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Edited by Gregory K. Ingram and Yu-Hung Hong

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Changing Land Uses in Forestry and Agriculture Through Payments for Environmental Services

Sven Wunder and Jan Börner

Forests are key global environmental assets, as carbon sinks to mitigate climate change, as main reservoirs of biodiversity, or through their influence on macroregional water balances. Agriculture provides food and energy, but it is often also associated with considerable negative environmental externalities. Changes in agricultural land use strategies and production technologies can potentially trigger large (positive or negative) environmental off-farm impacts (McNeely and Scherr 2003).

In developed and developing countries alike, payments for environmental services (PES) have recently received much attention as potentially cost-effective and equitable means to yield environmental benefits (Ferraro and Kiss 2002) nested in forestry (Landell-Mills and Porras 2002; Pagiola, Bishop, and Landell-Mills 2002) and agriculture (FAO 2007; Ribaudo et al. 2010).[1] To date, there has been far more practical implementation of PES in forests than in agriculture. This chapter examines the bias in the implementation of PES and scrutinize the

We thank James Levitt, Leslie Lipper, Nanete Neves, Stefano Pagiola, and Susanne Scheierling for comments on earlier versions of this chapter.

1. How cost-effective and equitable PES schemes are, is subject to ongoing scientific scrutiny. There are preliminary indications that PES schemes are widely successful on both counts (Wunder, Engel, and Pagiola 2008), yet robust empirical assessments using modern evaluation techniques are, just as for other applied conservation instruments, still missing (Pattanayak, Wunder, and Ferraro 2010).

respective sectoral potentials for environmental benefits from land use, land use change, and forestry (LULUCF). It concentrates on developing countries, where most PES schemes with a climate change focus are being implemented.

PES and Land Use Change

PES schemes are voluntary and conditional cash or in-kind transfers from at least one buyer to at least one seller, aimed at increasing environmental services (ES) provision relative to a given baseline value (Wunder 2007). Although economic activities affect a wide range of ES, real-world PES schemes have almost exclusively focused on four externalities: carbon, biodiversity, water, and landscape beauty (Landell-Mills and Porras 2002).

PES as voluntary economic incentives are viable only when service users' willingness to pay (*WTP*) exceeds service providers' willingness to accept (*WTA*). Providers typically suffer opportunity costs—that is, they lose income by switching from their preferred land use plan to a privately second-best, but environmentally more benign one. Together with transaction costs (discussed later in this chapter), opportunity costs usually have to at least be covered by payments in order for providers to accept a deal. Conversely, the more benign land uses trigger service gains for users; these gains, after deducting the transaction costs, have to be larger than the payments to allow sufficient *WTP*:

(1) Incremental service value – User transaction costs => *WTP* > *WTA* <= Opportunity costs + Provider transaction costs

In this chapter, we are interested in exploring what types of land uses are being promoted in LULUCF-oriented PES schemes. Two fundamentally different approaches exist here: (1) reducing agricultural expansion and encouraging cropland retirement and forest conservation; and (2) changing agricultural technology and practices. Accordingly, Zilberman, Lipper, and McCarthy (2008) distinguished between "land-diversion schemes," where land is set aside for conservation, and "working-land schemes," which change agricultural production to achieve ES gains. Wunder (2007, 51) uses the terms "use-restricting" and "asset-building" PES. The first approach essentially compensates farmers for reducing or suspending environmentally problematic economic activities; the second aims to change the latter without scaling back economic output. In this chapter, we use the terms "use-restricting" and "use-modifying" PES schemes.

Conceptually, there is no full overlap between the forestry-agriculture and use-restricting–use-modifying PES distinctions (see table 11.1). According to global PES surveys (e.g., Landell-Mills and Porras 2002), however, perhaps as much as four-fifths of implemented PES schemes lie in the northwest corner of use-restricting forestry. As for agricultural PES, basically all the schemes are use-modifying (southeast corner), with the exception of the two borderline cases—retirement of agricultural land and introduction of agroforestry. Some forestry schemes also have been implemented in reforestation and forest certification; the

Table 11.1
Sectoral Origin Versus Usage Impacts of Landscape Interventions

Use-Restricting	Use-Modifying
Forest conservation, including REDD, biodiversity, water	Reduced-impact logging—forest certification
	Afforestation and reforestation (AR), including CDM
Agricultural land retirement	Agroforestry—silvipasture
	Improved agriculture (organic, no tillage, no burn, etc.)

white = forestry; gray = agriculture.

latter is counted by some analysts as PES (Wunder 2005) and by others as PES-like, because it works more indirectly through eco-premiums in product markets (Ferraro 2009).

Table 11.2 provides six examples of PES schemes worldwide. Government-financed schemes, such as the U.S. Conservation Reserve Program (CRP) and the PSA program in Costa Rica, are typically large, national-scale initiatives that embrace multiple services and political side objectives. In these programs, the government pays on behalf of service users. User-financed schemes, often involving single private-sector buyers (e.g., Profafor in Ecuador and Vittel in France), are small to medium size, buy just one or two services, and are more targeted in design (Wunder, Engel, and Pagiola 2008). The examples in table 11.2 include both use-restricting (Pimampiro, CRP, PSA) and use-modifying (RISEMP, Profafor)[2] schemes. The French water bottler Vittel's watershed scheme pays for both components (improved dairy farming and reduction of animal stocks).

With the notable exception of afforestation and reforestation, most PES schemes, especially in developing countries, are use-restricting. In the tropics-wide sample of watershed PES in Porras, Grieg-Gran, and Neves (2008), only a handful out of 50 ongoing schemes are purely use-modifying; in the Asian review by Huang et al. (2009), the ratio is 2 to 15; and in the African review by Ferraro (2009), there are none. The term "use-modifying" refers almost exclusively to industrialized countries' agri-environmental schemes, such as those in the EU (Baylis et al. 2008), the U.K. (Dobbs and Pretty 2008), and the United States (the Environmental Quality Incentives Program, or EQIP) (Baylis et al. 2008; Claassen, Cattaneo, and Johansson 2008).

