Payment for Ecosystem Services from Forests

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Abstract

Every year from 2000 to 2010, our planet lost native forests roughly the size of Costa Rica (FAO 2010). This rapid deforestation has dramatically changed the chemical composition of the world's atmosphere, the level of biodiversity, and the presence of vegetation key to maintaining watershed function and preventing landslides. There has been a boom in the design of local and international policy instruments to prevent further deforestation and to encourage forest growth. This article reviews the theory and evidence surrounding forest-related payment for ecosystem services (PES) schemes intended to slow and reverse deforestation. We cover the most recent work touching on a range of issues related to PES programs, including research on targeting, contract design, environmental effectiveness, challenges to program implementation, spillovers, and distributional considerations of conditional cash transfers. We also highlight areas of potential future research.

1. INTRODUCTION

During the first decade of this century, global net forest loss totaled more than 5 million hectares per year, with 13 million hectares completely destroyed on a yearly basis (FAO 2010). This corresponds to the disappearance of 1.5 soccer fields of forest every two seconds, or the equivalent of the area of Costa Rica annually. It has recently been estimated that the total contribution of deforestation and forest degradation to anthropogenic carbon emissions is approximately 12%, making these activities the second most important carbon source behind fossil fuels (Friedlingstein et al. 2010). In addition to carbon, forests also provide many important local environmental goods such as erosion control and the maintenance of watershed functions. Furthermore, forest in the tropics houses tremendous biodiversity—a warehouse of potential future benefits to society as a whole.

Driven partially by estimates in the *Stern Report* (Stern 2006), the global perception has been that reducing emissions from deforestation or providing additional sinks through the regeneration and expansion of forest would be inexpensive ($1–2 per ton of carbon) relative to other approaches to climate change mitigation. This perceived low cost is the result of the global distribution of forests and deforestation, which is shown in **Figure 1**. **Figure 1** reveals that Europe and South America contain the largest areas of forest resources, whereas Central America houses the highest rate of forest loss, followed by Africa and South America. In comparison, forest resources in North America, Europe, and Asia appear to be at little risk. The possibility of potential low-cost carbon sequestration in developing countries accelerated the development of various afforestation, deforestation, and reforestation programs, including the growing movement of payments conditional on forest conservation that had been developing in Latin America. In 1997, Costa Rica developed one of the first national-level forest payment for ecosystem services (forest PES) programs explicitly intended to pay for avoided deforestation. Other national-level programs quickly followed.[1] Throughout the late 1990s and early 2000s, a large number of city- and regional-level PES programs were initiated. Landell-Mills & Porras (2002) documented more than 300 payment incentive programs worldwide.

That class of forest PES programs is the topic of this review. In particular, we intend to examine the recent theoretical and empirical literature on forest PES and to highlight areas in which further research is necessary. Specifically, we examine work on programs that award transfers conditional on either afforestation (expansion of forest), reforestation (replanting of forest), or avoided deforestation (prevention of forest loss). China's Sloping Land Conservation Program (SLCP) (Uchida et al. 2005), the Greening India program (Balooni & Singh 2001), and Costa Rica's Programa de Pagos por Servicios Ambientales (PSA) (Arriagada et al. 2012) provide well-known cases of programs focused on afforestation, reforestation, and avoided deforestation, respectively.

There are several excellent recent reviews of forest PES, including Pattanayak et al.'s (2010) review of the environmental effectiveness of avoided deforestation payments and the PES section of Pfaff et al. (2013a), which examines how PES might address underlying drivers of deforestation. Jack et al. (2008) provide an illuminating review of policy lessons from early work on the topic. There are also extensive overviews of the relationship between PES and poverty alleviation (see Bulte et al. 2008 and, more recently, parts of Samii et al. 2013, among others). Our purpose here is not to repeat the discussions of these papers. Instead, we make an effort to focus on the latest developments, citing mostly papers that appeared recently—roughly from 2005 to

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[1]Historically, the earliest versions of PES payment schemes have been in existence since at least the 1980s, when payments to upstream farmers were designed to preserve water quality in Munich and New York City (Grolleau & McCann 2012).
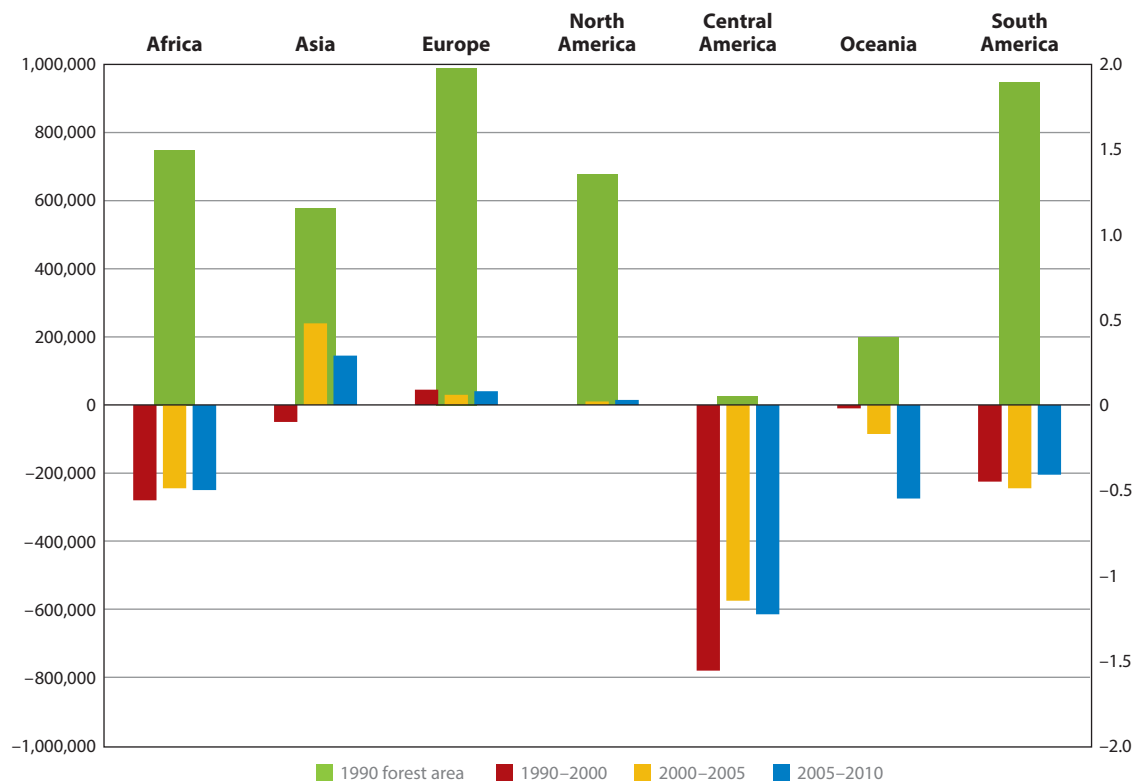
Figure 1

