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Designing payments for environmental services in theory and practice: An overview of the issues

Stefanie Engel[a,*], *Stefano Pagiola*[b], *Sven Wunder*[c]

[a]*Institute for Environmental Decisions, ETH Zürich, Universitätstrasse 22, CHN K 76.3, CH-8092 Zürich, Switzerland*
[b]*Environment Department, World Bank, 1818 H St. NW, Washington DC 20433, USA*
[c]*Center for International Forestry Research (CIFOR), Brazil, Embrapa Amazônia Oriental, Trav. Enéas Pinheiro s/n, CEP 66.095-780, Belém, Pará, Brazil*

ARTICLE INFO

Keywords:
Payments for environmental services
Incentive mechanisms
Conservation
Ecosystem services

ABSTRACT

Payments for environmental services (PES) have attracted increasing interest as a mechanism to translate external, non-market values of the environment into real financial incentives for local actors to provide environmental services (ES). In this introductory paper, we set the stage for the rest of this Special Issue of *Ecological Economics* by reviewing the main issues arising in PES design and implementation and discussing these in the light of environmental economics. We start with a discussion of PES definition and scope. We proceed to review some of the principal dimensions and design characteristics of PES programs and then analyze how PES compares to alternative policy instruments. Finally, we examine in detail two important aspects of PES programs: their effectiveness and their distributional implications.

PES is not a silver bullet that can be used to address any environmental problem, but a tool tailored to address a specific set of problems: those in which ecosystems are mismanaged because many of their benefits are externalities from the perspective of ecosystem managers. PES is based on the beneficiary-pays rather than the polluter-pays principle, and as such is attractive in settings where ES providers are poor, marginalized landholders or powerful groups of actors. An important distinction within PES is between user-financed PES in which the buyers are the users of the ES, and government-financed PES in which the buyers are others (typically the government) acting on behalf of ES users. In practice, PES programs differ in the type and scale of ES demand, the payment source, the type of activity paid for, the performance measure used, as well as the payment mode and amount. The effectiveness and efficiency of PES depends crucially on program design.

1. Introduction

The recent Millennium Ecosystem Assessment (MA) defines ecosystem services (ES) broadly as "the benefits people obtain from ecosystems" (MA, 2003, 2005).[1] Between 1960 and 2000, the demand for ecosystem services grew significantly as world population doubled and the global economy increased more than six fold (MA, 2005). At the same time, the assessment revealed that nearly two thirds of global ecosystem services are in decline. As the report puts it, "the benefits reaped from

* *Corresponding author.*
E-mail addresses: stefanie.engel@env.ethz.ch (S. Engel), spagiola@worldbank.org (S. Pagiola), s.wunder@cgiar.org (S. Wunder).

[1] These include *provisioning services* such as food, water, timber, and fiber; *regulating services* that affect climate, floods, disease, wastes, and water quality; *cultural services* that provide recreational, aesthetic, and spiritual benefits; and *supporting services* such as soil formation, photosynthesis, and nutrient cycling (MA, 2005, p. 9). Many of the so-called 'provisioning services' may be better characterized as products. For a critique of the definition of ES in the MA, see Buyers (2007).

0921-8009/$ – see front matter © 2008 Published by Elsevier B.V.
doi:10.1016/j.ecolecon.2008.03.011

our engineering of the planet have been achieved by running down natural capital assets" (MA, 2005, p. 5). While not all conversion of natural capital is undesirable, the existence of many forms of market failure means that natural capital depletion is often much greater than would be socially optimal. These market failures include, among others, the presence of external effects, the public good nature of many ES, imperfect property rights, as well as insufficient knowledge and information (Tietenberg, 2006).

Payments for environmental services (PES) have attracted increasing interest as a mechanism to translate external, non-market values of the environment into real financial incentives for local actors to provide such services. Examples include national-scale PES programs in Costa Rica (Pagiola, 2008-this issue) and Mexico (Muñoz-Piña et al., 2008-this issue), agri-environmental schemes in Europe and the USA (Claassen et al., 2008-this issue; Dobbs and Pretty, 2008-this issue; Baylis et al., 2008-this issue), conservation concessions and easements (Niesten et al., 2004; Hardner and Rice, 2002), and forest-carbon plantations (Smith and Scherr, 2002). Despite considerable interest in the use of PES worldwide, however, few PES mechanisms have been carefully documented. Discussion of PES mechanisms has remained confined largely to the grey literature (in which, moreover, proposals for PES mechanisms are more common than assessments of actual working mechanisms). There are a growing number of published articles, but they tend to focus narrowly on specific aspects of PES, such as their impact on poverty. Moreover, the discussion of PES mechanisms in developed countries has remained largely separate from that of the emerging PES mechanisms in developing countries.

