation programs and poverty alleviation

5.1 On-site vs. off-site effects of environmental degradation on poverty

In a series of works, Dasgupta has argued persuasively that the welfare of poor rural households is linked to environmental resources more strongly than the welfare of other households (e.g., Dasgupta 2003). A World Bank (2008) review of empirical work on poverty and the environment confirms that environmental income tends to be a larger share of total income for poor rural households than for others. "Environmental income" in that review referred to provisioning services, such as food, fuel, and fiber harvested from forests, fisheries, and rangelands. Dasgupta has also argued that institutional failures of various types are often the common cause of rural poverty and environmental degradation. The classic example is the "tragedy of the commons": open access to common-pool resources induces resource users to expend too much effort on harvesting, which degrades the resource and drives income down to a subsistence level. Addressing the root institutional failure of the tragedy—the lack of restrictions on harvest effort—can yield a "win-win" outcome of a healthier ecosystem and higher household income.

In the tragedy of the commons model, the benefits of restricting harvests accrue fully to the community using the resource. The case of regulating services is different in that some of the benefits of changed behavior by a resource-using community are typically external to that community. For example, improved water quality resulting from reduced conversion of upland forests might benefit downstream communities as much or more than it benefits upland communities. Barbier (2010) cites numerous case studies of how the protection of fragile lands can benefit poor offsite communities by increasing the supply of regulating services. In such cases, institutional reforms that eliminate open access to, say, an upland forest might help restore forests and alleviate poverty in the upland community, but they might not supply regulating services at the higher level that is optimal when the welfare of downstream communities is added in. Achieving the optimum would require additional policy interventions, such as protected areas and PES programs, and these additional interventions could have more complicated effects on poverty. For example, although a protected area in the uplands might alleviate poverty in downstream communities by increasing the supply of watershed services, it might worsen poverty in the upland community by restricting the community's use of upland forests.

Evidence on the effects of conservation programs on poverty comes from a mix of theoretical models, case studies, and impact evaluation studies. It indicates that a "win-win" outcome is far from assured. Tallis et al. (2008) report that only five of 32 World Bank biodiversity conservation projects approved during 1993-2007 achieved substantial gains in terms of both environmental protection and poverty alleviation according to Bank's own evaluation system. We review evidence first for protected areas and then for PES programs.

5.2 Protected areas and poverty alleviation

Robalino (2007) presents a basic theoretical model of the effects of protected areas on the income of workers vs. landowners in a perfect markets setting. The economy is closed and has two sectors, agriculture and manufacturing, whose prices vary with distance from a central city. Agriculture requires two inputs, land and labor, while manufacturing requires just labor, which is perfectly mobile between the two sectors. Creation of a protected area reduces the amount of land available for agriculture. Robalino finds that this causes unit land rent to rise in all locations, and the increase is sufficiently large that aggregate land rents rise despite the reduction in cultivated area. Real wages decrease, however, and so does worker consumption. If the rural poor tend to be landless workers instead of landowners, which is typically the case, then this model implies that protected areas worsen rural poverty.

Several recent impact evaluation studies have examined the empirical effects of protected areas on poverty in Thailand and Costa Rica. In contrast to the theoretical result just mentioned, they find that protected areas reduce poverty in their vicinity, and by considerable amounts. Sims (2010) used instrumental variables to control for the nonrandom locations of protected areas in Thailand. She found that an increase in the protected share of a locality from zero to half raised household consumption by nearly a fifth and reduced the poverty headcount ratio by 0.1, which was about half of the sample-wide mean. Andam et al. (2010) applied matching methods to data from both Costa Rica and Thailand and found similarly large and beneficial effects.

The contrast between the theoretical and empirical effects of protected areas on rural poverty has an obvious explanation: protected areas do not provide any ecosystem services in Robalino's model. Protection does not directly benefit either of the sectors in his model: it does not benefit agriculture by enhancing regulating services (e.g., watershed protection or pollination), or manufacturing by enhancing provisioning services (e.g., greater timber supply if forests are not converted to agriculture). It simply reduces the stock of one of the two factors in the model, agricultural land, and the cost of this reduction is borne by the other factor, labor.