Forest area and net forest change, 1990–2010. The left vertical axis measures forest areas as of 1990 by world region, and the right vertical axis measures subsequent rates of forest loss from 1990. From FAO (2010).

mid-2013—although historical context is often provided by older work as well. This article is meant to complement the existing reviews and should hence be of interest to the reader who aims to update herself on the current status of the literature. Furthermore, we hope to broaden the perspective by examining afforestation and reforestation PES efforts in addition to avoided deforestation efforts.

A forest PES contract at its core is a Coasian market mechanism to internalize the negative externalities associated with forest loss and to subsidize the positive externalities associated with forest expansion. Therefore, we begin our review in Section 2 by considering the nature of demand for forest services. This section discusses how, on the demand side, the identity of the PES consumer dictates the scale of a PES program. Examples range from single-city watershed services to global transfers addressing climate change. Section 3 examines the supply side. Issues related to the identity of the PES seller, targeting, pricing, and the extent to which PES programs generate environmental services are reviewed. Section 4 describes additional challenges to implementation that are important for the scaling up to a national or global level: the common problem of insecure land tenure in many developing countries, and the possibility of program leakage. Section 5 considers potential social spillover effects of PES programs, with much of the space dedicated to work discussing whether PES can alleviate poverty and can affect other household or community behaviors. Finally, our conclusion highlights areas for further research.

2. DEMAND: WHO BUYS FOREST ECOSYSTEM SERVICES?

This section presents a variety of examples of environmental service buyers.[2] On a spatial level, forest PES projects can be categorized into three groups: local, regional, and global. For example, landslides resulting from localized deforestation represent a very distinct environmental service from carbon emissions. The location of the forest has almost no impact on the forest's contribution to the global carbon service. In principle, when we move from local to global, the transaction costs of organizing and implementing the PES scheme increase considerably, as the examples in the following subsections demonstrate. Interestingly, projects of all spatial scales have both private and public sector buyers with specific interests in the goods produced.

2.1. Local Public Goods

Local public goods associated with forest preservation and expansion include hydrological services and erosion prevention, among others. These types of programs are now relatively common in Latin America, where they have often been spearheaded by cities concerned with their water supply (Wunder & Albán 2008, Arriagada et al. 2012, Van Hecken et al. 2012). In Ecuador, for example, the municipality of Pimampiro targeted 27 households holding land in the watershed that provides the city's water, eventually contracting with 19 to halt their agricultural expansion into the forest and alpine grassland (Wunder & Albán 2008). Similar arrangements have been developed in Coatepec, Mexico, where efforts to address water security led to Mexico's first hydrological services PES (Scullion et al. 2011).

An excellent example of a private buyer is the water-bottling company Vittel, which negotiated agreements with the farmers in the catchment area feeding the company's spring source (Wunder & Wertz-Kanounnikoff 2009). The current Vittel scheme, which has been in place for more than ten years, consists of 1 buyer and 26 sellers of ecosystem services and covers an area of approximately 3,500 ha. Further examples involve hydropower producers, water bottlers, and tourism, all in Costa Rica (Pagiola 2008). These local agreements embody the Coasian ideal of private consumers dealing directly with suppliers to arrive at mutually agreeable terms (Coase 1960), although in many cases, such as in Costa Rica, intermediary nongovernmental organizations (NGOs) play an important role in brokering deals.

2.2. Regional or National Public Goods

Because of the large number of landowners involved, regional agreements are usually driven by public buyers at the state or national level. Common examples of regional public goods include hydrological services and erosion control. A prime example of this type of policy is China's Grain for Green program, which has the goal of preserving and improving water as well as soil quality for entire river basins through set-asides of sloped land for re- and afforestation. Uchida et al. (2005) summarize that, between 1999 and 2001, participating farmers converted nearly 1.2 million hectares of cropland into forest and pasture and afforested nearly 1 million hectares of land. The program goal is to have set aside nearly 15 million hectares of cropland by 2010, an area almost

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[2]This overview does not by any means constitute an inventory of PES schemes (which can be found in Landell-Mills & Porras 2002 and Pattanayak et al. 2010) but rather hopes to emphasize how the nature of the externality affects the scale of its purchase.

equivalent to that covered under the US Conservation Reserve Program (CRP) (Uchida et al. 2005). Further examples are Costa Rica's PSA, in which the national government purchases avoided deforestation for regional hydrological services (Pagiola 2008), and Mexico's Payments for Hydrological Services program (PSAH).

Two interesting cases of private regional buyers are Ecuador's PROFAFOR and Panama's ForestRe. PROFAFOR is an extension of the Forests Absorbing Carbon Dioxide Emissions consortium (which is in turn financed by Dutch energy companies) and signs contracts for afforestation and reforestation, accounting for almost half of all Ecuadorian reforestation since its inception in 1993 (Wunder & Albán 2008). ForestRe, in contrast, is a reinsurance firm that established a watershed protection plan along the Panama Canal to reduce outlays on dredging and upkeep of the canal. The firm established a 25-year bond (funded by shippers and insurance firms) to restore forest ecosystems (**http://www.forestre.com/**).

