Essays

The Efficiency of Payments for Environmental Services in Tropical Conservation

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Abstract: *Payments for environmental services (PES) represent a new, more direct way to promote conservation. They explicitly recognize the need to address difficult trade-offs by bridging the interests of landowners and external actors through compensations. Theoretical assessments praise the advantages of PES over indirect approaches, but in the tropics PES application has remained incipient. Here I aim to demystify PES and clarify its scope for application as a tool for tropical conservation. I focus on the supply side of PES (i.e., how to convert PES funding into effective conservation on the ground), which until now has been widely neglected. I reviewed the PES literature for developing countries and combined these findings with observations from my own field studies in Latin America and Asia. A PES scheme, simply stated, is a voluntary, conditional agreement between at least one "seller" and one "buyer" over a well-defined environmental service—or a land use presumed to produce that service. Major obstacles to effective PES include demand-side limitations and a lack of supply-side know-how regarding implementation. The design of PES programs can be improved by explicitly outlining baselines, calculating conservation opportunity costs, customizing payment modalities, and targeting agents with credible land claims and threats to conservation. Expansion of PES can occur if schemes can demonstrate clear additionality (i.e., incremental conservation effects vis-à-vis predefined baselines), if PES recipients' livelihood dynamics are better understood, and if efficiency goals are balanced with considerations of fairness. PES are arguably best suited to scenarios of moderate conservation opportunity costs on marginal lands and in settings with emerging, not-yet realized threats. Actors who represent credible threats to the environment will more likely receive PES than those already living in harmony with nature. A PES scheme can thus benefit both buyers and sellers while improving the resource base, but it is unlikely to fully replace other conservation instruments.*

Keywords: economic incentives, integrated conservation and development projects, landowner compensation, stewardship

La Eficiencia de los Pagos por Servicios Ambientales en la Conservación Trópicos

Resumen: *Los pagos por servicio ambientales (PSA) representan una forma nueva y más directa para promover la conservación. Explícitamente reconocen la necesidad de abordar las ventajas los* trade offs *los intereses de los propietarios de tierra y de los actores externos mediante compensaciones. Las evaluaciones teóricas exaltan las ventajas de PSA en relación con métodos indirectos, pero la aplicación de PSA en los trópicos ha permanecido incipiente. Aquí trato de desmitificar a PSA y clarificar sus alcances en su aplicación como una herramienta para la conservación en los trópicos. Me concentro en el lado de la oferta de PSA (i.e., como convertir el financiamiento de PSA en conservación efectiva), el cual ha sido ampliamente descuidado hasta ahora. Revisé la literatura sobre PSA en países en desarrollo y combiné estos hallazgos con observaciones de mis propios estudios de campo en Latinoamérica y Asia. En pocas palabras, un esquema PSA es un acuerdo voluntario y condicional entre por lo menos un "vendedor" y un "comprador" sobre un servicio ambiental bien definido—o un uso de suelo que se presume produce ese servicio. Los mayores obstáculos para los PSA efectivos incluyen limitaciones por parte de la demanda y la falta de conocimiento sobre su implementación por parte de la oferta. El diseño de programas de PSA puede mejorar mediante la definición explícita de líneas*

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de base, el cálculo de los costos de oportunidad de la conservación, la adaptación de modalidades de pago y la identificación de agentes con credibilidad en su posesión de tierras y en sus amenazas a la conservación creíbles. La expansión de los PSA puede ocurrir si los esquemas pueden demostrar adicionalidad (i.e., efectos incrementales de la conservación frente las líneas de base predefinidos), si la dinámica de la subsistencia de los recipientes de PSA es entendida mejor, y si las metas de eficiencia son balanceadas con consideraciones de justeza. Se argumenta que los pagos de los servicios ambientales son más adecuados para escenarios de costos de oportunidad de la conservación moderados en terrenos marginales y en escenarios con amenazas emergentes, no materializadas aun. Los actores que representan amenazas creíbles al ambiente tendrán mayor probabilidad de recibir PSA que aquellos que ya viven en armonía con la naturaleza. Por lo tanto, un esquema PSA puede beneficiar tanto a compradores como a vendedores al mismo tiempo que mejora la base de recursos, pero es poco probable que reemplace completamente a otros instrumentos de conservación.

Palabras Clave: compensación a propietarios, corresponsabilidad, incentivos económicos, proyectos integrales de conservación y desarrollo

Introduction

Since the Brundtland Report (1987) and the Rio 1992 Earth Summit, tropical conservation has gradually headed in a more people-oriented direction (Sanderson & Redford 2003). The tenor of the report and the summit was that conservation would become sustainable only if poverty was alleviated. Integrated conservation and development projects (ICDPs) and sustainable forest management were two major approaches that emerged to simultaneously increase local incomes and conserve the environment (Salafsky & Wollenberg 2000; Pearce et al. 2003). Yet in spite of scattered successes, neither approach has achieved major shifts in tropical land-use trends (Sayer 1995; Brandon et al. 1998) or silvicultural practices (Rice et al. 1997; Poore 2003). Moreover, fundamental doubts exist as to how much sense it makes to forcedly link conservation and poverty alleviation agendas when trade-offs outweigh synergies (Wunder 2001; Adams et al. 2004).

Based on these insights much debate around the need for new conservation paradigms has emerged, especially in this journal (Ferraro 2001; Bawa et al. 2004; Berkes 2004; Romero & Andrade 2004). At the center of calls for more direct conservation approaches is the concept of payments for environmental services (PES) (Ferraro & Kiss 2002; Hardner & Rice 2002; Niesten & Rice 2004; Scherr et al. 2004). As wildlands and natural habitats shrink, environmental services previously provided for free are becoming increasingly threatened. This emerging scarcity makes them potentially tradable. The core idea of PES is that external beneficiaries of environmental services make direct contractual quid pro quo payments to local landowners and land users in return for adopting land and resource uses that secure ecosystem conservation and restoration. Environmental service demand can be generated through private preferences (e.g., ecotourism), public preferences (e.g., species protection), or international policies (e.g., capped carbon emissions). On the supply side the PES approach is particularly relevant where landowners' threats to service provision are legal and legitimate or where service users and society for a variety of reasons fail to secure service provision through taxation, regulation, and other command-and-control measures.