Two other studies in Costa Rica offer evidence that a sector excluded from Robalino's model, tourism, plays an important role in reducing poverty near protected areas. Robalino and Villalobos-Fiatt (2010) used matching methods and Costa Rican household data for 2000-7 to compare the wages of workers living near national parks to the wages of workers in similar regions that were not near parks. They found that the wages of agricultural workers were no different near parks than far from them. On the other hand, wages for workers in restaurants and hotels, and wholesale and retail trade, were higher if they were not merely near parks, but near their entrances. This suggests that parks can stimulate the growth of a tourism industry that increases local wages, but the effect is very local, just near park entrances.

Ferraro and Hanauer (2011a) used an augmented matching approach and data from Andam et al. (2010) and Robalino and Villalobos-Fiatt (2010) to estimate the contributions of three factors to poverty reduction: tourism development (proxied by the creation of park entrances), infrastructure development (change in the extent of the road network), and increased ecosystem services (proxied by the change in forest cover). They concluded that tourism was responsible for about half of the observed poverty reduction, with infrastructure development being responsible for about a tenth. Ecosystem services had no effect.

Results from these two studies indicate that the poverty effects of protected areas vary spatially. Two other recent studies have investigated the spatial correlation between protected areas' effects on deforestation and poverty: does protection reduce deforestation and poverty in the same areas (a spatial "win-win") or different ones? Both reanalyzed data from the Costa Rican and Thai studies cited above. Ferraro and Hanauer (2011b) focused on Costa Rica. They grouped land parcels according to high and low levels of five characteristics: land use capacity, slope, distance to major city, percent agricultural workers, and initial poverty. They found that protection either reduced or did not worsen deforestation and poverty in each subgroup pair, so in a broad sense protection was either "win-win" or at least "win-draw." The subgroup rankings were reversed for the two outcome variables, however. For example, protection reduced deforestation more on parcels with high land use capacity but poverty more on parcels with low land use capacity. So, the locations where protection had the greatest effect on deforestation were different from the locations where it had the greatest effect on poverty. Ferraro et al. (2011) examined both Costa Rica and Thailand and reached a similar conclusion.

5.3 PES programs and poverty alleviation

Evidence on the poverty effects of PES programs comes from theoretical studies and case studies. Apparently no rigorous impact evaluations have yet been conducted on this issue, in contrast to the case of protected areas. Starting with the theoretical studies, Muller and Albers (2004) consider a rural setting in which agricultural households extract a resource from a neighboring protected area. The agency managing the protected area is interested in reducing the rate of extraction. It can implement three conservation policies, alone or in combination: it can pay households according to the amount by which they reduce extraction, which is analogous to a PES program ("conservation payment"); it can fund a program that raises agricultural productivity and induces "conservation by distraction" ("agriculture development project"); and it can hire guards to prevent extraction ("patrolling"). Muller and Albers find that the effects of these policies on extraction and household welfare depend on the completeness of markets for labor and the resource. In all cases, however, the conservation payment (and the agricultural development project) raises household welfare, while patrolling lowers it. The latter result echoes Robalino's (2007) finding that protection reduces the real wages of rural workers. As in that study, however, Muller and Albers did not model the possibility that protection could stimulate the growth of a local tourism industry. Absent that possibility, the implication is that a PES program is more likely to alleviate rural poverty than stricter enforcement of a protected area.

In a subsequent theoretical study, Zilberman et al. (2008) broaden the scope by considering the effects of PES programs on not only different groups in rural areas (landowners, landless) but also urban households. Urban households are the beneficiaries of an ecosystem service, such as water quality, whose supply is diminished by conventional agricultural practices. The supply of this service can be enhanced by paying farmers either to reduce the area cultivated ("land-diversion PES") or to adopt more environmentally friendly farming practices ("working lands PES"). Zilberman et al. find that neither type of PES program has an unambiguous effect on poverty. Land-diversion PES programs can benefit poor landowners whose land is relatively more valuable for ecosystem services than agriculture and poor urban households who value the ecosystem service highly. If these conditions do not hold, however, then land-diversion PES programs tend to benefit larger landowners but make smaller landowners, the landless, and the urban poor worse off. The effects of working lands PES programs depend on the relative magnitudes of their effects on wages, which they tend to raise, and crop prices, which they also tend to raise but by a lower amount. The rural and urban poor sometimes gain and sometimes lose.