2.3. Global Public Goods

Carbon sequestration and biodiversity are the two most prominent examples of public goods whose benefits transcend national boundaries. Today, both public and private buyers are very active in this market. Public buyers include national governments as well as multilateral purchases organized through international agreements by the United Nations and the World Bank. In fact, without considerable internal political support, it remains difficult for a single government to justify individual action on global public goods. In one of the few examples of a single-country-driven PES for global public goods, Mexico in 2004 designed the Payments for Carbon, Biodiversity, and Agroforestry (PSA-CABSA) program. The program was later combined with the much larger PSAH, whose funding is predicated on providing public goods whose main beneficiaries are within the territorial boundaries of Mexico (Corbera et al. 2009).

Various global agreements attempt to overcome the cooperation dilemma. A long-standing example is the Clean Development Mechanism (CDM). Agreed upon in the United Nations Framework Convention on Climate Change (UNFCCC) and implemented by the Kyoto Protocol in 1997, the CDM allows firms to purchase certified emissions reductions (CER) from offsets brought about in developing countries. The main purchaser of CER has been the EU Emissions Trading System, which uses these CER as part of the approved offsets for member states (Newell et al. 2013). Currently, the CDM projects can involve afforestation and reforestation projects only. The CDM cannot cover avoided deforestation projects. Thomas et al. (2010) note that, worldwide, CDM forest projects account for less than 1% of the total of 8,866 CDM projects but could potentially expand in the near future if further cap-and-trade systems are implemented.

The second global mechanism of increasing importance is the UNFCCC initiative Reduced Emissions from Deforestation and Forest Degradation (REDD). REDD is a large-scale forest PES program, with funds being transferred from developed countries to developing countries. In contrast with the above-discussed forest CDM projects that focus on afforestation and reforestation, REDD, as the name suggests, focuses on deforestation and forest degradation projects. Initially, REDD started as an effort to design incentives to create value for stored forest carbon. Although the roots of REDD can be found in much earlier climate negotiations, the mechanism was first formalized in the 2005 eleventh Conference of the Parties (COP) in Montreal. The 2007 negotiations in Bali added reforestation and forest enhancement to the list of potential REDD strategies and rechristened the agreement as REDD+. REDD+ also broadens the agreement to allow for the simultaneous consideration of other environmental goals (e.g., biodiversity,

sustainable forest management) and of other socioeconomic aspects (e.g., indigenous rights and equity in distribution of funds and financing mechanisms).[3] Multiple international organizations have stepped in to facilitate the development and financing of REDD+ programs. Such organizations include the World Bank's Forest Carbon Partnership Facility (FCPF), which accepts both public and private funds, and UN-REDD.

National and local forest PES programs, however, are just one of a variety of strategies that countries might use to achieve REDD+ goals. Currently, most REDD+ programs are in the development stage. To date, UN-REDD has financed the development of REDD+ strategies in 16 partner countries, and FCPF has financed development in 36 countries. Although the economics literature on evaluating these REDD projects is scarce, below we discuss various recent papers, highlighting different examples of private, public, and NGO-based REDD-type initiatives.

Some of the best examples of up-and-running REDD projects have been created in the private sector and by NGOs. Asner et al. (2010) describe the Madre de Dios Amazon REDD project, created by the NGO Greenoxx (**http://www.greenoxx.com/en/madre-de-dios-the-project.asp**). Covering 100,000 hectares of the Peruvian Amazon, this project is expected to generate 9.5 million carbon credits over a 10-year period. The first tons of carbon traded by the project occurred in May 2010. A second privately financed REDD project is run by Marriott International, which pledged US$2 million toward the REDD project of the Juma Reserve in Brazil (Champagne & Roberts 2009). Other interest from the private sector has come from airlines, which offer carbon offsets for purchase by individuals wishing to offset carbon emissions generated by their air travel (Sarkar et al. 2009).

Private individuals and conservation groups have also engaged in the direct purchase of forestlands to provide environmental services. Land Matrix, an NGO devoted to tracking international land purchases, categorizes approximately 5% of the 70.5 million acres in its database as conservation-oriented purchases (**http://landmatrix.org**). Wealthy individuals have famously purchased large amounts of land, including more than 400,000 acres in the Amazon and several million acres in Patagonia (Vidal 2008, Mukerjee 2009). Although these private initiatives have been met with skepticism by some governments in developing countries, which have dubbed such efforts ecocolonialist, this type of direct action may be a viable alternative to waiting for national governments to come to international agreements. Finally, NGO purchases of forestland are an order of magnitude greater than private individuals' purchases. Many NGOs, such as The Nature Conservancy, Cool Earth, Woodland Trust, Wildlands Project, and Conservation International, work on both types of purchases, forming partnerships with local charities for conservation purchases and for outright purchases of tracts of land (see **http://adopt.nature.org/**, **http://coolearth.org/**, **http://www.woodlandtrust.org.uk/**, **http://www.wildlandsprojectrevealed.org/**). Beyond discussion of the ethics of such purchases (e.g., Fairhead et al. 2012), rigorous academic research on either the purpose or the impact of land purchase on environmental outcomes is extremely limited.

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[3]According to the United Nations program, REDD+ is distinguishable from REDD primarily through the following four aspects. (*a*) Equal priority is given to emissions reductions through deforestation and degradation and to removals through sinks such as conservation, sustainable management of forests, and enhancement of forest carbon stocks. (*b*) Long-term estimations of emission and removals should be done on a land basis instead of on an activity basis because land-based approaches more accurately reflect the land's true effect on the environment and these approaches are more consistent with the principle of environmental integrity. (*c*) The rights of indigenous peoples and new social and environmental safeguards are included. (*d*) Concepts of financial mechanisms and the equitable distribution of funds are introduced. For details, see **http://www.un-redd.org/FAQs/tabid/586/Default.aspx**.