Nevertheless, a perception of huge potentials for enhancing ES provision through agricultural change remains widespread. Niles et al. (2002) estimate the

2. RISEMP is the Regional Integrated Silvopastoral Approaches to Ecosystem Management Project.

Table 11.2
Six Examples of PES Schemes Worldwide

Program	Country	Service	Land Uses Paid For	Seller Agency	Buyer-Seller Scale	Extension (hectares)	Duration	Source
RISEMP	Colombia Costa Rica Nicaragua	Biodiversity Carbon	Silvipasture: UM	Global Environmental Facility (GEF)	International	3,500	2002–	Pagiola et al. (2005)
Pimampiro	Ecuador	Watershed	Conservation/ retirement: UR	Municipality	Local	496	2000–	Wunder and Albán (2008)
Conservation Reserve Program (CRP)	United States	Watershed Biodiversity Soil protection	Land retirement/ restoration: UR	Government	National	14.5 million	1985–	Claassen, Cattaneo, and Johansson (2008)
Profafor	Ecuador	Carbon	Reforestation/ afforestation: UM	Private	Selected regions	22,300	1993–	Wunder and Albán (2008)
PSA	Costa Rica	Carbon Watershed Biodiversity Landscape	Conservation: mostly UR	Government	National	270,000	1996–	Pagiola (2008)
Vittel	France	Watershed	Conservation/ low-impact agriculture: UR and UM	Private	Local	5,100	1993–	Perrot-Maître (2006)

Note: UR = use-restricting; UM = use-modifying.

potential for carbon emissions reductions through agricultural use modification in Latin America, Asia, and Africa at over 39 MtC (megatons of carbon) annually—higher than that for reforestation. As the globally largest water user, agriculture might make key contributions to improving water quantity and quality as well (CAWMA 2007). Biodiversity conservation and landscape beauty also could be enhanced through agricultural change (FAO 2007).

Why Do So Few PES Schemes Feature Agricultural Change?

Why might use-modifying agricultural change be at a disadvantage vis-à-vis predominantly forest- and tree-based PES models in the use-restricting sphere?

> *Hypothesis 1: Services provided.* Use-modifying agricultural change tends to produce fewer environmental services than does restricting use to secure the presence of trees and forests.

ES buyers usually prefer options that deliver more services. For many biodiversity conservation and recreational values, the presence of trees and natural habitat is vital. Standing forests represent a huge carbon stock, climate regulator, and biodiversity reservoir. With the exception of the carbon values produced by reforestation and afforestation, use-restricting conservation may thus be the most effective way to secure large amounts of ES, outperforming in particular agricultural use-modifying solutions.

> *Hypothesis 2: Provision costs and risks.* Opportunity costs and technological complexity in use-modifying PES are higher than in use-restricting PES, thus limiting the adoption of use-modifying schemes.

Competitiveness is not only about quantities delivered, but also about provision costs and complexities. For use-modifying measures, various improved cropping techniques privately pay for themselves through yield increases (Critchley 2009; Koohafkan and Stewart 2008). However, farmers are often too risk averse to adopt complex technologies requiring investments, maintenance, and training (Mercer 2004). In many developing countries, technological complexity may be off-putting to farmers who face constraints of capital, labor, and know-how and who lack supply for new required inputs or markets for new outputs. Thus, it is much simpler to delimit a forest area as a "no-go zone" or to set aside a marginal production area for natural regeneration. Use-restricting solutions usually entail positive but numerically small opportunity costs, and little or no investment is required, making them low-risk to adopt.

> *Hypothesis 3: Transaction costs.* In use-modifying PES, transaction costs tend to be higher than in use-restricting PES.

First, transaction costs can be high for some PES schemes, such as smallholder carbon programs (Cacho, Marshall, and Milne 2005). Population density is often higher in prime agricultural areas than in forest margins, where PES implementers can deal with a smaller number of payment recipients for any fixed-quantity area enrolled. Second, monitoring PES compliance can be cheaper in forested landscapes (e.g., using remote sensing) than in agricultural areas (e.g., monitoring active land management by making sure farmers are continuously using no-tillage farming, terracing, and mulching, then calculating how much carbon and other benefits these practices provide). One factor contrary to this hypothesis is the fact that land tenure conditions are usually more secure in consolidated agricultural areas than in forest frontiers. If frontier land tenure regularization has to precede PES, start-up transaction costs could become uncompetitive.

> *Hypothesis 4: Spillover effects.* In those exceptional cases where improved agricultural practices are very attractive to farmers, their upscaled adoption would tend to create more negative spillover effects than in the case of use-restricting interventions.

Imagine that, contrary to hypothesis 2, agricultural innovations are successfully introduced on a large scale using PES incentives that function as a transitory adoption subsidy. Conservationists often argue that this would create positive spillover effects on the environment. For example, by using an intensified cropping technique, farmers would need less land to match their current incomes and thus would decide to clear less land (Davidson et al. 2008). The underlying Chayanovian assumption is that farmers would then produce only what they need to survive, or at least up to some target-revenue production, and then prioritize leisure. More often than not, however, farmers actively expand the new, more profitable intensified production method into new areas, including at the cost of biodiversity and carbon-rich primary forests (Angelsen and Kaimowitz 2001). On the one hand, agricultural use-modifying PES schemes could thus become victims of their own success, at the cost of reduced environmental efficiency. On the other hand, use-restricting schemes could shift pressures to other areas (leakage). Both approaches could also have negative impacts on other, nontargeted services.

Data and Case Description

For the first hypothesis, we reviewed the literature on the ES provision potential of conservation alternatives, comparing approaches that involve use-restricting and use-modifying ES options. The Intergovernmental Panel on Climate Change's report on mitigation (IPCC 2007) and the Food and Agriculture Organization's report on the state of food and agriculture (FAO 2007) represent the most important sources for this debate. While some globally based empirical evidence exists for hypothesis 1, for opportunity and transaction costs (hypotheses 2 and 3) and spillover effects (hypothesis 4), we believe case study material is necessary to make meaningful quantitative tests. We thus draw on data about the Brazilian Amazon from the ASB Partnership for the Tropical Forest Margins (http://www.asb.cgiar.org) and the Secondary Forests and Fallow Vegetation in the Eastern Amazon Region—Function and Management Project of the SHIFT-Capoeira program (http://www.shift-capoeira.uni-bonn.de).

The Amazon region holds some advantages in comparing the promising opportunities of use-restricting and use-modifying PES. Smith et al. (2007, 2008) underline the high agricultural mitigation potential of South America, including the Amazon. Its potential for use-modifying interventions has been extensively documented (Börner, Mendoza, and Vosti 2007; Denich, Kanashiro, and Vlek 1999; Vosti et al. 2001). At the same time, the Amazon's vast forests hold great potential for reducing emissions from deforestation and forest degradation (REDD), as well as for achieving biodiversity and hydrological cobenefits (Börner et al. 2010; Grieg-Gran 2006; Turner et al. 2007). Figure 11.1 shows the selected case study sites in the Brazilian Amazon.

The western sites represent typical agricultural frontier settings, with dynamic land use patterns and large tracks of primary forests left on smallholdings. In the land reform settlements of Acrelândia and Theobroma, 228 randomly selected households were interviewed (Vosti, Witcover, and Carpentier 2002).

In contrast, the eastern Amazon is a consolidated agricultural zone in relative proximity to the large city of Belém. In this region, practically no primary forests are left, and a relatively stable landscape mosaic of agricultural crops, pastures, fallows, and secondary forests has evolved. In the districts of Castanhal, Igarapé-Açu, and Bragança, farm-level surveys included 271 randomly selected households.