Wunder (2008) reviews the case study evidence, which he refers to as preliminary and limited. (See also Wunder et al. 2008.) He considers effects on three groups: rural households that participate in PES programs and receive payments, nonparticipating rural households, and urban households that receive services the programs are intended to supply. He reports that the case studies indicate that "poor people can widely participate [as providers] in PES schemes, [and] that this participation usually makes them better off" (p. 279). He cautions, however, that "it seems that per-capita provider income gains are seldom impressively large," which he attributes to urban service recipients being "in a better negotiating position to appropriate the 'gains from trade'" by keeping payments low (p. 294). He notes that whether urban households receive benefits that exceed their payments depends fundamentally on whether PES programs actually increase the supply of services. Evidence presented earlier in this paper suggests that not all programs do: watershed PES programs might not mitigate floods and droughts, though they might improve water quality; PES programs that restore ecosystems might not provide significant services for many years; and forest-related PES programs might reduce deforestation by only small amounts.

Regarding nonparticipating rural households, Wunder refers to opposing effects of the sort identified by Zilberman et al. and concludes that "In most cases, these effects are mixed, and minor in size" (p. 295). Nonparticipating rural households include the landless, who he notes are typically the poorest group in rural areas. Exclusion of the landless from PES programs limits the ability of the programs to alleviate the worst rural poverty.

Although impact evaluations have not yet been conducted on the poverty effects of PES programs, they have been conducted on conditional cash transfer (CCT) programs, which are conceptually similar. Under CCT programs, poor households receive cash payments in return for agreeing to invest in their children's human capital through enrollment in health or education programs. PES programs are like CCT programs, only the investments are in natural capital.[10] A recent review of CCT programs concludes that they have generally "raised [short-run] consumption levels, and have reduced poverty—by a substantial amount in some countries" (Fiszbein & Schady 2009, p. 12). A potential concern with the programs has been that they might reduce adult labor, but the evidence indicates that this either hasn't happened or, if it has, that the disincentive to work has been modest. On the other hand, CCT programs have reduced

[10] At least one study has considered the environmental effects of CCT programs. Alix-Garcia et al. (2011a) analyzed *Oportunidades*, a CCT program in Mexico, using differences-in-differences and regression discontinuity models. They found that the program increased deforestation by inducing households to adopt more land-intensive forms of agriculture.

child labor, which is not surprising, particularly for ones linked to educational enrollment. PES programs are less likely than CCT programs to reduce household labor, as they do not require adults or children to spend additional time on children's health or educational activities. For this reason, a PES payment of given size seems more likely to boost near-term growth than a CCT of the same size.

CCT programs aim to raise consumption and reduce poverty in the long run through their positive effects on human capital. They could also have these long-run effects if households invest some of the payments or are better able to obtain credit thanks to the payment stream. Evidence on this channel for long-run effects is limited, but it generally indicates that little or none of the payments is invested (Fiszbein & Schady 2009, p. 12). This is consistent with the programs having raised short-run consumption, as noted above. If households similarly consume most of the payments under PES programs, then the local growth stimulus of PES payments—as opposed to the growth stimulus from ecosystem services—tends to be more near-term than long-term. Wunder's (2008) finding that PES payments tend to be low suggests that any such stimulus is small. Whether a local stimulus contributes to an economywide increase in near-term growth depends on how the marginal propensity to consume (MPC) of households that receive payments compares to the MPC of ones that make payments (Spilimbergo et al. 2009, Ono 2011): a net increase occurs only if the former is higher than the latter. The MPC is usually assumed to be higher for poor households, so the income transfers that occur under PES programs are more likely to boost near-term growth if richer households make the payments and poorer ones receive them.

Many authors are deeply skeptical about the ability of PES programs to alleviate poverty in addition to achieving their environmental objectives, and they view the addition of poverty-alleviation side objectives as ultimately counterproductive. Wunder (2008, p. 280), Ferraro (2009, p. 544), and Kinzig et al. (2011, p. 604) argue that such side objectives increase the transactions costs associated with PES programs, delay implementation, and make the programs less likely to succeed in supplying ecosystem services, all of which reduce the programs' ability to improve social welfare. If the adoption of PES programs is politically impossible without the inclusion of poverty-alleviation side objectives, CCT programs might offer lessons on how to include these side objectives more efficiently. According to Fiszbein & Schady 2009, p. 12), payments under CCT programs have "generally have been well targeted to poor households."