Instead of presupposing the existence of win-win solutions, this contingent conservation approach explicitly recognizes hard trade-offs and seeks to bridge conflicting interests by means of compensations. Compelling conceptual arguments have been made that PES schemes are more cost-effective than ICDPs (Simpson & Sedjo 1996; Ferraro & Simpson 2002). Schemes for PES are used in developed countries (e.g., Australia, European Union, United States) as public agroenvironmental payments inducing farmers to change practices (e.g., OECD 1997; Baylis et al. 2004) and as market-based schemes addressing both "green" and "brown" environmental problems (e.g., Bayon 2004; Fox & Nino-Murcia 2005).

In developing countries, PES remain poorly tested. There are many incipient PES initiatives (Landell-Mills & Porras 2002; Pagiola et al. 2002), but for PES schemes with some experience and with money conditionally changing hands, one is typically referred only to Costa Rica and a couple of dozen other pioneer experiences, mostly in Latin America. Until now, mainly four types of environmental services have been sold: (1) carbon sequestration and storage (e.g., northern electricity companies paying tropical farmers to plant or maintain additional trees), (2) biodiversity protection (e.g., conservation donors paying landholders for creating set-aside areas for biological corridors), (3) watershed protection (e.g., downstream water users paying upstream farmers for adopting land uses that limit soil erosion or flooding risks), (4) protection of landscape beauty (e.g., tourism operators paying a local community not to hunt in a zone used for wildlife viewing).

Reactions to PES in conservation and rural-development circles have been mixed. Advocates of PES stress that innovation in conservation is urgently needed because current approaches provide too little value for money, that PES can provide new (especially private

sector) conservation funding, and that poor service-selling communities can improve their livelihoods. Skeptics fear that PES will "bring back the fences" by delinking conservation from development, that communities risk being deprived of their legitimate land-development aspirations by asymmetrically powerful conservation consortia, and that business-type conservation may erode culturally rooted not-for-profit conservation values (Vogel 2002; Karsenty 2004; Romero & Andrade 2004).

At this stage there are three key obstacles to mainstreaming PES in the tropics. First is demand-side limitations: too few service users are sufficiently convinced to pay. Second, too little is known about the supply-side dynamics (i.e., what resource-use incentives and institutional preconditions are needed and how will the benefit transfer affect livelihood dynamics in often remote, cash-poor communities). Finally, communicating the PES concept is a problem. Proponents often use an economic rationale, whereas skeptics draw on other social sciences (anthropology, sociology, psychology, political science).

The lack of willingness to pay for environmental services is dealt with elsewhere (James et al. 2001; Balmford & Whitten 2003), so I did not address this obstacle. Instead I examined the PES supply side, which so far has received less attention. I focused primarily on forest-based examples, drawing on the PES literature and field assessments by the Center for International Forestry Research (CIFOR) in Bolivia, Vietnam, Ecuador, Indonesia, Costa Rica, and Brazil. I addressed the following questions: What defines a PES scheme? How can one evaluate to what extent an environmental service has been delivered? Is there a trade-off between efficiency and fairness in PES schemes? How much economic preanalysis is needed? Who should eventually be paid?

Definition and Key Features of PES

To my knowledge, no formalized definition of PES schemes exists in the literature, which causes some conceptual confusion. In CIFOR's field work, five criteria based on the theoretical literature are used in a working prescriptive definition of the PES principle (Wunder 2005): (1) a voluntary transaction in which (2) a well-defined environmental service (or a land use likely to secure that service) (3) is "bought" by a (minimum of one) buyer (4) from a (minimum of one) provider (5) if and only if the provider continuously secures the provision of the service (conditionality).

Genuine PES schemes are thus those that fulfill all five criteria, whereas I will refer to "PES-like schemes" as those that fulfill most but not all criteria. First, PES is conceptualized as a noncompulsory, negotiated framework, which distinguishes it from command-and-control measures. This presupposes that potential providers have real land-use choices, which for instance in Vietnam they do not. There, participation is imposed on farmers and incentives are negligible (Wunder et al. 2005). Second, what is bought needs to be well defined—it can be a directly measurable service (e.g., additional tons of carbon stored) or land-use caps that are thought to help provide that service (e.g., forest conservation provides clean water). In reality what is thought to provide a service, especially hydrological services, is often not based on scientific proof (e.g., forest cover always increases water flow); so some PES schemes may build on perceived rather than factual linkages (Calder 1999; Bruijnzeel 2004).

Third, there should be resources going from at least one service buyer to, fourth, at least one provider, although the transfer often occurs through intermediaries. Finally, in a PES scheme user payments need to be contingent upon the continuous provision of the service. Service buyers thus normally monitor compliance (e.g., has hunting or deforestation been contained in the manner stipulated in the contract?). In developed countries, permanent easements are backed by a legal enforcement apparatus (Sokolow & Zurbrugg 2003; Bayon 2004), but in developing countries this option is usually lacking, more so in agricultural frontiers with weak governance. This implies that tropical PES schemes need to pay periodically and be tied to monitored compliance. Service buyers thus must be able to withdraw from a PES contract if they do not get what they paid for. Conversely, service providers may also have an interest in flexible contracts so that they can pull out (or renegotiate the terms) of a PES scheme if changing conditions induce them to do so.

In our thorough assessment of two countries, Bolivia (Robertson & Wunder 2005) and Vietnam (Wunder et al. 2005), no single scheme satisfied all criteria, but several were PES-like schemes. For instance, watershed payments were being made, but the money came from donors rather than from service users. Conversely, sometimes money had been charged from the users but so far not been spent to pay potential environmental service suppliers. The more precise nature of the service and the link to land use often remained fuzzy. Nevertheless, the hardest criterion to meet was conditionality: many initiatives were either loosely monitored or not monitored at all, payments were up front instead of continuous, and payments were made in good faith rather than being truly contingent on service provision. This business-like PES feature—you only pay for what you get—raises political resistance in many developing countries; PES are often seen as a neoliberal Trojan horse. In sum, although the number of tropical PES-like initiatives is considerable—Landell-Mills and Porras (2002) reviewed 287 such schemes—there are many fewer genuine PES that conform strictly to the simple definition above. For instance, the Dutch FACE Foundation has for more than a decade run a carbon-sequestrating reforestation scheme in the Ecuadorian highlands that satisfies all five criteria (Albán & Argüello 2004).