Service Provision

Let's first take a look at hypothesis 1, comparing the quantity of three services (carbon mitigation, biodiversity and landscape beauty, and watershed protection) likely provided by the two PES types.

CARBON MITIGATION

In a report to the United Nations Framework Convention on Climate Change (UNFCCC), Verchot (2007) identified agroforestry and grassland management, each with an annual global mitigation potential of more than 2,000 $MtCO_2eq$ per year, as prime agricultural mitigation options. In absolute terms, the mitigation potential of changes in agricultural practices is impressive. However, according to Sohngen and Sedjo (2006), it is still almost three times lower than the annual mitigation potential of reducing deforestation to almost zero. IPCC (2007) includes detailed estimates for agriculture's mitigation potential in different climate zones. Table 11.3 summarizes these estimates and compares them with land retirement. A quick look reveals that the agricultural management options provide

Figure 11.1
Case Study Sites in the Brazilian Amazon

Table 11.3
Average Annual Carbon Dioxide Emission Reduction Potential for Selected LULUCF Options

Activity	Practice	Dry	Moist
		(tCO_2/ha/yr)	
Land retirement		1.61	3.04
Manure/ biosolids	Management/application	1.54	2.79
Croplands	Water management	1.14	1.14
	Tillage and residue management	0.33	0.70
	Agroforestry	0.33	0.70
	Agronomy (e.g., improved varieties)	0.29	0.88
	Nutrient management	0.26	0.55
Grasslands	Grazing, fertilization, fire	0.11	0.81
Ecosystem restoration	Organic soils (e.g., wetlands)[a]	73.33	73.33
	Degraded lands	3.45	3.45
Forestry		1–35	

[a] Some of these potential gains are offset by increased methane emissions from restored organic soils.
Source: IPCC (2007).

returns in the 0–3 tCO_2/ha/yr range. This is much lower than the potential for use-restricting options in forestry and soil restoration. Restoration requires agricultural retirement, but captures 73.33 tCO_2/ha/yr in organic soils—that is, more than 20 times the emission reduction potential of the best agricultural option. In heavily degraded or otherwise poor soils, natural carbon accumulation rates after land retirement can be significantly lower (Feldpausch et al. 2004). Reforestation and afforestation sequester 1–35 tCO_2/ha/yr, depending on tree species and site characteristics. Related soil carbon changes can be positive (1–1.5 tCO_2/ha/yr) or negative (up to –2.2 tCO_2/ha/yr) (IPCC 2007). A caveat is that agroforestry, with a low value in table 11.3, has been shown in other studies to accumulate up to 18 tCO_2/ha/yr in aboveground vegetation (Mutuo et al. 2005).

Although emissions reductions from avoided deforestation are not directly comparable with the per-year figures in table 11.3, high estimate ranges of 350–900 tCO_2/ha of prevented forest loss put both land retirement and agricultural change mitigation options into further perspective. Even if forests were replaced with highly efficient energy crops such as sugarcane, “neutralization” of emissions from preceding tropical rain forest conversion would take decades (Gibbs et al. 2008). In conclusion, use-restricting options such as avoided deforestation, reforestation, and land retirement tend to have the upper hand in carbon services delivered, both in absolute and, even more so, in relative per-hectare terms.

BIODIVERSITY AND LANDSCAPE BEAUTY

Beyond biodiversity's overarching protection of the integrity of ecosystems, humans use biodiversity for four different purposes, ranked here according to their approximate ability to generate funding:

1. Existence values (the nonuse pleasure of knowing about the survival of other species).
2. Option values (preserving, for example, landraces for cropping or genes for pharmaceutical research).
3. Landscape values (the aesthetic pleasure of using biologically diverse ecosystems).[3]
4. Tangible production-enhancing values (e.g., pollination or pest control services for agriculture).

The heterogeneity and scale dependence of biodiversity's environmental services make a quantitative side-by-side comparison of use-restricting versus use-modifying interventions much more complicated than for carbon, where the service can be reduced to a single unit. However, at least species diversity can be assessed. Comparing tropical rain forests in Africa, Asia, and Latin America, Gillison (2005) calculated land cover–specific biodiversity indexes (see table 11.4). Species numbers were clearly higher on conserved or retired land (near-natural or disturbed forests and fallows) than on land under agricultural uses (agroforestry, crops, or pastures). Thus, agricultural systems are generally poorer in plant diversity than corresponding near-natural areas, although their internal differences can be very large.

Obviously, some caveats are needed. First, specific species diversity (as for plants) is not always a good indicator of biodiversity in a multidimensional sense. For instance, a limited extent of landscape-level agricultural intervention can increase ecosystem diversity, and in some secondary forests biodiversity is higher than in primary ones (although endemism is usually highest in the latter). Second, existence and landscape values are subjective; hence the links to land use are highly variable. For example, Costa Rica's national PES system pays for natural forest protection, including for its aesthetic values. Yet in central European mountain areas (e.g., the Alps and the Black Forest), farmers are conversely paid not to abandon economically marginal agriculture, and thus to impede natural forest regrowth, in order to safeguard a forest-farm landscape mosaic that is particularly valued by tourists.

FAO (2007) emphasizes the potential of both strict conservation and agricultural modification to improve biodiversity services. Best agricultural practices, such as on-farm habitat enhancement, pollution control, and improved natural resource management, can lead to successful on-farm biodiversity enhancement.

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3. Landscape values can also be treated as a separate service.

Table 11.4
Plant Species Counts in Different Land Cover Types

Land Use/Cover Type	Number of Plant Species	Number of Plant Functional Types
Intact and moderately disturbed rain forest	80–102	35–44
Old secondary forest	50–111	24–43
Fallow	54–82	32–43
Agroforestry/tree plantation	15–66	13–33
Annual crop	14–51	12–37
Planted pasture	7–18	5–10

Source: Based on Gillison (2005).

On-farm biodiversity (e.g., of agricultural crop species) can provide external value to research as a preselected genetic pool for the development of new crop species (Pimentel et al. 1992). Some conceptual work exists to set up PES schemes that would pay farmers for preserving landraces (Narloch, Pascual, and Drucker 2009). A recent PES project used funds from the Global Environment Facility (GEF) to pay for farmers' adoption of silvipastoral systems and was successful in enhancing carbon and biodiversity benefits in the respective landscapes (Pagiola et al. 2005).

We consider agrobiodiversity interventions particularly valid for degraded landscapes, where the reintroduction of biodiversity can result in large productivity gains, as is convincingly shown by McNeely and Scherr (2003). However, the case for agrobiodiversity is usually much less clear in landscapes where natural vegetation still abounds, but is threatened by gradual conversion. In those cases, the payoffs for preserving pristine or near-natural systems are normally superior, in part because of the significant time lags (at best) and irreversibilities (at worst) in regenerating lost biodiversity.