This Special Issue of *Ecological Economics* brings together detailed case studies of PES mechanisms from both developed and developing countries, along with several conceptual papers that delve deeper into specific issues of importance in PES design. It is largely based on the workshop 'Payments for Environmental Services — Methods and design in developing and developed countries', which was held in Titisee, Germany, from June 15–18, 2005. The case studies were all specially commissioned for this issue, with authors being requested to answer a list of detailed questions, so as to maximize comparability across examples. The specific cases examined cover some of the longer-lived examples of PES mechanisms in both developing and developed countries. We aim to assess the relevance of more advanced PES tools throughout the development process. What worked and what did not, in terms of reaching the various objectives that PES mechanisms often have? What lessons are applicable to other countries? What conclusions can be drawn about how best to design PES mechanisms in different settings? Our primary focus is on the efficiency of PES instruments in reaching *environmental* objectives. A secondary focus is on the impact of PES mechanisms on *human welfare* — a common implicit or explicit side objective of many PES schemes.

In this introductory chapter, we set the stage for this Special Issue by reviewing the main issues arising in PES design and implementation and discussing them in the light of environmental economics theory. We start with a discussion of PES definition and scope (Section 2). We proceed by reviewing some of the principal dimensions and design characteristics of PES programs (Section 3) and analyze how PES compares to alternative policy instruments (Section 4). We then examine in detail two important aspects of PES programs: their effectiveness and efficiency (Section 5) and their distributional implications (Section 6). Finally, Section 7 provides an overview of the structure of this Special Issue of *Ecological Economics*.

2. Definition and scope of PES[2]

Despite the growing interest in PES, there have been remarkably few efforts to define the term. In this section, we first define PES and discuss its basic logic. We then examine its properties in theory, and compare it to other policy or management approaches.

2.1. *Definition*

In many cases, the term PES seems to be used as a broad umbrella for any kind of market-based mechanism for conservation, including, for example, mechanisms such as eco-certification and charging entrance fees to tourists.[3] For the purposes of our discussion, in this Special Issue we follow Wunder (2005) in defining PES as

(a) a *voluntary* transaction where
(b) a well-defined *environmental service* (or a land use likely to secure that service)
(c) is being 'bought' by a (minimum one) *service buyer*
(d) from a (minimum one) *service provider*
(e) if and only if the service provider secures service provision (*conditionality*).

As we will see, not all PES programs presented in this Special Issue fit this definition in all regards.[4]

The basic logic of PES mechanisms is shown in Fig. 1. Ecosystem managers, whether they be farmers, loggers, or protected area managers, often receive few benefits from land uses such as, for example, forest conservation. These benefits are frequently less than the benefits they would receive from alternative land uses, such as conversion to cropland or pasture.[5] But deforestation can impose costs on downstream

[2] This section draws in part from Wunder (2005) and Pagiola and Platais (2007).

[3] This broad use of the term is often quite strategic: many donors and NGOs have told the authors, off the record, that they like to apply the term 'PES' broadly because it is a fashionable term that helps 'sell' programs.

[4] Wunder (2005) also discusses product-based payments like eco-certification as a type of PES under this definition. Instead, we focus here only on direct payments for environmental services (Grieg-Gran and Bishop, 2004), where contracts stipulate land- and/or resource-use restrictions or environmental outcomes for a pre-agreed number of land units.

[5] Although PES is often used to conserve forests, it should be stressed that the approach can in principle be used to preserve, restore, or establish any land use that generates external benefits, including many agricultural land uses.