Internally PES schemes constitute a diverse family. One main distinction is the vehicle used to achieve conservation or restoration effects. The most common type is an area-based scheme in which a contract stipulates land- and/or resource-use caps for a pre-agreed number of land units. Examples are conservation concessions, easements, and contracts for protecting catchments, or establishing forest-carbon plantations. The second most common type is a product-based scheme in which consumers pay a "green premium" on top of the current market price for a production scheme that is certified as environmentally friendly, especially vis-à-vis biodiversity. This green premium could be for a product that is meticulously linked to the use or nonuse values of pristine habitat (e.g., ecotourism, extractive jungle rubber), for agroecological production modes preserving relatively high service levels (e.g., shade-grown coffee, organic farming), or for environmentally conflictive production types that use best practice to minimize their negative environmental impacts (e.g., certified timber or soybean). The service price signal is less clear here than for area-based schemes because product-based schemes depend on market access and price fluctuations of the "host" commodity.

Another PES division is based on the type of buyers. In public schemes (e.g., in Costa Rica, Mexico, China), the state acts on behalf of service users as the buyer by collecting taxes and grants and paying alleged service providers—sometimes with earmarked contributions from selected service buyers. In private schemes the focus is more local (e.g., watershed schemes in Pimampiro-Ecuador, Valle del Cauca-Colombia, Los Negros-Bolivia), and buyers pay more directly. Public schemes are generally larger in scope, have lower average transaction costs, and the state provides the system with legitimacy that many private schemes struggle to attain. On the downside public schemes can become overloaded with side objectives catering to voters rather than service supply proper, are less flexible to varying payments in space, and tend to be less efficient in securing additional service provision.

Some PES, especially carbon-sequestration schemes, use markets as platforms for service buyers and sellers to interact. Nevertheless, it is often ignored that the majority of existing PES are not market based; rather, they build on bilaterally negotiated agreements between individual actors or groups of buyers and sellers (e.g., upstream communities and downstream beneficiaries in a watershed). To insist on using a market terminology for PES can raise ideological resistance and often become counterproductive to PES implementation, especially in developing countries (Wunder & Vargas 2005).

Finally, "use-restricting" PES schemes reward providers for conservation (including natural regeneration), putting caps on resource extraction and land development or fully setting aside areas (e.g., as protected habitat). Landowners are paid for their conservation opportunity costs and possibly for their active protection efforts against external threats (Hardner & Rice 2002). In contrast, in "natural asset building" schemes PES are used for environmental restoration of an area (e.g., for bringing trees back into a treeless, degraded landscape). Whether PES are an economic land rent for an agreed-upon inaction or a reward for actively improving environmental services matters for reasons of income generation: *activity-capping* PES reduce downstream employment and incomes; *activity-enhancing* schemes have opposite multiplier effects and thus raise less political resistance.

Evaluation of PES Efficacy

If one goes to the market and buys a fish, one knows in advance what one buys. When buying an environmental service, it is much less self-evident what is being paid for. Because the environmental services are provided over time, one always needs to have an idea about what would hypothetically happen without the PES scheme (i.e., construct some counterfactual service baseline). Unfortunately, with few exceptions (such as the Regional Integrated Silvopastural Ecosystem Management Project biodiversity restoration initiative [Pagiola et al. 2004] and most carbon projects), PES programs usually do not include clear, explicit frameworks for monitoring and evaluating the degree of their own success—a regrettable feature they share with other conservation interventions (Ferraro & Pattanayak 2006). The primary question service buyers ask is whether the PES scheme has a sufficiently large "additionality" (the difference in service provision between the with-PES scenario and the without-PES baseline). The additionality question (Does PES really make a difference?) has been much debated for forestry's status in the Clean Development Mechanism (CDM) of the Kyoto Protocol. Currently, only reforestation and afforestation are eligible for carbon credits, whereas improved forest management and protecting forests that in a without-PES baseline would disappear are not.

Current CDM rules are an example of a static baseline (Fig. 1a): forest carbon stocks are assumed to remain constant in a historically extrapolated laissez-faire scenario. The difference in this static baseline is then attributed to specific interventions that qualify for carbon credits. Critics of these CDM rules argue that in many tropical countries deforestation occurs as an integral part of forest-rich countries' development process. They implicitly adopt a declining baseline instead (Fig. 1b). A halt or even slow-down in deforestation would then qualify for additionality and carbon credits (avoided deforestation). Nevertheless, regions or countries in advanced stages of their forest-transition process also regain forest cover as a result of land-saving and forest-valuing development features, even without specific interventions. An example of this improving baseline (Fig. 1c) is Costa Rica, where a

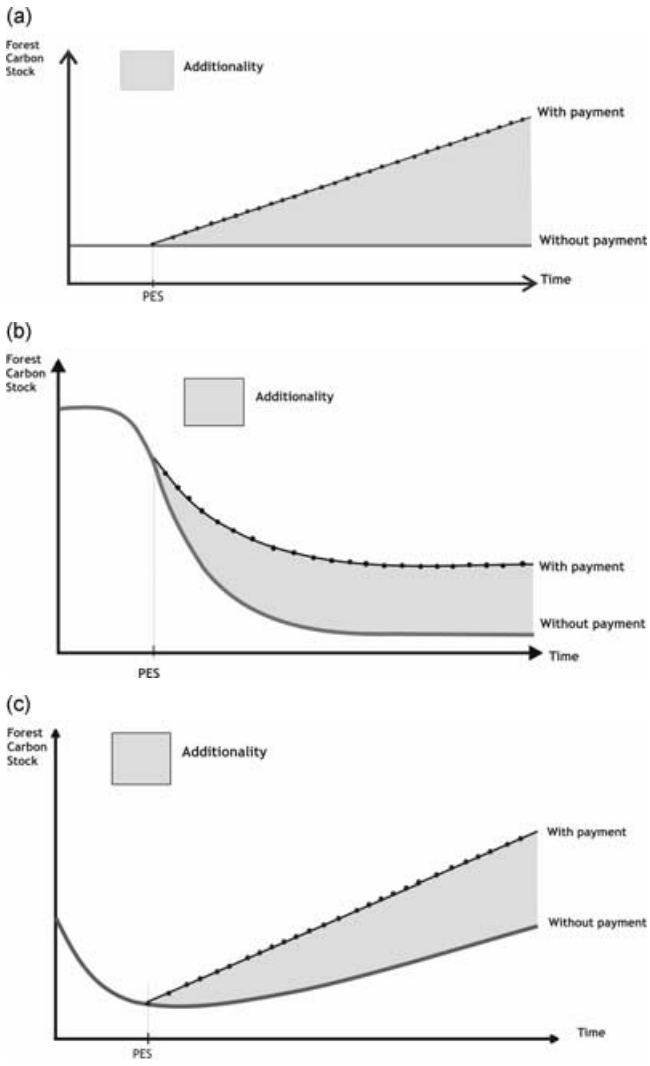

Figure 1. Three different payments for environmental services (PES) scenarios: (a) static, (b) deteriorating, and (c) improving service-delivery baseline. Dotted lines show de facto service delivered "with PES"; solid lines show counterfactual baseline "without PES." Additionality is the incremental service delivered through PES vis-à-vis the counterfactual baseline.

historical turnaround of deforestation started in the early 1990s (i.e., before the PES system was implemented from 1996 onward).