In summary, the vast majority of locally and nationally financed forest PES programs are associated with hydrological services. This scenario is consistent with the fact that national governments need to engage with services whose benefits accrue most directly to local taxpayers. International environmental services, such as those involving carbon sequestration and biodiversity, are most frequently financed by international organizations and NGOs. The challenges of organizing players to fund global environmental goods suggest that their provision will be considerably farther from optimal than are more localized benefits. Private initiatives by the airline industry, hotel chains engaging in offsets, and private purchases of land are relatively new developments, and there is very little empirical evidence on the effectiveness and the distributional effects of these projects.

3. THE SUPPLY OF FOREST ENVIRONMENTAL SERVICES

In contrast to the case for environmental service demand, which can operate on a great variety of spatial scales, the forest environmental service suppliers who participate in PES programs are mostly individuals or small communities. The literature on the supply side broadly focuses on two issues: the targeting of the individuals providing the services and the environmental effectiveness of PES programs. An important driver of effectiveness is the ability to target and calibrate payments such that the owners of land desired for inclusion in the program choose to enroll. This section begins by briefly summarizing the literature on targeting and pricing, then discusses the characteristics of the suppliers, and finally examines some existing PES programs with respect to their effectiveness in supplying environmental services.

3.1. Targeting and Contracts: Theory and Reality

The problem of studying which land to enroll into PES programs (targeting) has much in common with the challenge of targeting conditional cash transfer programs in general in the sense that the efficacy of such programs depends upon identifying willing sellers in whom the policy can induce an actual behavior change. In this section, we review work that describes optimal targeting strategies, issues of hidden information regarding opportunity costs, and ways in which these costs might be circumvented in the PES context. We also compare the theory on contracts with the reality of implementation in existing programs.

According to standard economic theory, any program whose goal is to effectively provide environmental services should seek to maximize expected net benefits. To do so at least cost, payments should be equivalent to the opportunity cost of the supplier. This principle is best described by Babcock et al. (1997) in the context of the CRP and is further discussed in the general equilibrium literature on climate change (i.e., Falk & Mendelsohn 1993) as well as in the simulation-based literature that examines the costs of large-scale forest carbon sequestration policies. Among this work, key papers include those of Sohngen & Mendelsohn (2003), who are the first to point out the relatively low cost of carbon sequestration in forests, and Lubowski et al. (2006), who estimate opportunity costs by using detailed observational data in the United States. Although these are important calculations, this literature typically abstracts away from institutional details of implementation. The remainder of this section describes recent work that focuses on these institutional aspects to elucidate the difference between theory and reality in the field.

Targeting for afforestation and targeting for avoided deforestation present different challenges. The latter turns out to be much more difficult than choosing land for afforestation or for silvopastoral practices, because policy makers need to predict where individuals would like to deforest in the near future (Alix-Garcia et al. 2008). This hidden-information problem between the landowner and the government is significant and is described in the literature both for projects in

developed countries like the United States (Mason & Plantinga 2013) and in developing country contexts (Ferraro 2008). Ferraro (2008) nicely lays out the informational challenges of PES contracts, as well as the advantages and disadvantages of auctions: On the one hand, auctions clearly reduce informational rents, but on the other hand, in low- and middle-income countries there can be substantial equity trade-offs due to auctions if payments are differentiated by landowners.

Although an auction may be the theoretically preferred tool to help resolve the hidden-information problem, there is little experience in actually implementing these types of systems in the context of forest PES.[4] Ajayi et al. (2012) and Jack (2013) are two notable exceptions. These two papers describe an experiment in Malawi designed to elicit the exact willingness to accept a PES contract for afforestation. The experiment uses an undifferentiated payment scheme and shows that an auction mechanism can generate significant cost savings by identifying users with high private benefit-to-opportunity cost ratios for tree maintenance. This result suggests that participants hold significant private information that is otherwise hidden from the policy designer. As the authors are unable to make comparisons with alternate targeting schemes, assessing the scalability of such an approach is difficult. To our knowledge, no auctions have been conducted for avoided deforestation contracts. An alternative to auctions—selecting program recipients based on observable risk factors—also offers significant cost savings in simulations (Alix-Garcia et al. 2008, Mason & Plantinga 2013). Although this approach has not been directly applied in actual policy settings, the more recent cohorts of Mexico's program do use a deforestation risk measure as part of targeting (Sims et al. 2014).

An additional targeting complication is that environmental services are typically connected to trees in highly nonlinear ways. Calder (2002) presents a fascinating review of the relationship between scientific evidence and population perceptions of the connection between forest and environmental services. Calder shows that, although the popular perception is that forests are uniformly good for increasing water flows and for reducing erosion, evidence suggests that net increases in water depend very much on forest type, particularly for dry season flows, and that disturbing forests may either increase or decrease erosion, depending upon soil conditions. In the extreme, afforestation has been linked with decreased water flows in some settings (Farley et al. 2005). Clearly, biodiversity is not linearly increasing in forest area and likely depends upon the connection between contiguous areas of feasible habitat (McDonnell et al. 2002, Drechsler et al. 2007). Although the CRP literature dedicates significant energy to the design of contracting mechanisms that promote agglomeration of properties (Parkhurst et al. 2002, Nelson et al. 2008, Drechsler et al. 2010), we know of no work that investigates the design and implementation of agglomeration incentives in the forest PES context.

When should the payment be made? Theory suggests that the optimal payment scheme is to compensate the landholder at the very end of the contracted period (Salas & Roe 2012, Salas et al. 2012). This arrangement is often politically unfeasible, especially in low-income countries. Hence, in practice, most existing PES programs, whose contracts vary from 5 years (in Mexico) to 20 years (in Ecuador), tend to pay, on a yearly basis, at the end of each contract year. These contracts are contingent on preserved forest or standing trees, which are relatively easy to observe. In the case of re- or afforestation, the standing-tree contingency approaches the theoretical ideal, whereas for avoided deforestation it is more difficult to achieve the ideal due to the unobserved counterfactual baseline scenario.