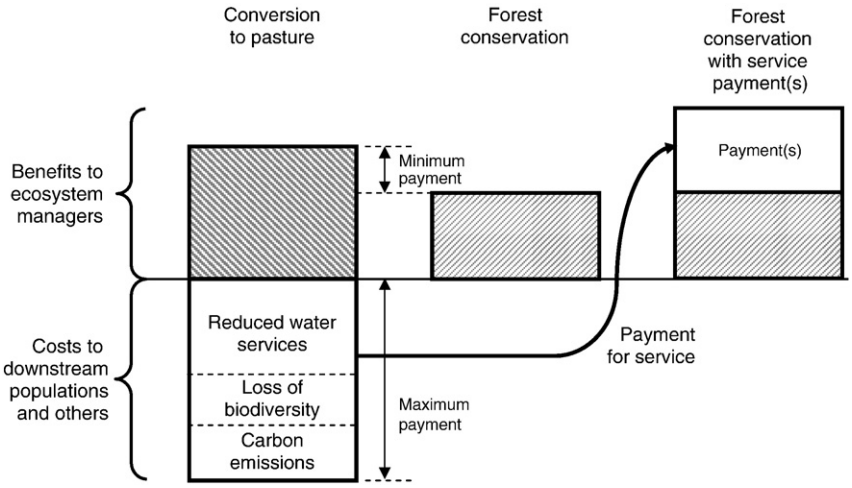

Fig. 1 – The logic of payments for environmental services. *Source*: Adapted from Pagiola and Platais (2007).

populations, who no longer receive the benefits of services such as water filtration, and on the global community, because of reductions in biodiversity and carbon storage (the actual impacts will, of course, vary from case to case). Payments by the service users can help make conservation the more attractive option for ecosystem managers, thus inducing them to adopt it (or, in the case of protected area managers, giving them the resources to do so). PES thus seeks to internalize what would otherwise be an externality (Pagiola and Platais, 2007). In effect, PES programs attempt to put into practice the Coase theorem, which stipulates that the problems of external effects can, under certain conditions, be overcome through private negotiation between affected parties (Coase, 1960). PES programs can also be seen as an environmental subsidy (to ES providers) combined, in some cases, with a user fee (on ES users).

2.2. *Scope*

It is important to note that PES is not intended as a silver bullet that can address any environmental problem. Ecosystems may be mismanaged for many reasons, not all of which are amenable to PES as a solution (Pagiola, 2003). Local ecosystem managers may not have the authority to manage ecosystems, because the ecosystems belong to nobody or to the state (which amounts to the same if the state is unable to enforce management rules) and thus tend to neglect even the on-site impacts of their management decisions (Ostrom, 2003). The suitable response in this case would be to ensure that local ecosystem managers have appropriate property rights. If ecosystem mismanagement is associated with a lack of awareness or information about land-use practices that are in the private landholder's own financial interest to adopt, then education and awareness building are appropriate responses (Bulte and Engel, 2006). Similarly, if capital market imperfections prevent landholders from adapting privately profitable technologies or practices that enhance ES provision, then providing access to credit is the most promising approach (Engel, 2007). Thus, determining whether PES is the best approach will require a careful analysis of the underlying source of market failure.[6]

The scope for application of PES, then, is to a narrow set of problems: those in which ecosystems are mismanaged because many of their benefits are externalities from the perspective of ecosystem managers (Pagiola and Platais, 2007). If a substantial portion of an ecosystem's benefits are externalities, other voluntary approaches are unlikely to bear fruit. Giving local managers property rights over the ecosystem may not be sufficient, as they will only perceive a small portion of its total benefits, and these may be less than the benefits of alternative land uses (Behera and Engel, 2006; Palmer and Engel, 2007). Likewise, training or awareness building will be unlikely to suffice, as awareness of benefits to others is unlikely to be of much weight against definite benefits to oneself to all but the most altruistic of actors.

Within the problems to which PES might be applied, an important distinction can be made on the basis of whether the ES provided are public goods and those in which they are not. It is often assumed that all ES are pure public goods, *i.e.*, that users cannot be prevented from benefiting from the ES provided (non-excludability), and that consumption by one user does not affect consumption by another (non-rivalry).[7] This certainly holds for some ES: carbon sequestration, for example, is possibly the clearest example of a public good. But

[6] In many cases, the source of the problem may not be market failure, but also policy distortions (Heath and Binswanger, 1996). In those cases, addressing the distortions is obviously the first-best solution and should be undertaken before addressing any remaining market failures.

[7] We are speaking here of ecosystem services whose benefits are indirect, or are received outside the ecosystem itself. Clearly, 'provisioning services' (in the MA's sense) are not usually externalities.

many other ES are, in fact, either excludable or rival in consumption.[8] In particular, many water services are club goods[9]: only those holding water rights or those located in a well-delineated watershed benefit. As we shall see below, this has important implications for how PES can be implemented.

3. Characteristics of PES programs

PES programs differ with respect to various design characteristics. Some reflect differences in the specific ES they are trying to generate or in the social, economic, or political context in which they operate, while others are deliberate design choices. This section examines some of the principal dimensions of PES programs.