This example shows that the choice of baseline is of tremendous importance for the evaluation of the environmental impacts of PES. The Costa Rican system, which implicitly uses a static baseline (payments with respect to status quo), is likely to pay for some forest-cover establishment or conservation that would have happened anyway (reduced additionality) because the true baseline is increasing. Conversely, current CDM rules bypass important opportunities to slow down forest loss through economic incentives due to the use of a rigid static baseline (additionality foregone). Adopting the wrong baseline can thus lower PES financial efficiency or, in the worst case, waste all the money spent: if no de facto change in behavior is achieved, no additional environmental services will be produced.

Two other PES efficiency concepts are relevant whenever the intrinsic scope of the environmental service exceeds in time or space the scope of the specific PES intervention (e.g., when a PES scheme wants to address a global and/or long term problem through a local pointwise and/or short-term intervention). This is particularly relevant for carbon sequestration—a global, long-term service that is enhanced through a series of interventions specific in time and space. If a carbon PES scheme finances reforestation of a certain area, but this directly causes deforestation pressures in a neighboring area, then the PES scheme had a high "leakage": it achieved high additionality only for the project area and not for the broader, global goal. If after the scheme's termination all the reforested trees are harvested immediately for firewood uses, it would have had a lower "permanence" than if the trees were left standing. Leakage and permanence can also be relevant concepts for watershed, landscape beauty, and biodiversity goals, depending on how focused these goals are in time and space, compared with the scope of specific PES interventions.

Finally, PES efficacy also depends on transaction costs, that is, the costs to start up (search, negotiation, contracting) and run a PES scheme (administration, monitoring, enforcement). Transaction costs are distributed among buyers/intermediaries and sellers. Prima facie assertions state that direct payments should also exhibit competitive transaction costs (Ferraro & Simpson 2002). In practice, the size of PES and PES-like schemes' transaction costs is highly variable. Among developed countries, transaction costs of Canada's land-diversion program is around 25% of total costs; the U.S. Conservation Reserve Program is probably lower running but has high start-up costs (OECD 1997).

In the tropics buyer transaction costs in Costa Rica's PES program average 7%, and participating farmers pay 12–18% to intermediaries for certification (S. Pagiola, personal communication). A review of carbon-sequestration schemes in various developing countries shows costs with a range of 6–45% (Cacho et al. 2005). The PES start-up costs can be considerable, whereas running costs are moderate. My own research in Ecuador shows that the small Pimampiro watershed scheme had high start-up costs (US$69/ha) and low running costs (US$1.6/ha/year), more so than the larger Profafor-Face carbon scheme (start-up costs US$17/ha and running costs of US$6/ha/year).

In general, transaction costs are highest when many smallholders and multiple PES actors are involved, when institutions and property rights are weak, and when costs of getting baseline information and of monitoring land use and service provision are high. Economies of scale seem to reduce average transaction costs in larger-scale

schemes, such as public, nation-wide payment schemes. Unfortunately, this often comes at the cost of lower service-delivery targeting and additionality. Start-up costs tend to be more prohibitive than running costs, especially for pilot schemes; hence, donor subsidies can play a positive role here. Transaction costs can exclude smallholder participation, although collectively bundled contracts are under experiment (e.g., in Costa Rica) to overcome this obstacle.

PES Efficiency and Fairness

In conservation and rural development circles, many look to PES as a source of just reward for poor rural people who take care of the environment and continuously produce environmental services—until now, for free (Gutman 2003; Rosa et al. 2003; van Noordwijk et al. 2004). From an efficiency point of view, however, only those who constitute a credible threat to service provision or are likely to actively increase provision should be paid. Schemes for PES face intrinsic contradictions, having to balance additionality and financial efficiency goals with fairness and stewardship-reward considerations.

Consider a hypothetical example from the Brazilian Amazon. Assume a global biodiversity fund sets aside US$100,000 for a pilot PES scheme and receives three applications from potential PES suppliers. The first is from a large farmer in Mato Grosso in the so-called arc of deforestation. This farmer is rapidly clearing land because of the high profitability of soybeans, which has been facilitated by the government's road-building program. The farmer acknowledges the biological value of the forest and would prefer to preserve it, if only someone would cover his considerable conservation opportunity costs—the forgone high profits of not putting the land under soybeans.

The second application is from a group of cattle-ranching smallholders at the biodiversity-rich Andean foothills of the Amazon Basin, who are slowly enlarging their farms by clearing forestland for pasture. The economic returns to ranching on the sloped land are poor, but gradual deforestation is still improving their livelihoods. Nevertheless, if someone would pay for their opportunity costs (i.e., forgone future cattle-ranching profits), they would rather halt the expansion of the agricultural frontier.

The third application is from an indigenous community in a highly remote area of the interior of Amazonas state. Over centuries they have lived in isolation from modern society and in relative harmony with the forest. Their forest clearing does not exceed regrowth because population density is low, human mortality is high, production technology is rudimentary, and because they treasure the forest for its cultural values. External threats from loggers and ranchers still remain remote because access to the area is extremely costly. Visiting anthropologists informed the community about the new biodiversity fund, and helped them write an application to claim their just reward for being good environmental stewards who contribute to global biodiversity conservation.

All three areas are equally valuable for biodiversity, but unfortunately restricted funding will only make it possible to fund one initiative. From a fairness point of view, the indigenous community would seemingly have the strongest case. But no credible internal or external threat to their biodiversity exists, so there would be no additionality—PES would not make any tangible difference. Rewarding them might help build an alliance to meet potential long-term environmental threats, but this is a hard case to argue in the presence of multiple here-and-now threats elsewhere. The large soybean farmer clearly is at the opposite end of the threat and opportunity-cost scales. Through his aggressive continuous forest clearing, he constitutes a real threat to biodiversity—and making him a conservationist would clearly be "additional." But because net yearly per-hectare profits from soybean-cultivated land are in four-digit figures, using the US$100,000 to cover his opportunity cost would effectively only buy a tiny piece of land for conservation. The largest conservation "bang for the buck" would be achieved in the cattle-rancher setting. Ranching returns from deforestation would be far lower than for soybeans, so only modest per-hectare compensation is required. The cattle ranchers are thus probably the most attractive conservation option to achieve relatively large extra protection, rather than zero or low current additionality.