In addition, many agrobiodiversity-related services generate benefits primarily at the farm level, such as enhancing soil fertility (Jackson et al. 2007). In the absence of genuine externalities, this disqualifies PES: there is no reason society should pay farmers to enhance their soil fertility, which only the farmers profit from. Other activity-modifying services, such as beekeeping, may not benefit any but the closest neighbors (Losey and Vaughan 2006). The low overall willingness to pay for biodiversity conservation has so far been almost exclusively restricted to larger-scale services, such as existence and option values. Almost no PES schemes exist for local-range services, such as landscape- and production-enhancing values.

Thus, on-farm biodiversity management may be considered an important complement to activity-restricting biodiversity conservation. However, biodiversity existence values (e.g., vis-à-vis charismatic species such as large terrestrial mammals) are currently more able to tap into beneficiaries' willingness to pay through private donations. In most cases, it appears that agriculture is at a clear disadvantage in providing such services, at least compared with the conservation or restoration of natural habitats. To the extent that biodiversity's existence and option values are concerned, the protection of natural ecosystems appears to be the most direct and quantitatively rewarding means to provide biodiversity-related ES.

WATERSHED PROTECTION

Watershed protection typically responds to human demands for services such as water quality and quantity, dry-season stability of flow, macro-regional regulation, and landslide and storm flow protection (Darghouth et al. 2008). Watershed services are similar to biodiversity in the sense that benefits are highly space specific. If water services are to be protected, the choice for downstream service users often boils down to a single watershed (or at best a handful of watersheds), and PES initiatives then have to work with the people and the vegetation covers that happen to be there. This may also limit service users' choices between use-restricting and use-modifying PES approaches.

The links between vegetation cover and watershed services are often ambiguous. For forests, there is a consensus that standing natural forests are good at providing clean and relatively stable water flows, yet the impact of tree cover on dry-season flows and storm flow protection is highly site specific, asymmetric between forest conservation and reforestation, and sometimes disputed among hydrologists. Soil conditions may eventually matter more than vegetation cover (Bruijnzeel 2004). In other words, it is often not clear which use-modifying interventions will produce the desired results. From a precautionary point of view, nothing is more logical than relying on use restriction when natural vegetation is still in place and has provided the desired services in the past. This is a strong advantage for use-restricting over use-modifying watershed approaches when both approaches are possible. Some services, such as macro-regional hydrological regulation, are provided only by standing forests, such as in South America. No equivalent agricultural use-modifying intervention could ever be as important as conserving a large share of the Amazon forest biome.

Obviously, if anthropogenic landscape interventions and/or soil degradation processes have already advanced significantly, or if opportunity costs for the conservation of forest islands in deforested landscapes are simply too high, the emphasis may necessarily have to be on use-modifying approaches. In those cases, improved agriculture can certainly be an effective solution (Koohafkan and Stewart 2008) and has sometimes also been combined with PES approaches (Porras, Grieg-Gran, and Neves 2008). Often the focus is on reducing erosion and sedimentation to safeguard water quality. In areas with variable soil and topographic conditions, some 5–20 percent of the watershed may be responsible for almost all of the erosion potential (Quintero, Wunder, and Estrada 2009). If those erosion-prone areas happen to be under agriculture, then use-modifying interventions become essential for service provision. Another frequent intervention is the natural or aided reforestation of degraded riverine areas. These scenarios underline the role of spatial specificity: in few cases do implementers have the option to choose freely between either use-restricting or use-modifying approaches. More realistically, in some watersheds where only use-modifying PES schemes are potentially possible, a PES scheme may never see the light of day. This occurs when either the service impacts are not fully known or the approaches are too costly or complex to implement.

Opportunity Costs and Technological Complexity

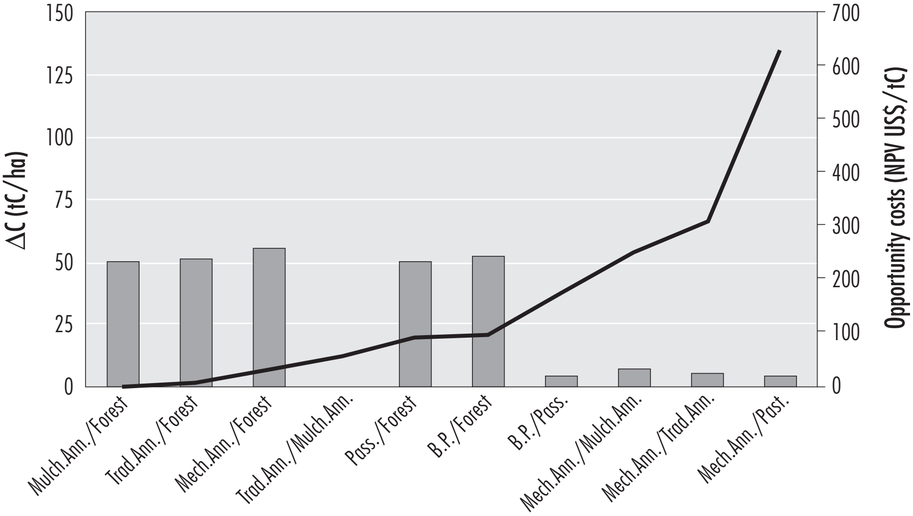
In the previous section, we concluded that use-restricting interventions in most cases provide higher quantities of demanded environmental services, although use-modifying approaches frequently play important complementary roles. It is also important to think about what these services cost and how complex they are to implement (hypothesis 2). To provide ES via land retirement or reduced agricultural expansion, farmers can simply suspend or reduce productive activities. For comparison, ES provision through changes in production technologies and practices requires the adoption of technological innovations, which may be a more complex process. In both cases, farmers forgo incomes from the activities they would have realized without PES, minus their current productive incomes under PES (i.e., their opportunity costs). Is there any inherent difference in opportunity costs across the two types of PES?

Figures 11.2 and 11.3 compare opportunity costs and carbon service returns from alternative land use options and technologies in the eastern and western Amazon study sites, respectively. Opportunity costs are illustrated by pairs of potential "business as usual" and "desired" land use options. For example, the pair "pasture creation" and "forest conservation" denotes the loss in private returns (net present value, or NPV) from abandoning pasture creation in favor of forest conservation. The horizontal axis shows pairs of more versus less profitable land use options (see table 11.5 for descriptions of these options) ranked by increasing opportunity costs, as measured on the right-hand vertical axis and the point-connecting line. The left-hand vertical axis and the bars denote corresponding carbon increments for these shifts. High bars and low-lying points combined make for cost-effective carbon-friendly land use shifts.