3.1. *Who are the buyers?*

A critical first issue concerns who the 'buyers' of the ES are. In particular, there is an important distinction between cases in which the buyers are the actual users of the ES, and cases in which the buyers are others (typically the government, an NGO, or an international agency) acting on behalf of the users of the ES.

In a 'user-financed' PES program, the buyers are the actual users of an ES: a PES program in which a hydroelectric power producer pays upstream land users to conserve the watershed above its plant would be an example of this kind of PES program. Pagiola and Platais (2007) argue that this kind of PES program is particularly likely to be efficient, as the actors with the most information about the value of the service are directly involved, have a clear incentive to ensure that the mechanism is functioning well, can observe directly whether the service is being delivered, and have the ability to re-negotiate (or terminate) the agreement if needed. They refer to this kind of PES program as 'Coasian' as it most closely resembles the negotiated solution envisaged in the Coase theorem.[10]

In 'government-financed' PES programs, the buyers are a third party acting on behalf of service users. This is typically a government agency, but could also be an international financial institution or conservation institution in the case of global externalities.[11] As the buyers in this case are not the direct user of the ES, they have no first-hand information on its value, and generally cannot observe directly whether it is being provided. They also do not have a direct incentive to ensure that the program is working efficiently; on the contrary, they are often likely to be subject to a variety of political pressures. Because of these factors, Pagiola and Platais (2007) argue that such programs are less likely to be efficient. However, it should be noted that government-financed PES programs may be more cost-effective than user-financed PES because of economies of scale in transaction costs (see below).[12]

In some cases, PES programs that are operated by the government are financed through compulsory fees charged to service users rather than from general revenue. An interesting question then is whether such programs should be considered 'user-financed' or 'government-financed'. We believe that they are properly considered to be government-financed. Mexico's program of Payments for Hydrological Environmental Services (PSAH) is a good example. The PSAH program aims to preserve water supplies and is financed from a portion of the revenue generated from water use fees (Muñoz-Piña et al., 2008-this issue). At first glance, it may thus seem reasonable to consider it user-financed. In fact, however, water users make none of the decisions in this program. All program design decisions were made by the government, and although many stakeholders were consulted in the process, water users were notably absent from the table. Nor do water users have any option to withhold payments if they do not receive the water services they seek. We thus believe that this program, and others that rely on compulsory fees, should properly be considered as government-financed.[13] The key distinction between user-financed and government-financed programs, then, is not just who is paying the bills, but who has the authority to make decisions about paying the bills.

A similar question concerns PES programs that are financed directly by users, but where the users are in the public sector (for example, public sector hydroelectric power producers). Should these programs be considered as 'user-financed' or 'government-financed'? We consider them as user-

[8] Water quality, as an attribute is non-excludable, for example: if the water is clean for one user, it's clean for everybody. But that attribute is tied to actual water consumption, which is rival in consumption, and so water quality fails the definition of a public good: it doesn't do a user any good that the water is clean if another user has used it all. This emphasizes the need to consider very carefully what specific services are being provided, rather than speaking generically of 'water services'.

[9] Club goods are an intermediate category between private and public goods, that can be consumed by many individuals (the members of the 'club') without affecting the consumption of the others, but whose consumption by non-members can be prevented. A fourth category of goods are common-pool resources, but these are less relevant in our case.

[10] Other authors have called such PES programs 'self-organized' (Perrot-Maître and Davis, 2001) or 'private' (Wunder, 2005). Note that the mapping between these alternative terms and our own 'user-financed' terminology is not perfect, as the former terms focus on other characteristics.

[11] The Global Environment Facility (GEF), for example, was established by the international community to preserve global benefits such as biodiversity and carbon sequestration, so its financing for PES programs that protect global ES can be considered a payment by the users' representative, and would thus in our classification be considered a 'government-financed' program.

[12] Pagiola and Platais (2007) call government-financed PES programs 'supply-side PES' as they are only likely to have efficiency benefits on the supply side of the ES problem. Other authors have called such PES programs 'public' (Wunder, 2005), 'direct payments for conservation', (Ferraro and Kiss, 2002; Grieg-Gran and Bishop, 2004), or 'conservation concessions' (Hardner and Rice, 2002). Here, too, the mapping between these alternative terms and our own 'government-financed' terminology is not perfect.