There are practical lessons to learn from this simplified example. For biodiversity buyers it may be best to keep a diversified portfolio, acting on both current and projected threats. A PES scheme needs to strike some balance between short-term efficiency and fairness, the latter influencing long-run conservation viability. But it seems certain that neither the community that fully safeguards its environment nor the impoverished farmer too poor to do much damage will emerge on the scene as major sellers of environmental services. These groups do not constitute a credible threat, so paying them creates zero additionality (payment has no impact). Is this unfair? Perhaps it is not because they also do not suffer conservation opportunity costs from forgone development. The ideal seller of environmental services is, if not outright environmentally nasty, then at least at the edge of becoming so.

Needs for Economic Analysis

A basic assessment of site-specific threats, service-provision levels, and opportunity costs helps target PES strategically in space and sets PES rates competitively. But it is often not necessary to do a full economic valuation

study of ecosystem services (buyer's benefit) and of alternative land-use profitability (provider's cost). In principle any price providers and buyers agree to can be the right price, just as right as the price negotiated for fish in a market. Yet some quick calculations can set a price range, strengthen negotiating positions for either side, or even predetermine whether a PES scheme is economically feasible. Three real-world examples can illustrate this.

First, a watershed PES pilot scheme in Los Negros (buffer zone of Amboró National Park, Bolivia) offered a low annual PES (in-kind values corresponding to about US$7/ha/year) to landowners to set aside forest land for conservation. Opportunity costs varied much in space according to slope, soil fertility, and access but generally exceeded the US$7/ha/year substantially. With such uncompetitive PES rates, surely no one would participate in the scheme? Various farmers did, however, to cash in rents for forest land that they would have conserved anyway. Although the scheme made important headway in locally piloting basic PES principles, it probably has exhibited little service additionality so far (Robertson & Wunder 2005). Knowing more about the size and spatial variability of opportunity costs could have made it possible to offer fewer but higher-priced conservation contracts to those farmers located strategically for service delivery, inducing them to actually change their deforestation plans.

Second, Brazil's nascent PES program Proambiente considers promoting shifts from slash-and-burn annual cropping to environmentally more desirable systems, inter alia by providing PES-like subsidized credits for perennials. Nevertheless, the per-hectare returns of perennials are already far more profitable (approximately US$800/year) than annual crops (US$33/year) (Almeida & Uhl 1995). Adding a marginal, recurrent capital-cost subsidy through PES is thus completely unlikely to change break-even points and to affect land-use choices because the profitability gap is too large. Perennial crop adoption might be more effectively promoted by reducing crop-disease risks, price fluctuations, credit constraints, and other barriers to entry. Traditional project approaches may be more suitable for this task than PES.

On the other hand, sometimes threat only unambiguously reveals itself when it is too late to protect. Targeting PES scenarios of projected threat may be an effective insurance against future degradation. This is the logic in a community conservation concession scheme that CIFOR is trying to develop in Setulang, East Kalimantan, Indonesia. Most neighboring villages have sold out their forests to timber companies, but Setulang has preserved 5000 ha of primary forest, mainly to protect the local water supply. The bids from logging companies are rising, however, and the internal village conservation consensus is endangered. In this situation an external biodiversity payment to local people for not selling off logging rights could help sustain proconservation attitudes. It may also strengthen communal protection efforts against logging companies' illegal invasions. Hence, strategic situations in which decisions have not yet been made are probably a scenario in which PES has a potential for achieving real and additional conservation impacts. Once the balance has tipped and the community has sold off logging rights, it is obviously too late for PES to have any impact.

Recipients of PES

Three matters relate to the selection of PES recipients that are particularly relevant in the tropics: the value-added chain, insecure land tenure, and the danger of perverse incentives with respect to illegal resource uses. Value-added concerns are related to the vertical distribution of opportunity costs. Consider the case of the Setulang conservation concession. For a conservation PES, one needs to compensate agents who would have benefited from the biodiversity-threatening activity, logging. Figure 2 shows the approximate financial and commodity flows of timber extraction and stumpage-value distribution (right-hand bar). Logs are being extracted from de jure state forests, the user rights of which are de facto claimed by different local communities through traditional land rights (*adat*) that are generally recognized by the post-Suharto Indonesian state, but claims overlap internally. Communities' negotiation power varies, causing their shares in total timber rents to diverge. Yet agents such as intermediaries (fees), timber companies (sales value), local government (taxes, bribes), and timber-product consumers (consumer surplus) are currently getting the lion's share of timber rents, and would thus also lose out most from forest conservation. Should all these actors be compensated in a PES scheme?

From an efficiency point of view, one would want to compensate enough (not necessarily all) actors with site-specific claims to form a politically resistant conservation alliance. Generally, consumers, intermediaries, and timber companies would not need compensation because their interests are mobile. Unless they buy, rent, or forcedly occupy an area, they cannot make area-specific claims, so they are unable to log it without the landowner's permission. Communities acting as direct local guardians have a vital stake and hence do need compensation. Yet if the community predictably is too weak to protect its land from various loggers' invasion, then there is no basis for a PES scheme. Local government, recently strengthened by decentralization, can be a catalytic actor that may need to be rewarded. For a PES to be fair, one would want to compensate all losers, but in this specific case that practice would be prohibitively expensive. Buying conservation for a relatively low price, aligned with timber companies' low local payments, could eventually trigger losses in national income by forgoing large timber rents paid to downstream agents (activity-reducing scheme). The pressures actors exercise