The results shown in figures 11.2 and 11.3 initially reconfirm our global LULUCF findings: the per-hectare carbon mitigation potential is consistently larger for use-restricting options (in this case, REDD) than for use-modifying ones. Basically, all the high bars have "forest" in their denominators, because they provide the largest carbon returns. The difference is clearest in the eastern Amazon cases (figure 11.2), although absolute carbon stocks in previously degraded areas are lower there than in the western Amazon (figure 11.3). In the latter, shifting from annual crops to agroforestry provides similar per-hectare carbon increments (around 50 tC/ha [tons of carbon per hectare]) as REDD in the eastern Amazon's more degraded forest sites.

If forest conservation provides the highest service "bang," what about the

Figure 11.2
Opportunity Costs and Biomass Carbon for Alternative Land Use and Technology Options in Eastern Amazon Sites (2003 US$/ha)

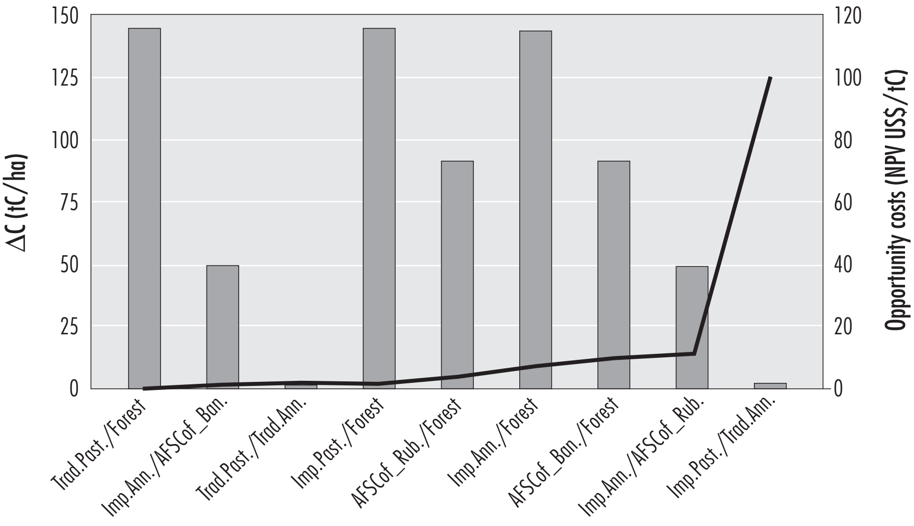


Note: See table 11.5 for descriptions of the abbreviations used here.
Source: Adapted from Börner, Mendoza, and Vosti (2007).

"buck"? Cost-wise, secondary forest–based REDD also is cheapest in the eastern Amazon, where only the shift from slash-and-burn to mulching is competitive among the clearly REDD-dominated low-cost options on the left-hand side. The opportunity costs of most use-modifying mitigation options are prohibitively high (>US$170/tC). Since they also come with low absolute carbon gains, they appear out of range for cost-efficient PES incentives. In the western Amazon, the picture is more mixed, given that use-modifying conversions to agroforestry and to annual crops are also relatively low-cost. But to match the absolute mitigation potential, use modification would have to occur on areas 3–30 times larger than those for most use-restricting, forest-conserving options.

Moreover, use modification for carbon mitigation is usually not only less cost-effective, but it also tends to be technologically more complex than use restriction, which in principle just requires farmers to do nothing on contracted land. In contrast, use-modifying PES schemes require providers to adopt new technologies and land use practices. Both the western and eastern Amazon studies identified technological alternatives where per-hectare net returns were far above those from traditional practices. Theoretically, farmers would face income gains (negative opportunity costs) by adopting the former alternatives, even without PES. Yet adoption of most of these alternatives has remained negligible. Under-

Figure 11.3
Opportunity Costs and Biomass Carbon for Alternative Land Use and Technology Options in Western Amazon Sites (2003 US$/ha)



Note: See table 11.5 for descriptions of the abbreviations used here.
Source: Adapted from Vosti et al. (2001).

standing this apparent paradox is essential in determining to what extent PES could have a positive effect in alleviating adoption constraints.

Many observers have blamed past top-down extension approaches for poor adoption rates, calling for a new paradigm in agricultural research and extension that would put "farmers first" on the rural development agenda (Chambers, Pacey, and Thrupp 1989). But what does a favorable adoption climate look like? Why do some villages and households adopt technologies more easily than others? In a review of the adoption literature, Lee (2005) stresses that, apart from agronomic constraints, family labor and labor market constraints often limit farmers' ability to adopt labor-intensive technologies, such as integrated pest management. Pattanayak et al. (2003) confirmed this notion for the adoption of agroforestry systems. Sustainable agricultural technologies are often technologically complex and thus demand more management skills than conventional practices.

While limited information about the performance of technology may increase the perceived risks of adoption, some technological alternatives are indeed more risky than others. In the eastern Amazon, for example, farmers are well acquainted with the advantages of chemical fertilizers, but apply them only to certain crops. On-farm trials demonstrated that fertilizers could almost double expected cassava yields (Kato et al. 1999), but only 3 percent of a sample of 270

Table 11.5
Activity Legend for Figures 11.2 and 11.3

Abbreviation	Description
AFSCof_Ban.	Agroforestry system integrating coffee and bandarra (*Schizolobium amazonicum*)
AFSCof_Rub.	Agroforestry system integrating coffee and rubber trees
B.P.	Permanent black pepper plantation
Forest	Forest conservation, as under REDD
Imp.Ann.	Annual staple crop production using leguminous-enriched fallows
Imp.Past.	Improved pasture establishment
Mech.Ann.	Permanent annual staple crops with mechanical land preparation
Mulch.Ann.	Mechanized chop-and-mulch fallow-based annual staple crops
Pass.	Semipermanent passion fruit plantation
Trad.Ann.	Traditional fallow-based annual staple crop production using slash-and-burn land preparation
Trad.Past.	Traditional pasture establishment

farm households applied it. Farmers who used fertilizers in cassava production were almost exclusively large-scale producers with secure commercialization channels, as opposed to smallholders depending on highly fluctuating local wholesale prices. Farm-household-level mathematical programming confirmed that fertilizer use decisions in the eastern Amazon can be explained by aversion to price and yield risks (Börner et al. 2007). Moreover, Börner (2009) and Hoch, Pokorny, and de Jong (2009) identified the risk of accidental fire as a potentially important smallholder adoption barrier for agroforestry and reforestation in areas where slash-and-burn farming dominates.

In conclusion, it appears that high opportunity costs, low cost-effectiveness, technological complexity, and high risks can frequently put use-modifying PES opportunities at a disadvantage. In some cases, however, PES incentives can be effective in helping farmers to overcome adoption hurdles, as shown for the aforementioned GEF-financed PES program for silvipastoral adoption (Pagiola et al. 2005). In general, however, hypothesis 2 was predominantly confirmed: more often than not, use-restricting approaches are cheaper, less risky, simpler, and thus more attractive for ES buyers and providers alike.