[13] Even the phrase 'government-financed' isn't a misnomer. Revenue from water fees would have gone into general revenue if it had not been used for the PSAH program. The impact on the government budget of PSAH payments is thus exactly the same as if any other source of general revenue had been used.

financed if these users are relying on their own budgets and have ultimate decision-making authority over whether to enter into, or continue participating in, a PES program. Thus a PES program in which a municipal water utility such as that of Pimampiro, Ecuador, pays for watershed protection (Wunder and Albán, 2008-this issue) is most appropriately considered a user-financed PES program.

A somewhat intermediate case is the case where an NGO or another agency financed by voluntary contributions takes on the role of the ES buyer. Examples would include an NGO paying for an existence value like biodiversity conservation, or the Forest Carbon Partnership Facility. These are government-financed in the sense that a third party is taking the decision on how to spend funds; but they are also somewhat like a user-financed program in the sense that users' contributions are voluntary, and, thus, users could, in principle, withdraw future funding if they do not feel that the agency is investing funds appropriately.

Although there are good reasons to expect user-financed PES programs to be more efficient than government-financed ones, there are many instances in which government-financed programs may be the only option. The conditions for a Coasian solution being possible include that property rights are clearly defined and enforced and that transaction costs are low (Coase, 1960).[14] This suggests that there are situations in which user-financed programs are likely to emerge, and others where they are not. If the ES are private goods or club goods (as in the case of many water services), it is usually possible to identify the users and arrange for them to pay for service provision. If PES benefits a small number of actors, incentives to free ride and transaction costs of coordinating a joint PES program are relatively low. User-financed PES is also likely to emerge if individual users have sufficiently large ES benefits that they stand to gain from ES provision even when bearing all the costs, and/or if users have a sufficiently large share of total ES benefits that it would be unrealistic for them to expect to free ride on the efforts of others. An example is the case of a hydroelectric power producer benefiting from hydrological services from a well-defined watershed. Thus, user-financed PES programs are often implemented in situations with local monopsonies or oligopsonies.

As the number of ES buyers increases, transaction costs and incentives for free riding increase as well. Moreover, when the ES are public goods, such as biodiversity, for example, then it is often difficult to identify and delimit the users, and non-excludability implies that users have strong incentives to free ride. When appropriate conditions for user-financed PES to emerge do not hold, government involvement may be the only way that PES can be implemented. Governments can overcome the free-riding problem by charging compulsory user fees. Furthermore, governments, NGOs, or international organizations can take an important role in facilitating a Coasian outcome by reducing transaction costs. For example, Costa Rica's PSA program provides a forum for voluntary contributors to channel their contribution through an already existing administrative structure (Pagiola, 2008-this issue).

[14] In practice, whether transaction costs are low has to be judged relative to the environmental rent that can be gained from negotiation.

As most ecosystems provide not one but a large variety of ES, efforts are sometimes made to either 'bundle' various services together for sale, or to 'layer' payments from multiple buyers into payments to providers.[15] The same coordination and free-riding constraints that are encountered when there are multiple users of a single ES are encountered to an even greater degree when multiple services are sold. With few exceptions (Asquith et al., 2008-this issue), therefore, bundling and layering have remained unattainable goals.

3.2. *Who are the sellers?*

The potential 'sellers' of an ES are those actors who are in a position to safeguard the delivery of the ES. Land-use practices affect downstream water services, for example, through their effect on infiltration, evaporation, erosion, and other processes. In general, this means that the potential sellers are landholders, and the vast majority of PES programs are aimed at private landholders. It should be recalled, however, that governments are also landholders, and so PES programs can also be aimed, wholly or partially, at public lands such as protected areas.[16] In other cases, local communities have joint property rights or at least use and management rights to land and may act as collective ES providers, raising issues of intra-community distribution of PES (Rojahn and Engel, 2005).

Whoever the sellers may be, PES seeks to take advantage of their knowledge of the cost of ES provision and to seek out the low-cost providers. As long as participation is voluntary, ES sellers are unlikely to accept a payment lower than their cost of providing the ES, while conditionality ensures that they actually comply with their contracts. Ferraro (2008-this issue) examines the issue of how to structure contracts with ES providers in detail.

3.3. *How do PES mechanisms work?*

Per our definition, in a PES program an ES buyer offers a payment to an ES seller if that seller undertakes an activity that benefits the buyer. This sub-section discusses some of the details of implementation of these programs.