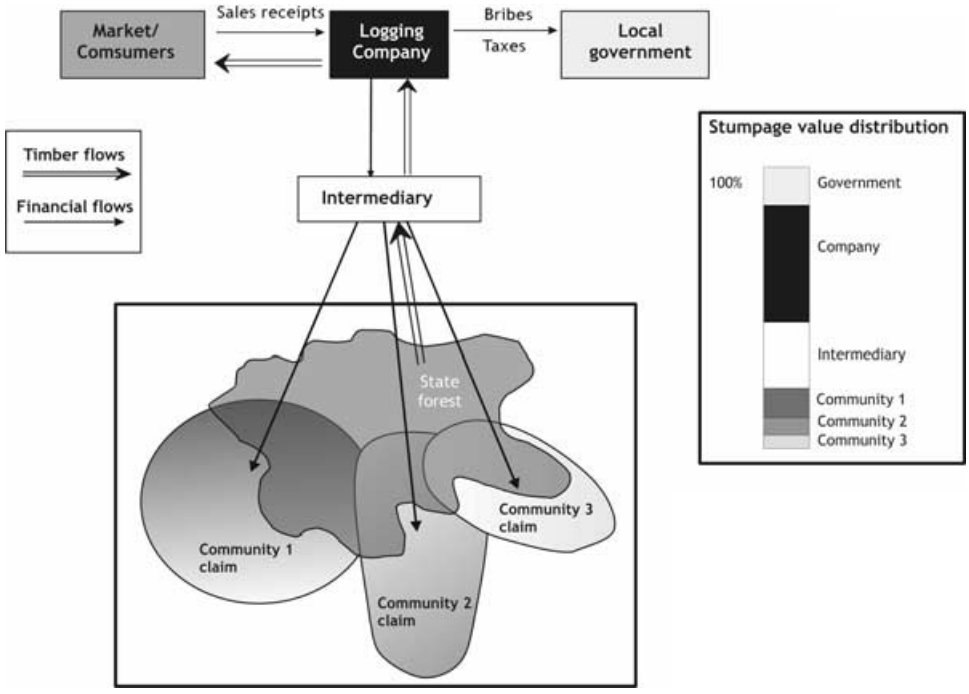

Figure 2. The flow of timber and financial resources in a compensation scheme of multiple logging beneficiaries for their losses from forest conservation through a planned payment for environmental services scheme in East Kalimantan (Indonesia).

in large, untouched forest where logging and road building may dominate are also often different from those in small forest fragments, where smallholders' incremental clearing may dominate; thus, PES recipients in these cases will also differ (Rudel 2005). Ultimately, who to pay is a question of analyzing agent-specific pressures, negotiation, and political feasibility, which includes perception of fairness and ethics because some agents lose illegal revenues, corrupt payoffs, and iniquitous profits.

As for tenure security, many land users in the tropics do not have formal land titles, especially in agricultural frontier areas. Can and should these people receive PES? The main preoccupation for service buyers here should not be de jure land rights, but de facto land- and resource-use control. Informal landowners whose land claims are widely recognized and respected can be efficient service providers when they can control access. Someone whose tenure is perceived as insecure, transitory, or weakly enforced cannot, because external agents can predictably occupy the land or harvest the resources. In disaggregating the complex concept of tenure rights, it is particularly the right-to-exclude layer that is decisive for service providers' efficiency. The more open-access type the scenario is, the less adequate is PES implementation.

Land tenure issues aside, does the legal status of resource uses matter for selecting PES recipients? Many existing legal caps on tropical land uses are weak (e.g., declared but not enforced protection forests), and some forest products (e.g., wild animals, logs, charcoal) are tropics-wide harvested illegally. Should these people receive PES to defer their threats of illegal use? If so, would until then legally complying actors receive perverse incentives to also drift into illegality to qualify for PES—or just to protest against an unfair system? Could crime eventually come to be endorsed by PES (Vogel 2002)? There is certainly a game-theory foundation for environmental blackmail (Mohr 1990), and perverse incentives have been a real concern for some PES schemes (Pagiola et al. 2004).

In many cases it is rational to use the stick-and-carrot approach (i.e., to supplement weakly enforced laws with PES compensations that partially cover the opportunity costs of compliance), especially when recent top-down protection declarations had been unfair to local land users in the first place. But a two-string strategy requires assessment of the danger of perverse incentives. When protection is already working relatively efficiently, as is the case for many protected areas globally (Bruner et al. 2001), PES will seldom be an appropriate tool. So, a minority of squatters in a national park should not be paid to stop expanding further into a park because this will create the risk of attracting new waves of squatters looking for similar rewards. Because PES presupposes de facto free land-use choices, it is normally inadequate as a protected-area management tool, except perhaps when the command-and-control potential is zero (paper parks). Ultimately, the decision whether to use the stick-and-carrot approach depends on a realistic assessment of how well a stick-only strategy can work.

More broadly, PES implementation should be preceded by an analysis of how efficient existing approaches and motivations for environmental-service provision are and how they would likely be affected by the introduction of PES. Will payments always increase recipients' effort? At

least part of the psychological literature claims that extrinsic rewards can undermine intrinsic motivation, such as a community's self-interest and pride in forest conservation (Deci et al. 1999). Large monetary transfers can sometimes be socially disruptive in cash-poor societies with weak institutions. They could also debilitate preexisting social ties and reciprocity arrangements, even when payments remain small (Heyman & Ariely 2004). At worst, conservation effort in exchange for a low monetary PES could thus be lower than for no payment. This is noteworthy because in most cases in the tropics, PES amounts have actually remained low.

Conclusion and Perspectives

Suitability of PES as Conservation Instrument

"Give a man a fish and you feed him for a day. Teach a man to fish and you feed him for a lifetime." This proverb expresses well the attraction of conservation-through-development approaches: removing the obstacles to sustainable development (poverty, shortages of capital, technology, and skills) would fix the problem by enabling people to realize their higher-level needs and embark on pro-conservation paths, in principle, forever. Unfortunately, the teaching-to-fish conservation strategy has not worked as well as planned. First, although the person may have been taught to fish, he or she might still have enough time and resources to extract logs, shoot game, and clear forests—nothing per se obliges the person to conserve. Second, what does it take to teach a person to fish? If it takes one strategy paper, two village-development plans, three participatory workshops, four action researchers, a fish-processing plant, and an army of project staff and consultants, it might be cheaper to buy the person a fish every day. This is precisely the justification for PES—the promise of higher efficiency from direct, contingent rewards.

Notwithstanding these pro-PES cost and incentive arguments, significant caveats remain. First, service buyers need sustainable finance, often into infinity—a harsh challenge not discussed here. Second, proving the linkages between land use and the environmental service can be challenging. Third, establishing PES requires trust building, which takes time. The person out in the wilderness may believe his or her land or livelihood is threatened. Finally, in donor-crowded areas (e.g., in the Andes region), communities may not accept a quid pro quo agreement when they are accustomed to donors and agencies offering benefits for free, with no demand of services in return.