Transaction Costs

In the context of PES schemes, transaction costs can be defined as all costs that are not payments proper (Wunder, Engel, and Pagiola 2008). Transaction costs per unit of ES generally tend to decline with total ES volume provided, which represents a key entry barrier for small-scale PES schemes (Cacho, Marshall, and Milne 2005). Table 11.6 lists commonly cited transaction cost categories and their likely determinants.

Empirical data on PES transaction costs are scarce, partly because most schemes are still fairly new, but the global review of Wunder, Engel, and Pagiola (2008) found that many schemes incur high start-up costs for establishing links between land uses and associated ES, reference scenarios, and payment negotiations. Start-up costs varied between US$76 and US$4,800 per hectare and were often high where the ES–land use link was ambiguous (e.g., in watershed PES). Recurrent annual transaction costs tended to be at least one order of magnitude below start-up costs. No systematic variation between use-restricting and use-modifying transaction costs were found in that study.

In a feasibility analysis of payments for avoided deforestation, Börner and Wunder (2008) estimated potential transaction costs of a REDD program on more than four million hectares of land in the Brazilian state of Mato Grosso. Estimates for transaction cost categories 3–5 in table 11.6 were US$7.50/ha for start-up and US$4.50/ha/yr for recurrent costs. These large-scale values compare rather favorably with most estimates in Wunder, Engel, and Pagiola (2008), pointing to potential economies of scale in curbing PES transaction costs. For comparison, in the Juma Sustainable Development Reserve, a use-restricting REDD initiative under the Bolsa Floresta program in Amazonas, Brazil, nonpayment costs reached

Table 11.6
Transaction Cost Categories and Determinants

	Transaction Cost Category	Determinants
Start-up costs	1. Information and procurement	• Knowledge about ES markets and entry requirements (–)
	2. Scheme design and negotiation	• Number of ES buyers and providers (+) • Knowledge about ES–land use links (–) • Tenure security (–) • Level of trust between stakeholders (–) • Accessibility (–)
	3. Verification and certification (approval)	• Complexity of ES–land use links (+)
Recurrent costs	4. Implementation	• Institutional capacity (–) • Local infrastructure/accessibility (–)
	5. Monitoring	• Easily observable ES provider compliance (–) • ES provider's land has clear spatial limits (–)
	6. Enforcement and protection	• Conditionality of payments (–) • Tenure security (–) • Accessibility (–)

Note: Direction of impact: (+) increases and (–) reduces transaction costs per unit of ES.
Source: Adapted from Milne (1999).

almost 60 percent of the total budget, which corresponds to somewhere between US$11.90/ha (simulated high deforestation baseline) and US$2,859/ha (historical low deforestation trend) (IIED 2009). Yet almost half of these costs (e.g., for community support and law enforcement) were predominantly added-on, non-PES program components. Use-modifying schemes also can be expensive. The aforementioned Vittel watershed program cost around US$600/ha/yr, due to heavy biophysical research components and customized implementation of an extremely high-value service (Perrot-Maître 2006; Wunder, Engel, and Pagiola 2008).

Program scale, service focus, and design clearly matter in PES transaction costs for land-diversion and agricultural-change PES alike. But site characteristics often vary between the two, with different cost factors pulling in opposite directions. Land-diversion schemes usually target forest margins with low population densities, and fewer potential ES providers reduce negotiation costs (table 11.6, category 2). Conversely, forest margins are often expensive to access and exhibit insecure land tenure conditions, thus potentially raising the costs of scheme design, monitoring, and enforcement (table 11.6, categories 2, 5, and 6). In an attempt to create an Indonesian conservation concession PES, implementation could not proceed before a long-standing land tenure dispute between communities had been settled (Wunder et al. 2008). Use-modifying PES schemes for agricultural change are often implemented in consolidated agricultural frontiers where transport costs are lower and tenure is clarified, thus reducing transaction costs. Our two Amazon study areas mostly exhibit similar conditions. According to the Instituto Brasileiro de Geografia e Estatistica (IBGE) district database (http://www.ibge.gov.br/cidadesat/default.php), population density in the western Amazon sites (agricultural frontier setting) is 5–8 persons per km^2; density in the eastern sites (consolidated agricultural zone) is 45–157 persons per km^2. Rural transportation infrastructure (i.e., road access and number of vehicles) is much better developed in the eastern Amazon, yet tenure conditions are highly variable.

While context-specific transaction cost impacts tend to be mixed for the two PES approaches, a more clear-cut case can be made for monitoring costs. Use-modifying technological and land use alternatives normally have to be monitored through costly field visits. Even very sophisticated remote-sensing techniques typically fail to detect, for instance, the type of land preparation technology used for agricultural crops. If use modification increases the presence of trees in land use systems, such as in agroforestry, low-cost remote-sensing monitoring can be employed in conjunction with some ground truthing. Nonetheless, field verification needs may still be far above the requirements for monitoring activity-restricting REDD programs, where much verification can be accomplished through remote sensing.

It seems that hypothesis 3 is less powerful than hypotheses 1 and 2 in helping to explain why use-restricting PES schemes outnumber use-modifying initiatives. In addition to local context, transaction costs are very much influenced by project design, ES type, and ES buyers' desire to monitor provision closely. Monitoring may become a serious problem when the adoption of alternative land use options is costly to observe with accuracy. However, we cannot generalize about whether use-modifying PES schemes trigger higher or lower aggregate transaction costs than use-restricting schemes.

Spillover Effects

PES interventions affect farmers' supply and demand decisions, which could create environmental spillover effects in three ways: (1) losing other services; (2) overshooting the adoption scale; and (3) leakage. First, the change promoted through PES could produce unintended environmental externalities—for example, when carbon forestry projects introduce fast-growing monoculture plantations that negatively affect biodiversity and groundwater reserves (Jackson et al. 2005). Second, use-modifying PES could make promoted uses so profitable that they are expanded onto previously unused land, such as natural forests. Third, by affecting output and production factor markets, PES interventions can cause price effects and production factor movements that shift spatial pressures and trigger unintended land use change in areas not targeted by the PES scheme. The extent to which each of these potential spillover pathways becomes relevant strongly depends on local contexts and PES scheme design.

In our two study areas, farm-household mathematical programming models allow us to evaluate the likely impacts of PES promoting use-restricting and use-modifying changes. (See the following studies for the results reviewed in this section: Börner 2006; Börner, Mendoza, and Vosti 2007; Carpentier, Vosti, and Witcover 2000, 2002; Vosti, Witcover, and Carpentier 2002.) The models suggest that PES incentives for both forest conservation (western Amazon) and land retirement (eastern Amazon) could be effective strategies for carbon mitigation at current offset prices in voluntary markets. The eastern Amazon model, however, also shows that such liquidity-providing payments and increased land scarcity would increase farmers' input-intensive cash-crop production (mainly passion fruit and black pepper), at the expense of annual staple crops. By increasing payments from US$30 to US$100, the production of cash crops would increase by 11 percent (see figure 11.4). This land intensification would require the use of pesticides and fertilizers and as a result could increase water pollution. This is an example of the first spillover mechanism: losing other environmental services.