Type of activity. In almost all cases, PES works by paying providers for specific land uses that are thought to generate the desired ES. Forest land uses are frequently promoted by PES programs, particularly in developing countries, but PES is by no means limited to forests (see footnote 5). Aside from their impact on ES provision the nature of the land use promoted by a PES program can also have other important impacts. When programs involve changing land uses (for example, reforesting land that has been deforested), costs

[15] The expression 'bundling' has been used in both these senses in the PES literature, leading to no small amount of confusion. In the marketing literature, 'bundling' has a precise meaning: selling various services to the *same* buyer. Combining payments from different buyers is known as 'layering' (Wunder and Wertz-Kanounnikoff, in press).

[16] In South Africa's Working for Water (WfW) program, the problem is one of clearing invasive alien plants, which affect both biodiversity and water supplies. As most of the area to be cleared is public land, WfW contracts directly with firms that eradicate the invasive alien plants (Turpie et al., 2008-this issue).

tend to be much higher than when programs focus on retaining existing land uses (for example, preserving forests threatened by clearing).[17] The nature of land uses promoted by PES programs may also have impacts on local economies: for example, maintaining forest rather than converting them to agricultural use would tend to reduce local labor demand compared to what it might have been, but replacing degraded extensive pastures with more intensive silvopastoral practices would tend to increase labor demand.

Performance measures. Conditionality is critical to the definition of PES. For payments to be conditional, it must be possible to verify the existence of the ES and to establish a baseline against which additional units 'provided' can be measured. This requires understanding causal pathways ('processes'), recognizing spatial extent and distribution ('patterns'), developing 'proxies' or 'indicators' for easy recognition and monitoring, and simplified, yet accurate and validated measures of environmental services provided (Tomich et al., 2004).

Ideally, payments would be made directly on the basis of the ES provided (as, e.g. payments for carbon sequestration or wildlife offspring). Such 'output-based' payments are often not possible, however, as the level of provision of many ES cannot be observed by land users, preventing them from managing their land appropriately.[18] As noted, most PES programs thus base payments on adoption of particular land uses. In these 'input-based' PES programs, payments are often made on a per-hectare basis (referred to also as area-based PES; e.g., payment per hectare of forest conserved). Alternatively other metrics have been used to measure inputs (e.g. number of trees planted, or working hours spent for clearing exotic species).

Input-based PES programs generally split monitoring into (i) monitoring whether ES providers are complying with their contracts by undertaking the specified land uses, and (ii) monitoring whether these land uses are in fact generating the desired ES — although in practice many PES programs go no further than monitoring land-use compliance (Pagiola and Platais, 2007).

Payment amount and mode. As illustrated in Fig. 1, the payment offered to ecosystem managers must exceed the additional benefit they would receive from the alternative land use (or they would not change their behavior) and must be less than the value of the benefit to ES users (or users would not be willing to pay for it). Many PES programs use fixed payments per hectare for given activities; alternatively, payments may be differentiated in space and/or across agents on the basis of ES provided (benefit targeting), costs of ES provision (cost targeting) or a mixture of both (see Section 5 and Wünscher et al., 2008-this issue). Regarding the payment mode, PES is usually made in cash, but may also involve in-kind benefits (Asquith et al., 2008-this issue).

3.4. *How are PES mechanisms established?*

PES mechanisms are not created in a vacuum by social planners or economic theorists. They develop in particular environmental, economic, social, and political contexts, and are subject to the push and pull of many stakeholders (path dependence). Whether the initiative for the PES program comes from the ES buyers, from the sellers, or from third parties is likely to have a profound impact on the shape of the program. Side objectives such as poverty alleviation, regional development, or improving governance – whether implicit or explicit – can also have a significant influence on program design. Some programs are developed from scratch, while others build on pre-existing arrangements, perhaps originally intended for very different purposes. A related issue is whether it is better to introduce a PES program and then later improve it, or whether improving such a program later is harder than carefully designing it from the start.

4. PES vs. other policy instruments

The environmental economics literature provides a rich set of potential government interventions that can be applied to overcome problems of external effects (Baumol and Oates, 1988; Sterner, 2003). In this sub-section we compare PES to the most commonly discussed alternative approaches, namely environmental taxes and command-and-control regulation, and with the Integrated Conservation and Development (ICDP) approach that has been widely used in developing countries.