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[4]In the vast majority of large-scale ecosystem services programs in developing countries, payments are made per hectare of standing forest (as in Costa Rica, Mexico, Colombia, and Ecuador) or per hectare of trees planted (as in China's SLCP). See appendix tables I and II in Pattanayak et al. (2010) for further examples.

In summary, the disconnect between the PES contract and the service, the difficulty of measuring the ecosystem services in question, and the fact that payment programs usually involve a single, flat payment per hectare (perhaps stratified by some ecosystem-type criterion) suggest that current PES programs are unlikely to approach cost-effectiveness. Likely because of these difficulties, we are not aware of any convincing work directly empirically measuring biodiversity value, water quality, and carbon sequestration benefits of existing forest PES, although an active literature simulates these effects in hypothetical programs (e.g., Caparrós et al. 2010 and papers cited therein, Sathaye et al. 2011).

3.2. Who Receives Payments?

Because forest cover and poverty are positively correlated at a global scale, policy makers have pushed to promote PES as an instrument for both environmental protection and poverty alleviation (e.g., Landell-Mills & Porras 2002, Turpie et al. 2008, Lipper et al. 2009, Rios & Pagiola 2011). The question of who receives payments from existing programs begins to speak to the distributional issues that have occupied a substantial part of the discussion surrounding avoided deforestation projects and their social impacts. In this section, we outline theory and evidence on program participation, leaving for Section 6 the analysis of the evidence on poverty alleviation. In a conceptual paper, Wunder (2008) highlights potential situations in which the poor are left out of PES programs. Wunder shows that participation is determined by owning enough environmentally strategic land, by trusting the purchaser, by having sufficient capacity to meet program monitoring/production requirements, and by having a low enough opportunity cost to make payments attractive. Clearly, some of these factors favor the poor and some the rich. Because environmental desirability can be positively or negatively correlated with the opportunity cost of participation, the question of who participates remains largely empirical.

What is the evidence on these characteristics from the empirical PES literature? A variety of cases show mixed empirical evidence on typical characteristics of PES participants and non-participants. Simulating the effects of a carbon sequestration program by using land use data from Costa Rica, Pfaff et al. (2007) note that the poor tend to hold more forested land but that this land is at relatively low risk of deforestation. Pfaff et al. highlight the possibility that, although carbon sequestration payments may be pro-poor, programs seeking to limit transaction costs by enrolling large areas of land may end up in the hands of the relatively rich living in poor areas, because the landholdings of the poor tend to be smaller. Data from Costa Rica's payments for hydrological services program show that large landholders with formal tenure are significantly more likely to put land into the program. In comparison, households with more family farm labor are less likely to participate (Zbinden & Lee 2005). Sims et al. (2014) find that in Mexico the distribution of municipal poverty across program participants tends to be generally indistinguishable from the distribution of municipal poverty across all forested areas. Sims et al. further find that the participation of communal, generally poor, landholding households has increased over time. Participants in silvopastoral PES programs in Colombia tend to be poorer but have larger farm sizes, whereas in Nicaragua participants have higher income per capita but have smaller farm sizes than nonparticipants (Rios & Pagiola 2011). A recent experiment in Uganda shows less take-up of an avoided deforestation PES scheme by credit-constrained individuals (Jayachandran 2013), a result explained by a model showing that the timing of PES payments (posteffort) makes PES unappealing to those needing immediate liquidity. In Malawi, a small tree-planting experiment showed participants to be slightly poorer than the national average, although the scale and structure of the experiment limit inference on participation constraints (Ajayi et al. 2012). Finally, in China, the land enrolled in the Grain for Green program has had a high probability of

contributing to soil erosion; enrolled plots have more than 15 degrees in slope and are relatively low yielding (Uchida et al. 2005). Participants and nonparticipants in the program appear to have similar levels of poverty (Gauvin et al. 2010).

3.3. Do Payments Actually Generate Environmental Services?

Do PES produce additional environmental services? Differentiating by program type, the evaluation literature shows positive results for reforestation and afforestation programs, although the number of studies is quite small. In comparison, the work on avoided deforestation programs has yielded mixed outcomes. Here we begin with the avoided deforestation literature and below move on to examples of afforestation.

Pattanayak et al. (2010) thoroughly discuss the question of environmental effectiveness of avoided deforestation programs. We do not wish to repeat their analysis here but rather use their conclusions as context for the few additional recent papers we discuss here, and we also suggest some additional conclusions to draw from this body of work. As Pattanayak et al. (2010) point out, few studies use rigorous impact evaluation methodologies to control for selection bias, and most of these studies are conducted in Costa Rica (Sánchez-Azofeifa et al. 2007, Arriagada et al. 2012, Pfaff et al. 2013b, Robalino & Pfaff 2013), although there is a recent increase in work in Mexico (Honey-Roses et al. 2011; Alix-Garcia et al. 2012a,b, 2013). The vast majority of these studies use some form of matching (Honey-Roses et al. 2011; Alix-Garcia et al. 2012b, 2013; Robalino & Pfaff 2013), sometimes combined with a difference-in-differences approach (Honey-Roses et al. 2011, Arriagada et al. 2012, Alix-Garcia et al. 2013) by using forest measures based upon satellite imagery.

Avoided deforestation tends to be modest when measured in terms of decreased deforestation but larger when reported in percentage change relative to controls, because the places in which avoided deforestation has been measured tend to have low deforestation rates. In Mexico's PSAH, a study of the 2004 cohort finds that the program reduces the percent area deforested by 1.19 percentage points, which amounts to an approximate decrease in deforestation of 50% relative to the mean percent deforested in matched control properties, which is 2.4% (Alix-Garcia et al. 2012b). A more recent analysis of all program cohorts from 2003 to 2009 in Mexico's PSAH finds a 40–50% decrease in the downward NDVI (Normalized Difference Vegetation Index) trend in the properties of PES participants relative to the control group (Alix-Garcia et al. 2013), although the downward trend in the controls is relatively small. Honey-Roses et al. (2011) show that the combination of PES payments and protected area designation preserved 3–16% of high-quality forest habitat for monarch butterflies in the Reserva de la Biosfera Mariposa Monarca in Mexico, relative to properties that did not benefit from these policies. The findings on cohorts enrolled in 1997 and 1998 in Costa Rica suggest zero to small avoided deforestation effects, with positive impacts coming from afforestation rather than from avoided deforestation (Pattanayak et al. 2010). A recent working paper on Costa Rica, however, finds larger and more significant impacts for land enrolled between 2000 and 2005, as well as spatial heterogeneity in impact, with larger effects estimated in areas that have lower slope and are closer to cities (Pfaff et al. 2013b). The authors cannot explain the increase in impact over time. The data in these cases have not been sufficient to analyze what occurs when properties finish their term in a program or disenroll early.