Now imagine that, in addition to compensation for forest set-aside areas, farmers were also optionally provided with access to land-intensive technologies (e.g., mechanization or fertilizers) so that they could increase the productivity of their remaining farmland. Figure 11.5 shows that, using traditional technologies in the eastern Amazon, the first 3–4 hectares of set-asides could be bought at stable, low opportunity costs. Enrolling more land would then rapidly increase costs. If intensive technologies were introduced, costs would be even lower for the first set-aside hectare, but then the set-asides would become more expensive than before. Given relaxed labor constraints, farmers could expand the new high-yield technologies to larger areas. As a result, they could raise per-hectare

Figure 11.4
Input-Intensive Annual Cash-Crop Areas and Set-Aside Payments, Eastern Amazon

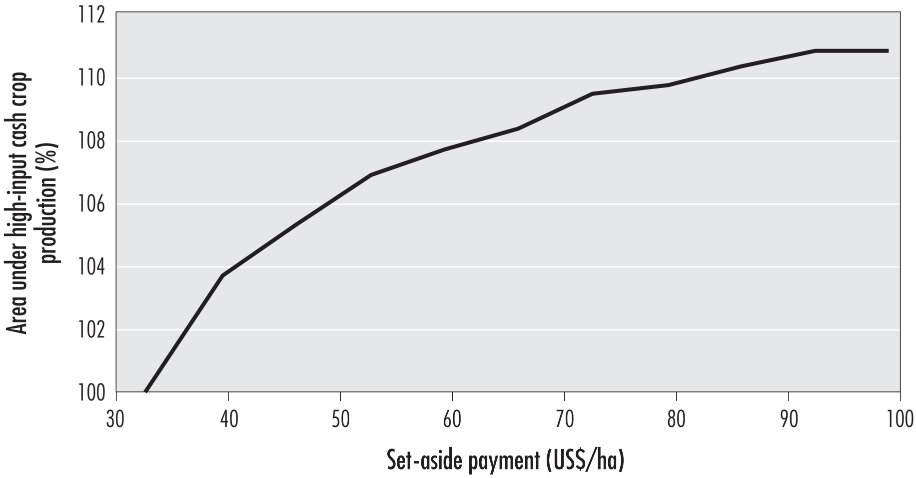
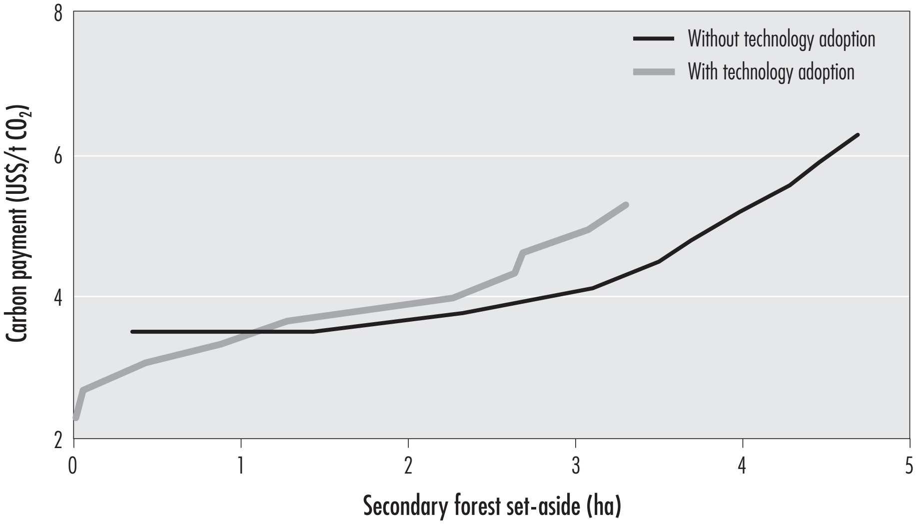


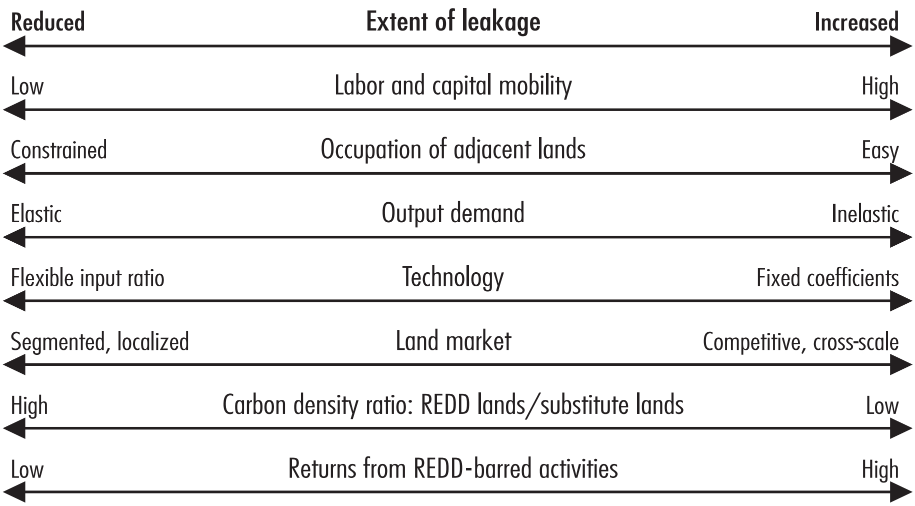
Figure 11.5
Opportunity Costs of Set-Asides With and Without Technology Adoption, Eastern Amazon



productivity and demand higher payments for set-asides. Model simulations thus predict that capital-intensive technologies could lead to increased forest loss, an example of the second spillover mechanism: overshooting adoption scale. In the eastern Amazon, adopting mechanized chop-and-mulch methods would reduce secondary forest area by 2 percent and average vegetation age by 25 percent (see figure 11.4). The compilation of global case studies by Angelsen and Kaimowitz (2001) shows that this is not an exception. It is quite common that agricultural innovations cause locally higher deforestation, even when the innovative technologies allegedly are "land-intensive."