Certainly, in some scenarios outright land purchases are a more rational conservation choice than the PES approach (e.g., when PES transaction costs are prohibitive). In others, command-and-control will be a better choice than PES (e.g., when society strongly favors conservation in spite of elevated opportunity costs). An ICDP-type strategy is preferable when a switch to environmentally friendly and simultaneously privately profitable production is feasible through point-wise interventions. A new generation of contingent ICDPs (i.e., people are taught to fish only if they [start to] conserve) may emerge. In other words PES will likely become one among several valid conservation approaches.

Conservation practitioners are often irresistibly attracted by high-threat scenarios, where intervention seems most badly needed. Is this also where PES should be used preferably? Payments for environmental services make sense only when there is some degree of (current or projected) threat; without threat PES makes no difference. But when high threat correlates with elevated opportunity costs, PES will usually not be the answer. Like other economic incentives, PES makes most sense at the margin of profitability, when small payments to landowners can tip the balance in favor of the desired land use. Conversely, if the desired land use is already privately more profitable than the undesired one (negative opportunity costs), it normally makes no sense to apply PES because paying is unlikely to achieve the desired shift. Payment for environmental services can really make a difference in the intermediate range of positive but numerically small opportunity costs (e.g., on degraded pastures, marginal croplands, hillside forests in slow-moving agricultural frontiers). It may also work well as environmental insurance under scenarios with projected threats (i.e., prior to the balance tipping against conservation).

PES Design

If one has chosen to go the PES route, what hints about desirable PES design can at this early development stage be given? First, it is essential for service buyers to develop a baseline in order to assess PES additionality—failing to do so can waste all PES funding by paying for things that would have happened anyway. Some idea about service providers' conservation (or restoration) opportunity costs is helpful—often more helpful than hard-fought attempts at the full economic valuation of the service itself. To reward, in the name of fairness, anybody who delivers an environmental service seems a dangerous avenue. First, current conservation funding falls far short of requirements for indiscriminate payments. For example, in the Costa Rican system applicants exceed funded PES grants by about a factor of three (Rojas & Aylward 2003). Second, being a so-called environmental service provider often means not being an environmental destructor. Across-the-board entitlements to PES could endorse blackmail by anybody owning an unthreatened environmental asset, from Scandinavian forest owners menacing to cut down their trees for receiving carbon credits, to upland settlers threatening to deliberately pollute a

river to receive watershed payments. It seems crucial not to take the PES-underlying victim pays principle to such absurd extremes. On the contrary PES payments need to be applied strategically so that additionality can be demonstrated clearly. Only in this manner can users' willingness to pay over time be enhanced. Yet this also means people already living in approximate harmony with nature without any credible internal or external threat to service provision will generally not qualify as PES recipients.

If "ecologically noble savages," nature lovers, and remote farmers too poor to seriously threaten the environment are not the prime targets of PES, who should then be paid? One should pay a critical mass of agents that both bear some (current or projected) conservation opportunity costs and have credible site-specific claims. For a timber company, that only applies if it has a concession and likely makes some profit on it. For land squatters this holds if they have an informal but widely respected claim to the land and are likely to privately benefit from an environmentally degrading use. Buyers should not necessarily refrain from contracts with (individual or collective) informal tenants as long as these have proven enforcement capacity to exclude third-party access. Buyers may also use carrots on top of existing legal paper sticks unless this glaringly leads to perverse incentives.

The preferential mode of PES payment has not been discussed so far. Consultation with PES recipients is recommended and could lead to the choice of cash, in-kind, or technical assistance—or combinations of these. When paying in cash, small but frequent payments mimicking regular income flows make the best socioeconomic sense. Large up-front payments and de facto irreversible benefits such as tenure-security provision would generally not be incentive compatible. Once benefits are handed over, leverage on service providers is lost (Wunder 2005).

Finally, I also ignored here the frequently asked question of whether PES will become a motor for poverty alleviation. Comparative assessments of PES livelihood effects (e.g., Landell-Mills & Porras 2002; Rosa et al. 2003) confirm that service-selling smallholders are bound to benefit. The effects of PES on impoverished nonsellers are mixed, but those landless poor living from environmentally degrading activities (firewood collectors, charcoal makers, bushmeat hunters) could lose out from activity-restricting schemes. This caveat applies not only to PES, but also to any intervention that succeeds in curbing environmentally degrading activities. Nevertheless, in some cases it will be necessary to compensate these losers to make PES implementation politically feasible.

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Literature Cited

Adams, W. M., R. Aveling, D. Brockington, B. Dickson, J. Elliott, J. Hutton, D. Roe, B. Vira, and W. Wolmer. 2004. Biodiversity conservation and the eradication of poverty. Science **3006:**1146–1149.

Albán, M., and M. Argüello. 2004. Un análisis de los impactos sociales y económicos de los proyectos de fijación de carbono en el Ecuador. El caso de PROFAFOR-FACE. Mercados para Servicios Ambientales, 7:74. IIED, London.

Almeida, O., and C. Uhl. 1995. Developing a quantitative framework for sustainable resource use planning in the Brazilian Amazon. World Development **23:**1745–1764.

Balmford, A., and T. Whitten. 2003. Who should pay for tropical conservation, and how could the costs be met? Oryx **37:**14.

Bawa, K. S., R. Seidler, and P. H. Raven. 2004. Reconciling conservation paradigms. Conservation Biology **18:**859–860.

Baylis, K., G. Rausser, and L. Simon. 2004. Agri-environmental programs in the United States and European Union. Pages 210-233 in G. Anania, M.E. Bohman, C.A. Carter, and A.F. McCalla, editors. Agricultural policy reform and WTO: where are we heading? Edward Elgar, Cheltenham, United Kingdom.

Bayon, R. 2004. Making environmental markets work: lessons from early experience with sulfur, carbon and wetlands. Forest Trends, Washington, D.C.

Berkes, F. 2004. Rethinking community-based conservation. Conservation Biology **18:**621–630.

Brandon, K., K. H. Redford, and S. E. Sanderson, editors. 1998. Parks in peril. People, politics and protected areas. Island Press, Washington, D.C.

Bruijnzeel, L. A. 2004. Hydrological functions of tropical forest, not seeing the soil for the trees? Environmental services and land-use change: bridging the gap between policy and research in Southeast Asia. Agriculture, Ecosystems, and Environment 104(special issue): 185-228.

Bruner, A., R. E. Gullison, R. E. Rice, and G. A. B. da Fonseca. 2001. Effectiveness of parks in protecting tropical biodiversity. Science **291:**125–128.