The existing studies on avoided deforestation programs have uniformly taken place in countries (e.g., Mexico and Costa Rica) with deforestation rates that decreased over the period of the program. Robalino & Pfaff (2013) estimate that in Costa Rica fewer than 0.4% of parcels enrolled in the program would have been deforested in the absence of payments. One lesson that can be extracted from this literature is that it is difficult for an avoided deforestation program to have an impact in the absence of deforestation risk. Other important lessons are as follows. First, analyses

from countries where deforestation risk is high but institutional strength is low will be essential for clarifying the conditions under which PES programs can be successful. Second, although deforestation is a decreasing problem in middle-income countries such as Mexico and Costa Rica, degradation remains an importance source of ecosystem services loss. There are significant remote sensing challenges to measuring degradation, particularly in tropical countries where the phenology presents considerable challenges to techniques developed for temperate forests (Blackman 2012). NDVI measurements do partially capture degradation, but clearly separating decreases in forest quality from decreases in forest area is still difficult. Improving these measures will remain an active research area.

In the afforestation literature, two recent field experiments by Jack (2013) and Jack et al. (2013) focus on measuring the impact of heterogeneous incentives to plant trees. The outcomes are simply measured as the number of trees alive at the end of the contract period. In Malawi, farmers who participate in an auction experiment to elicit their opportunity cost tend to have more live trees at the end of the contract than do farmers receiving a fixed-rate contract by lottery (Jack 2013). In preliminary work on a similar afforestation experiment in Zambia, an increase in a performance incentive results in positive impacts on the intensive margin in terms of the numbers of surviving trees and on the extensive margin with respect to the likelihood to participate in the program (Jack et al. 2013). Rios & Pagiola (2011) offer an alternative outcome methodology. They measure environmental benefits by using an index that aggregates over a variety of environmental characteristics of enrolled land, ranging from annual agricultural crops to primary and secondary forest, and find little impact of the PES program on this index.

In summary, considerable theoretical work on optimal contracts and targeting exists. Actual program rules, however, tend to lag behind the theoretical ideal. Despite this disconnect, there is evidence that afforestation and reforestation programs have had significant impacts, although additional work in more varied institutional and deforestation risk environments is necessary, and there is a large hold in the literature with respect to postprogram behavior by PES recipients. Recent experiments show significant potential for the use of auctions to increase program cost-effectiveness, but more information is required on the administrative costs of such programs before scaling up can be recommended without reservation. In terms of the effectiveness of avoided deforestation programs, however, the results are more mixed.

4. CHALLENGES TO FOREST PES IMPLEMENTATION

In addition to the challenges discussed in the above sections, many PES projects face two further complications: missing property rights and leakage. This section discusses the recent literature contributing to the study of these two problems.

4.1. Missing Property Rights

In the world of Coase, externality problems are resolved through the definition of property rights over the externality. In the world of forest PES, lack of secure tenure over the assets that produce the externality poses one of the biggest implementation challenges, particularly in developing countries. For avoided deforestation payments to achieve additionality, payments must be targeted to areas at risk of deforestation, and both theoretical work and empirical work suggest that insecure tenure can be a primary driver of deforestation. Robinson et al. (2011) provide an excellent review of the relationship between tenure and forest management. There is little formal theoretical work relating forest PES to property rights. One exception is the work of Barbier & Tesfaw (2013), who present a dynamic model that describes the interaction between forest PES and

customary land rights. They show that if the probability of eviction is decreased by participation in a PES program, individuals allocate more land to the project.

So far, the literature relating property rights to program effectiveness tends to be descriptive rather than causality establishing. Costa Rica's long-standing PES program, widely regarded as among the more successful, is predicated on well-defined property rights (Blackman & Woodward 2010, Arriagada et al. 2012). In Mexico, although much forested land is managed under communal tenure systems, the tenure rights of communities are mostly unchallenged, and the question of who should receive payments is easily resolved (Sims et al. 2014).

Distributional implications of tenure insecurity exist. Lack of formal title can be a significant barrier to participation by the poor (Wunder 2008), and large resource inflows may potentially encourage capture of forestland previously considered to be of low value. A revealing study in Uganda shows that differences in the definition of land tenure may lead to adverse impacts on women's land rights in the presence of PES payments that change the value of land (Bomuhangi et al. 2011). In Brazil, efforts to formalize tenure have proceeded hand in hand with REDD+ investments, and leveraging REDD to formalize titles shows promise (Duchelle et al. 2014). In Indonesia, in contrast, tenure conflicts and instability appear likely to undermine REDD+ effectiveness (Reso-sudarmo et al. 2014). A group of case studies from Brazil, Cameroon, Tanzania, Indonesia, and Vietnam reveal a variety of efforts to formalize tenure in anticipation of REDD but note a significant lack of coordination with national-level tenurization efforts (Sunderlin et al. 2014).