As for leakage, the third spillover pathway, it is particularly relevant for use-restricting schemes, since they can "push" economic activities into nontarget areas. For the CRP land-retirement program, Wu (2000) estimated that for every 100 hectares retired by the CRP in the United States, 21 hectares would be brought into production, in part because of agricultural price effects. In the Noel Kempff REDD pilot project in Bolivia, a national park was extended in 1997 to stop deforestation and logging. The stop-logging component was later estimated in different models to have had leakage in the 2–42 percent range—that is, at worst, logging demand would compensate for 42 percent of the shortfall by moving extraction elsewhere, including abroad (Sohngen and Brown 2004). This large range illustrates the uncertainties involved in quantifying leakage effects. As figure 11.6 shows, leakage in REDD projects depends on a series of parameters of

Figure 11.6
Likely Explanatory Factors Behind High Versus Low REDD Leakage Scenarios



Source: Wunder (2008, 70).

flexibility in output and production-factor markets. Generally, the more flexible the economy is, the more it will succeed in substituting production in space, and thus likely raise leakage.

In conclusion, spillover effects are real issues for both use-restricting and use-modifying PES schemes. First, other services could be lost by creating new negative externalities, especially through use-modifying interventions such as carbon plantations or intensive agriculture. Use-restricting PES schemes must be rather large (e.g., national REDD programs) to evoke economy-wide spillover effects, with leakage effects dominating. In turn, ill-designed use-modifying PES incentives could render even small-scale interventions ineffective at the farm level. However, use-modifying PES cannot generally be regarded as environmentally more risky than use-restricting options. Experiences in developed countries, such as EQIP in the United States, show that large-scale use-modifying schemes can be implemented without major undesired side effects. Keeping negative spillovers at bay is thus also a matter of careful scheme design that takes relevant local particularities into account.

Conclusions and Discussion

This chapter provides a general background on PES as a tool to enhance ES provision, including climate change mitigation, in agriculture and forestry. It distinguishes between two fundamentally different PES approaches: (1) use-restricting PES, providing incentives to reduce or suspend agricultural and forestry activities on land with ES provision potential; and (2) use-modifying PES, providing incentives to adopt technologies and practices that enhance ES provision on land under productive uses. Several recent global assessments have emphasized the technical potential for ES enhancement through both use-restricting and use-modifying measures. Especially in developing countries, however, the vast majority of PES schemes have been use-restricting.

The chapter presents four hypothetical explanations for the observed mismatch between the high technical potential for use-modifying PES schemes and the small number of actual initiatives on the ground. These hypotheses are related to service provision, opportunity costs, transaction costs, and spillover effects. Based on a literature review and a comparative quantitative case study, we assessed the degree of explanatory power for each hypothesis. We found that the first two hypotheses can realistically be reaffirmed in most cases. Use-restricting ES generally compares favorably with use-modifying ES, especially in a comparison between forest-based and agricultural options. The biophysical service potential of tree-based interventions is clearly highest for most services, such as carbon retention and sequestration, biodiversity and landscape values, and watershed services in areas of not-too-large previous interventions.

This general conclusion vis-à-vis hypothesis 1 obviously hinges on how many trees an agricultural system can have before it becomes a forest. Tree-based use modification, such as the adoption of agroforestry, reforestation, and afforestation, can sometimes be as productive a carbon sequestration option as land retirement, and at the extreme it can approach the effectiveness of conserving at least degraded natural forests. Most of these measures, however, fail to achieve the habitat quality of natural ecosystems. Thus, use-restricting conservation often remains the first best option to conserve ecosystem values when these are still present in an integral landscape threatened by degradation. If degradation has already progressed to advanced stages, use-modifying schemes tend to attract more interest.

While a higher ES provision potential may often make use-restricting options more attractive in terms of per-unit service provision, adopting use-modifying measures sometimes appeals to farmers on a cost basis. The case studies in this chapter show that near-profitable technological alternatives may be readily available at lower opportunity costs than use restriction. In fact, many use-modifying agricultural technological alternatives look like no-regret opportunities when evaluated purely by cost-benefit analysis.

However, there are two problems. First, the amount of services delivered may not be high enough for buyers to engage in use-modifying schemes (see hypothesis 1). Second, adopting use-modifying technologies and practices is inherently more complex for farmers than applying a simple use-restricting rule. Adoption barriers include risk aversion, liquidity and know-how constraints, and diverse market imperfections—many of which apply to the case study sites, as well as to much of the developing world. Adoption constraints clearly feature among the most powerful explanatory factors with regard to why PES implementers have so far focused their efforts on use-restricting opportunities.

The overall evidence for the last two hypotheses, on transaction costs and spillover effects, is much more ambiguous, with many scenario-specific differences. Transaction cost proxies, such as population density and the degree of tenure security, may work in opposite directions, depending on where and how schemes are implemented. Transaction costs are clearly context dependent and must be evaluated on a case-by-case basis. That said, quite a few promising use-modifying interventions are expensive to monitor for land use compliance and service provision because a significant on-the-ground presence is needed. This applies especially to non-tree-based measures, such as low-impact land preparation and improved nutrient management, which account for significant shares of agriculture's ES provision potential. In comparison, low-cost monitoring tools such as remote sensing can play a much more prominent role in use-restricting schemes.

Heterogeneity also rules in regard to negative environmental spillover effects. First, use-restricting approaches are generally more likely to produce leakage effects, especially when the interventions are large and substitutive enough to affect output and production factor markets. This feature is particularly relevant for fledgling REDD initiatives, many of which intend to scale up PES schemes to yet unseen dimensions. Second, the eastern Amazon case study suggests that use-restricting and use-modifying schemes both could encourage investments in input-intensive agricultural technologies and thus produce negative environmental externalities, such as water pollution (i.e., losing other services). Third, use-modifying PES schemes in agriculture and forestry can sometimes become victims of their own success, with farmers overshooting adoption into environmentally sensitive areas. The case examples suggest that relative resource endowments play an important role in determining land use decisions in response to technology-specific adoption incentives. Promoting laborsaving technologies under conditions of labor scarcity, for example, will almost inevitably encourage farmers to increase operational scale and eventually expand into previously unused lands, such as forests, thus possibly causing environmentally adverse effects.

Arguably, forestry advocates and conservationists alike have been remarkable at selling conservation to government and donors, thus putting use restrictions at the center of an environmental debate. Agriculture has been perceived more as the enemy, and has only recently started to better promote its environmental potential (e.g., in carbon mitigation) (Lipper 2009). Is the dominance of use-restricting, forest-dominated approaches in PES perhaps a marketing mirage?

Although the marketing effects certainly cannot be denied, our findings point to strong underlying substance factors. It seems that PES implementers prefer use-restricting schemes primarily because they tend to provide more value for the money, and because they often appear more predictable and straightforward to design and implement. While our evidence to support this general finding is certainly biased toward the Amazon setting, most of our observations about biophysics, cost drivers, and adoption barriers persist throughout at least the developing world, and thus represent real and universal stumbling blocks for use-modifying PES. For many options involving agricultural innovation in particular, PES might not be the first best policy instrument to achieve ES provision via use modification, and it is important not to inflate its potential.

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