4.1. *PES vs. environmental taxes*

From the perspective of PES recipients, PES acts like an environmental subsidy: a payment aimed at inducing increases in environmentally beneficial activities. Theory tells us that environmental subsidies can, like environmental taxes, help internalize the value of ES into private land-use decisions. Unlike environmental taxes, however, environmental subsidies suffer from several sources of potential inefficiency, and thus are usually considered a second-best solution (Baumol and Oates, 1988). First, subsidies can suffer from lack of additionality (*i.e.*, paying for activities that would have been conducted anyway; see Section 5) and leakage (*i.e.*, shifting environmentally-damaging activities elsewhere in space).[19] A careful assessment of the baseline is needed to avoid these problems. A subsidy program can also create perverse incentives (e.g., inducing an expansion of environmentally destructive activities to obtain higher subsidies later on). Setting a baseline from a period prior to the design of the program can help to avoid this problem. Finally, environmental subsidies raise the profitability of the subsidized activity relative to other activities and thereby may lead to an expansion of the subsidized activity, which can be a problem if the subsidized

[17] The terms 'activity creating' and 'activity reducing' are sometimes used to describe these two kinds of PES programs. These terms can be ambiguous, as short-term impacts may differ from longer-term ones. For example, a PES program that supports reforestation of land currently used in agriculture may be activity-creating in the short term (as trees are planted) and activity-reducing in the longer term (once forest has replaced agriculture).

[18] This could result either from the ES being perceived at some distance from the ecosystem that provides it (e.g., most water services) or from the impact of individual actions being hard to separate from those of their neighbors (e.g., many biodiversity services).

[19] Taxes, however, may induce excessive abatement if they are set too high.

activity then displaces others that are environmentally preferred.[20] A further potential source of inefficiency of environmental subsidies is that they may be misused for protectionist purposes. Environmental taxes (charges on environmentally-damaging activities) suffer from fewer of these problems, and so might be considered superior to environmental subsidies. Distributional concerns often militate against the use of environmental taxes, however, as taxes would impose the cost of environmental protection on land users rather than on service users. In developed countries, politically powerful agricultural producers have often been able to direct policies towards environmental subsidies rather than taxes. In developing countries, ES providers are generally thought to be worse off than the users of those services, thus creating a strong equity preference for environmental subsidies rather than taxes (Pagiola and Platais, 2007). This issue is further discussed in Section 6 below. There are also practical issues of monitoring compliance.[21] While these apply to both taxes and subsidies, it tends to be easier to secure cooperation from land users when offering them carrots than when threatening them with a stick.

4.2. *PES vs. command-and-control regulation*

Conventional command-and-control regulation, such as restrictions on access and land use, offers an alternative way to achieve conservation objectives. Like other market-based instruments (e.g. environmental taxes and tradable permits), PES programs are considered to be more efficient than command-and-control regulation. This well-known result in environmental economics is due to the fact that command-and-control regulation tends to prescribe the same level of activity to all ES providers, while market-based instruments are more flexible. A command-and-control regulation, for example, requiring that forest be conserved would apply to all forests, irrespective of either the level of benefits they provide, or to the cost of conserving them. A PES approach would be more flexible, seeking out forest areas of higher value and lower cost. For example, a PES approach with a fixed per-hectare payment for forest conserved would induce landholders with relatively higher marginal costs of conservation to conserve less forest land than those with lower costs. If a hectare of forest conserved provides the same level of ES everywhere, such a solution would be more cost-efficient than regulating each landholder to conserve the same amount.[22]

In developing-country settings, moreover, command-and-control approaches are hampered by weak governance, high transaction costs, and information problems associated with the design of effective usage rules, monitoring, and enforcement at the local level (Baland and Platteau, 1996). The inflexibility of command-and-control regulations can also have adverse distributional consequences. Many poor communities depend on forests for their livelihoods, for example, and imposing restrictions on their use of forest resources can create economic hardship and may induce social conflict (Bulte and Engel, 2006).

It should be noted that PES programs often operate in contexts in which various command-and-control regulations pre-exist. For example, many PES programs pay for forest conservation in countries where deforestation is legally prohibited. An important question concerns the extent to which these different instruments complement or conflict with each other. PES can be thought to provide a carrot that makes the stick of regulations more palatable (Pagiola, 2008-this issue). In other cases, even weakly enforced regulations can reduce the expected gain from non-compliance, thus complementing PES programs by increasing incentives to participate and reducing the required payment rates (Wunder and Albán, 2008-this issue). More complex interactions are also possible: by raising the value of the conserved resource to local communities, PES programs can increase local people's incentives to self-enforce resource-use restrictions, thereby helping to overcome a lack of state enforcement (Engel and Palmer, 2008-this issue).