4.2. Spillovers

In the PES context, programs can generate unintended consequences, both negative and positive, through a variety of avenues. We discuss both types of spillovers in this section, noting first that most of the literature in this area focuses on the problem of the displacement of forest exploitation by program payments known as leakage. The possibility for leakage, or slippage, poses perhaps the most serious challenge to efforts to conserve or expand forests by using PES-type mechanisms (Jack et al. 2008, Plantinga & Richards 2008). Following the debate about slippage effects of the CRP (Wu 2000, 2005; Wu et al. 2001; Roberts & Bucholtz 2005, 2006), we define two types of slippage: substitution and macro price effects. Substitution slippage effect occurs when a land-owner who removes one parcel of land from production (by enrolling it in the PES program) shifts the planned production to another parcel within his landholdings. Well-functioning markets limit the possibility of substitution slippage (Roberts & Bucholtz 2005), but labor, credit, and land market rigidities in developing countries are more likely to lead to production displacement. The behavioral literature suggests that negative behaviors in reaction to exclusion from PES programs may generate additional leakage (Alpízar et al. 2013), but such additional leakage has not yet been observed in existing programs.

Macro price slippage occurs through general equilibrium effects if the removal of land from production increases the market price of land-intensive goods, thus changing production incentives on unenrolled land. Macro price slippage can occur on unenrolled areas within the same country or even globally if the market is integrated internationally (such as in the global carbon market). There are a variety of price slippage models, including those of Robalino (2007) and Murray et al. (2004), who nicely show that lower elasticities of forest product demand generate greater leakage. Rose & Sohngen (2011), using a general equilibrium approach, simulate the global impact of different combinations of afforestation and avoided deforestation policies to elucidate trade-offs and synergies between the two approaches. Rose & Sohngen find an afforestation-only scheme to be suboptimal, because price effects can actually increase deforestation in the short run, and conclude that incentivizing both activities simultaneously is necessary.

Much of the evidence regarding actual price leakage effects comes from non-PES policy interventions in the United States[5] and in Canada. Empirically, there is considerable work on the effects of limiting timber harvests, mostly from the United States. In a classic paper on price effects, Berck & Bentley (1997) find that the taking of 43% of the old-growth redwood tree inventory by the US government from private industry in the 1960s and 1970s increased the price of redwood timber by 46%, thus providing evidence of the price effect mechanism. Other US-based studies of leakage include the study of Wear & Murray (2004), who show how reduced public sales of timber increase private timber extraction in the United States and in Canada. General equilibrium simulations of potential forest carbon leakage constitute the largest portion of the price leakage literature. This literature suggests significant cross-border leakage from forest conservation (Murray et al. 2004, 2007; Gan & McCarl 2007; Meyfroidt & Lambin 2009).[6]

Almost no papers empirically measure substitution slippage resulting from forest PES. Two exceptions are Alix-Garcia et al. (2012b) and Arriagada et al. (2012). The former paper illustrates one possible slippage mechanism by using an agricultural household model with credit constraints. This paper shows evidence supporting the theory within Mexican common properties that applied to Mexico's PSAH in 2004. By matching between accepted applicants and rejected applicants, Alix-Garcia et al. (2012b) calculate that the substitution slippage effect reduces avoided de-forestation by approximately 4% on average, with larger impacts among poor communities and smaller impacts among the relatively wealthy. The same paper also finds suggestive evidence of output price leakage. Indirectly addressing slippage, Arriagada et al. (2012) conduct a whole-farm analysis of Costa Rica's PES program. By including the entire area owned by an individual farmer, rather than just the area enrolled in the program, the authors can implicitly measure the program impact net of leakage. Arriagada et al. (2012) find, during the PES contract of 8 years, a net increase of 11–17% in total farm forest cover on participating farmers relative to matched control farms. Finally, Yañez-Pagans (2014) provides evidence that in common property communities of Mexico, the distribution of PSAH payments as wages can reduce the time that households spend in other community service activities, including forest management activities that remain unpaid. This study cannot assess the impact of this shift on environmental outcomes, but it does suggest an avenue through which PES payments might spill over into other labor market decisions.

Although leakage is a major concern in the PES literature, there is little consensus on how to combat it. Using a two-period utility maximization model to characterize a static market equilibrium framework, Barua et al. (2012) show that complementing carbon payments with cash-crop taxes can be an effective method for discouraging deforestation. For substitution leakage, which involves landowners moving forest exploitation within their own properties, there is the theoretical possibility of controlling leakage with more complete contracts. There are clear practical barriers to implementing this setup, however, and programs with contracts over all land owned by participants do not yet exist.

Other work on spillovers suggests the possibility of positive leakage from forest conservation programs (Pfaff & Robalino 2012). Positive leakage can occur through the following two mechanisms. First, PES programs may discourage deforestation on lands adjacent to protected

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[5]There is considerable empirical work on slippage related to the CRP. Although this discussion is outside the scope of this review, among other effects, the CRP appears to increase production on neighboring lands (Fleming 2010), to shift nonconservation uses to later periods (Jacobson 2014), and to increase the value of farmland (Wu & Lin 2010).

[6]Although details of the carbon emissions leakage literature are outside the scope of this review, we refer to the paper of Karp (2012), who makes a useful point that is also applicable to the forest PES setting: Partial equilibrium models of the sort generally used to examine leakage are likely upward biased.

areas by raising expectations among landowners of future revenue from PES programs. Second, a given PES program may be a signal to private actors that the government will not be investing in infrastructure or industrial development in the broader region. This possibility may thus create spillover effects in neighboring land parcels by reducing the incentive for land clearing. Such a possibility is suggested by the empirics in Robalino & Pfaff (2012), who examine the impact of neighbor decisions on deforestation behavior. However, to date, there is little empirical evidence on such positive leakage in existing PES programs.

5. POVERTY ALLEVIATION AND PES

As is mentioned above, PES face considerable pressure to support both environmental protection and poverty alleviation goals (e.g., Landell-Mills & Porras 2002, Turpie et al. 2008, Lipper et al. 2009). Hence, there has been significant discussion in the literature of potential poverty effects of PES programs. Bulte et al. (2008), Lipper et al. (2009), and Palmer & Engel (2009) nicely review much of this work. Earlier work suggests that the poor may sometimes benefit from PES and that there may be trade-offs in targeting (Engel et al. 2008, Jack et al. 2008). However, robust conclusive evidence on either point is still lacking. In this section, we highlight the recent conceptual work and empirical work that have taken place since these reviews.