6 Conclusions

A sequence of four conditions must hold for ecosystem services to provide a near-term stimulus to economic growth:

5. A conservation program must either restore a degraded ecosystem or reduce the loss of an intact one.

6. This positive ecosystem change must increase the supply of an ecosystem service.

7. The increased supply must occur within the near term.

8. The service must be economically valuable.

This paper has reviewed available evidence on these conditions for the particular case of regulating services, which enhance economic production. It has furthermore focused on services that are domestic and do not cross national borders. Its findings for each condition can be summarized as follows:

1. Rigorous evaluation of the effectiveness of conservation programs is growing but still limited. So far, impact evaluations have primarily focused on reduced loss of ecosystems, not restoration, with loss almost always measured by deforestation. They generally find that conservation programs have reduced deforestation by only small amounts, but this dismal finding is potentially misleading because deforestation is not the primary objective of many forest conservation programs and because economically efficient conservation decisions do not necessarily entail protecting the most threatened sites. Conservation decisions are also affected by the benefits and costs of conservation programs. The highest expected net present value of protection can occur at sites other than those that are most threatened.

2. Rigorous evaluation of the effects of ecosystem changes on the supply of services is similarly limited. A common problem with statistical analysis of historical ecosystem changes is inadequate control for potentially confounding factors, which can result in exaggerated estimates of impacts on service supply. Estimates of flood mitigation and pollination services by forests are good examples of this problem.

3. The time required for ecosystem restoration projects to restore services is typically measured in decades, which is beyond the range for affecting near-term growth. Programs that reduce the loss of ecosystems and their associated regulating services are better candidates for stimulating near-term growth, especially when ecosystems and services are being lost rapidly and are approaching thresholds that can cause them to flip to a highly degraded state.

4. The value of ecosystem services has been studied more for forests than for any other ecosystem. These studies indicate that, on average, the aggregate value of forests' regulating services is less than the commercial value of timber, which in turn implies that the services typically account for less than a percentage point of GDP. This small magnitude limits the opportunity to boost economic growth through enhanced supply of regulating services. Service values vary tremendously across and within countries, however, and so opportunities for a growth stimulus might exist in particular locations within particular countries.

Although information on all four conditions is far from perfect, two points are clear: conservation programs that protect ecosystem services are more likely to provide a near-term growth stimulus than ones that restore ecosystems, and the stimulus is likely to be local and to have a small effect on the national economy. Impact evaluations of protected areas indicate that local effects can include reductions in poverty, although not necessarily in the same locations that experience the greatest conservation gains. Local poverty effects of PES programs are less clear, with case studies indicating that the programs provide little benefit to the poorest rural segment, the landless. A transboundary regulating service—climate stabilization—appears to have the greatest potential to have an economywide impact. This depends on pilot programs that compensate developing countries for reducing emissions of greenhouse gases from deforestation and forest degradation (REDD) becoming part of a mandatory global program to address climate change, and on the payments exceeding the opportunity costs incurred by REDD countries.

This assessment points toward the following topics as high-priority ones for research on the links between ecosystem services and near-term economic growth:

- rigorous evaluation of the impacts of conservation programs, with a broader range of outcomes measured besides deforestation, a stronger effort made to relate estimates of conservation effectiveness to conservation benefits and costs, and continued progress toward understanding spatial variation in conservation effectiveness, benefits, and costs;

- the time path of service losses when ecosystems degrade and the economic impacts of such losses, with careful control for potentially confounding factors that, if ignored, can exaggerate estimates of service losses and their economic impacts;

- improved methods for scaling up micro-level studies on ecosystem service values to scales, such as economic sectors or political subdivisions, that can be linked more readily to standard statistics on economic performance;

- given emerging evidence that tourism plays an important role in enabling conservation programs to alleviate poverty, more research on the growth impacts of cultural services, to complement the research on regulating services highlighted in this paper.

A final comment is a reminder that this paper's focus on the near-term economic growth effects of ecosystem services should not be taken as implying that ecosystem protection is justified only if it stimulates near-term growth. Protection of ecosystems is justified whenever it raises intergenerational well-being, regardless of the consequences for near-term growth.

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