Cacho, O. J., G. R. Marshall, and M. Milne. 2005. Transaction and abatement costs of carbon-sink projects in developing countries. Environment and Development Economics **10:**597–614.

Calder, I. R. 1999. The blue revolution: land use and integrated water resources management. Earthscan, London.

Deci, E. L., R. Koestner, and R. M. Ryan. 1999. A meta-analytic review of experiments examining the effects of extrinsic rewards on intrinsic motivation. Psychological Bulletin **125:**627–668.

Ferraro, P. 2001. Global habitat protection: limitations of development interventions and a role for conservation performance payments. Conservation Biology **15:**990–1000.

Ferraro, P., and A. Kiss. 2002. Direct payments to conserve biodiversity. Science **298:**1718–1719.

Ferraro, P., and S. Pattanayak. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biology **4:**e105.

Ferraro, P., and R. Simpson. 2002. The cost-effectiveness of conservation payments. Land Economics **78:**339–353.

Fox, J., and A. Nino-Murcia. 2005. Status of species conservation banking in the United States. Conservation Biology, **19:**996–1007.

Gutman, P., editor. 2003. From goodwill to payments for environmental services. World Wildlife Fund, Macroeconomics for Sustainable Development Program Office, Washington, D.C.

Hardner, J., and R. Rice. 2002. Rethinking green consumerism. Scientific American **May:**89–95.

Heyman, J., and D. Ariely. 2004. Effort for payment. A tale of two markets. Psychological Science **15:**787–793.

James, A., K. Gaston, and A. Balmford. 2001. Can we afford to conserve biodiversity? BioScience **51:**43–52.

Karsenty, A. 2004. Des rentes contre le développement? Les nouveaux instruments d'acquisition mondiale de la biodiversité et l'utilisation des terres dans les pays tropicaux. Mondes en Développement **127:**1–9.

Landell-Mills, N., and I. T. Porras. 2002. Silver bullet or fool's gold? A global review of markets for forest environmental services and their impact on the poor. International Institute for Environment and Development, London.

Mohr, E. 1990. Burn the forest!: a bargaining theoretic analysis of a seemingly perverse proposal to protect the rainforest. Kiel Institute of World Economics, Kiel, Germany.

Niesten, E., and R. Rice. 2004. Sustainable forest management and conservation incentive agreements. International Forestry Review **6:**56–60.

OECD (Organisation for Economic Co-operation and Development). 1997. The environmental effects of agricultural land diversion programs. OECD, Paris.

Pagiola, S., J. Bishop, and N. Landell-Mills, editors. 2002. Selling forest environmental services. Market-based mechanisms for conservation and development. Earthscan, London.

Pagiola, S., P. Agostini, J. Gobbi, C. de Haan, M. Ibrahim, E. Murgueitio, E. Ramírez, M. Rosales, and P. R. Ruíz. 2004. Paying for biodiversity conservation services in agricultural landscapes. Environment department paper 96. World Bank, Washington, D.C.

Pearce, D., F. Putz, and J. Vanclay. 2003. Sustainable forestry in the tropics: panacea or folly? Forest Ecology and Management **172:**229–247.

Poore, D. P. 2003. Changing landscapes. The development of the International Tropical Timber Organization and its influence on tropical forest management. Earthscan, London.

Rice, D., R. E. Gullison, and J. W. Reid. 1997. Can sustainable management save tropical forests? Scientific American **276:**34–39.

Robertson, N., and S. Wunder. 2005. Fresh tracks in the forest: assessing incipient payments for environmental services initiatives in Bolivia. Center for International Forestry Research, Bogor, Indonesia.

Rojas, M., and B. Aylward. 2003. What are we learning from experiences with markets for environmental services in Costa Rica? A review and critique of the literature. International Institute for Environment and Development, London.

Romero, C., and G. I. Andrade. 2004. International conservation organizations and the fate of local tropical forest conservation initiatives. Conservation Biology **18:**578–580.

Rosa, H., S. Kandel, and L. Dimas. 2003. Compensation for environmental services and rural communities. Programa Salvadoreño de Investigación Sobre Desarrollo y Medio (PRISMA), San Salvador, El Salvador.

Rudel, T. K. 2005. Tropical forests: regional pats of destruction and regeneration in the late twentieth century. Columbia University Press, New York.

Salafsky, N., and E. Wollenberg. 2000. Linking livelihoods and conservation: a conceptual framework and scale for assessing the integration of human needs and biodiversity. World Development **28:**1421–1438.

Sanderson, S. E., and K. H. Redford. 2003. Contested relationships between biodiversity conservation and poverty alleviation. Oryx **37:**389–390.

Sayer, J. A. 1995. Science and international nature conservation. Center for International Forestry Research, Bogor, Indonesia.

Scherr, S., A. White, and A. Khare. 2004. Tropical forests provide the planet with many valuable services. Are beneficiaries prepared to pay for them? ITTO Tropical Forest Update **14:**11–14.

Simpson, R., and R. A. Sedjo. 1996. Paying for the conservation of endangered ecosystems: a comparison of direct and indirect approaches. Environment and Development Economics **1:**241–257.

Sokolow, A. D., and A. Zurbrugg. 2003. A national view of agricultural easement programs. Center for Agriculture in the Environment, American Farmland Trust, Davis, California.

van Noordwijk, M., F. Chandler, and T. P. Tomich. 2004. An introduction to the conceptual basis of RUPES. Rewarding upland poor for environmental services. World Agroforestry Center (ICRAF), Bogor, Indonesia.

Vogel, J. 2002. Markets or metaphors? A sustainable livelihoods approach to the management of environmental services: two cases from Ecuador. International Institute for Environment and Development, London.

Wunder, S. 2001. Poverty alleviation and tropical forests—what scope for synergies? World Development **29:**1817–1833.

Wunder, S. 2005. Payments for environmental services: some nuts and bolts. CIFOR Occasional paper **42**. Center for International Forestry Research, Bogor, Indonesia.

Wunder, S., and M. T. Vargas. 2005. Beyond "markets"—why terminology matters. Guest editorial. The Ecosystem Marketplace, Katoomba Group, Washington, D.C. Available from http://ecosystemmarketplace.net/pages/article.opinion.php?component_id=1252&component_version_id=2354&language_id=12 (accessed April 2006).

Wunder, S., B. D. The, and E. Ibarra. 2005. Payment is good, control is better: why payments for environmental services so far have remained incipient in Vietnam. Center for International Forestry Research, Bogor, Indonesia.