Section 3 above discusses whether the poor are among those receiving payments. Receiving payments, however, is quite different from whether a PES program actually aids in moving households out of poverty. Clearly, the former is a necessary condition for the latter, but much of the poverty/PES literature focuses on participation of the poor, rather than on changes in their outcomes as a result of an incentive program. Ollivier (2012) uses a general equilibrium framework to identify key tensions generated by transfers conditional on forest conservation. Under the assumption that farmers can substitute capital for land, and in the absence of labor market frictions, she shows that low transfers can increase agricultural productivity, and thus raise welfare, by raising the capital-to-land ratio. At larger transfer levels, however, this ratio becomes too high, thus decreasing returns to agriculture. In the case in which the external transfer does not fully compensate for this decrease, welfare can be reduced by the transfer.

Zilberman et al. (2008) present a useful microeconomic framework for understanding the potential impacts of both land diversion and working-land programs on PES sellers. Using a separable household model of decision making (in which households vary in farm size, environmental benefits of their landholdings, and wealth), Zilberman et al. show that, in the case of land diversion programs, such as avoided deforestation PES programs, the poor landholders are most likely to benefit if the main impacts of the program are through increased agricultural rents, whereas wage and price effects are minimal. In the same setting, landless rural poor may benefit if PES leads to higher labor demand. In contrast, the landless lose if payments increase local food prices. Overall, however, as in the case of the working-land programs, the increase in labor demand may lead to poverty alleviation.

On the empirical side, applied work on poverty alleviation and environmental effects exists only for China and Mexico. China's SLCP, which pays for reforestation, does not appear to have major trade-offs between environmental goals and development goals (Xu et al. 2006, Uchida et al. 2009, Gauvin et al. 2010). Uchida et al. (2007), using matching between participants and nonparticipants, find SLCP participants to have greater livestock assets after the program relative to nonparticipants. This paper does not assess whether there are trade-offs between these asset gains and program environmental effectiveness. More recently, an analysis of Mexico's PSAH on accepted and rejected applicants reveals very interesting and significant trade-offs between targeting on poverty alleviation and targeting on environmental effectiveness (Alix-Garcia et al.

2013). In particular, using matching and panel data analysis, Alix-Garcia et al. (2013) find that the environmental impact is highest where poverty is low but that poverty alleviation is highest where risk of deforestation is low. On average, the wealth effects are small. These findings demonstrate that the claim that PES programs can both generate inexpensive carbon sequestration and alleviate poverty is not generalizable and that the underlying correlation between poverty and deforestation risk determines the ability of a PES policy to achieve the dual objectives of poverty alleviation and environmental conservation.

Although there is scant evidence of immediate poverty alleviation resulting from PES payments, work suggests potentially positive long-term effects. Uchida et al. (2009) use panel data on households participating in China's national PES program (the SLCP) to examine labor response to payments. The study uses a panel of data on participants and nonparticipants interviewed from the same village and uses a difference-in-differences approach to evaluate changes in off-farm labor supply.[7] The authors find that the program increased off-farm labor participation for participating households and that this impact was larger for households that had fewer liquid assets prior to the program. Under the assumption that off-farm labor eventually leads to poverty alleviation, this paper reveals a potential indirect source of poverty alleviation through PES. The study further provides indirect evidence that the SLCP does not generate sufficient local labor demand (or raise wages enough) to keep households on farm. Alix-Garcia et al. (2012a, 2013) show that households participating in Mexico's PSAH are significantly more likely than nonparticipants to have children between the ages of 15 and 17 in school. A possible future payoff to the household from higher education of its children thus offers another potential avenue for longer-term effects of PES payments on recipient households.

In sum, there is little evidence that PES is harmful to poor participants but also little case for promoting it as another antipoverty program. The only two cases studied rigorously—China and Mexico—show potential long-term investments enabled by forest PES payments, but little short-term increase in assets. Studies from poorer countries with different relationships between land use and poverty may yield different results, and the new PES programs in Africa and Asia provide an opportunity for research to help answer these questions.

6. THE STATE OF THE LITERATURE AND THE FUTURE

This article reviews the recent theory and evidence on the demand, supply, and indirect effects of forest PES programs. This work has taught us a considerable amount about how things should be: Cost-effective contracts pay the opportunity cost to land with highest expected net benefits, payments should be made at the end of contracting periods, and leakage may occur through various channels. We have also learned a bit about how things are: Auctions can be used (on a small scale) to elicit hidden information regarding opportunity costs; the poor participate in PES programs in many settings; and existing programs of payments for hydrological services in Latin America have been moderately effective at reducing deforestation, have not been particularly effective at alleviating poverty, and are probably not as cost-effective as they could be.

This review also highlights the need for more work to understand how such payment programs might function in weaker institutional settings, in particular, in places where land tenure is not well established. This review also shows that the implementation of forest PES contracts would benefit

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[7]Although identification in the Uchida et al. (2009) study is imperfect (because whether the parallel trends assumption is satisfied is not clear), the authors conduct a series of robustness checks by using matching to help reduce the bias due to differential time trends.

from attempting to link services more closely to contracts, particularly in the contexts of the concern for agglomeration in the provision of hydrological and biodiversity services and information regarding what happens when contracts end. Conspicuously absent from the present literature are rigorous analyses of growing players on the global conservation scene—NGOs and individuals purchasing environmental services, sometimes on a significant scale. We also note the disconnect between the outcomes measured in empirical work—forest—and the actual environmental services of interest, with the caveat that measurement of the former presents challenges significant enough to undermine the effectiveness of large-scale antideforestation schemes. Our tour through this literature also shows that much remains to be done to shed light on the interaction of PES payment programs with the local economy and, eventually, with the global economy, in terms of both the measurement of program leakage and proposals for reasonable policies to combat it.

Finally, PES programs impact both the owners and the purchasers of these services, and the information currently available does little to help us quantify the welfare effects of these new relationships. The spread of REDD+ projects across the globe offers significant opportunities to explore these questions in the years to come.

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