4.3. *PES vs. integrated conservation and development projects*

ICDPs encourage rural communities to maintain or provide ES by providing them with alternatives to environmentally-damaging activities. ICDPs have been popular, but empirically their success rates may be quite low (Wells et al., 1998). ICDPs mostly aim at absorbing local labor in alternative, environmentally-benign activities, such as tourism, product processing and value added (Ferraro and Simpson, 2002). Several studies have analyzed the strengths and weaknesses of the indirect approach followed in ICDPs vis-à-vis those of PES (Ferraro, 2001; Ferraro and Simpson, 2002; Kiss, 2004). The links between the activities supported by ICDPs and conservation are often dubious. Worse, these new income sources may be used as complements to existing activities rather than as substitutes, thereby failing to reduce pressure on resources or even increasing it. There is also no conditionality (beyond short-term conditionality such as participation requirements): the incentives ICDP projects provide for conservation are usually delivered up-front, in the hope that they will later result in less environmentally-damaging behavior. Should this not occur, there is no recourse. PES improves on both these grounds by directly recompensing conservation behavior and doing so conditionally.

4.4. *PES as part of a policy mix*

Although the academic discussion of PES and other instruments is often framed in terms of 'either-or', the more policy-relevant question concerns how different instruments should be combined to achieve conservation objectives. Environmental economic theory tells us that, in a second-best world where several sources of market failure coexist, a combination of instruments is needed. As Landell-Mills and Porras (2002, p.3)

[20] In principle, a similar problem might occur with taxes: by ignoring general equilibrium considerations, a tax on one activity could increase the level of another activity that is even worse than the taxed activity.

[21] For an in-depth discussion of monitoring issues in PES programs, see Meijerink (2007).

[22] Efficient PES design becomes more complex when the ES provided differ across space and when the total conservation funds available are insufficient to achieve the socially desired total level of conservation (see Section 5 on targeting). For user-financed PES, benefit and cost considerations are more likely to be taken into account automatically.

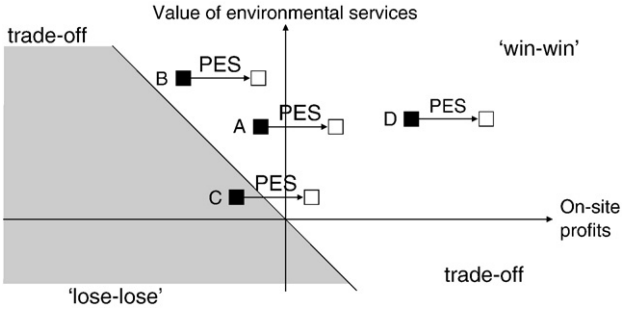

Fig. 2 – A framework to analyze the efficiency of PES. *Source*: Adapted from Pagiola (2005).

have put it, "the key question is, thus, not whether we should promote markets instead of government intervention, but what is the optimal combination of market, hierarchical and cooperative systems for governing forest sector utilization and management?" It is interesting to note, for example, that more recent World Bank-supported projects that apply the PES approach have moved away from standalone PES projects to projects that implement PES as part of broader policy approaches (Pagiola and Platais, 2007).

5. Effectiveness and efficiency of PES programs

An important question concerns the extent to which PES programs are able to meet their objectives. Does the program make sure that environmental services 'bought' constitute an improvement over the 'business as usual' scenario? Is there any mechanism aimed at ensuring benefits beyond the duration of the program? Does the program provide a mechanism to ensure environmental damages are not transferred to other areas/locations (e.g., people protecting forests under PES but then shifting pressures to other forest areas)?

Pagiola (2005) provides a framework to analyze the effectiveness of PES programs (Fig. 2). It maps land uses according to their net private profitability from the perspective of land users (horizontal axis) and the net value of the environmental services they generate to others (vertical axis). Thus, any practice in the top-right quadrant is 'win–win' in the sense of generating profits to land users while generating positive externalities. Likewise, any practice in the bottom-left quadrant is 'lose–lose'. The interesting quadrants are the top-left and bottom-right practices. At bottom-right, land-use practices are privately profitable but generate negative externalities; at top-left, practices are unprofitable to land users but generate positive externalities. It is land-use practice in this last quadrant that PES programs particularly seek to encourage.[23] The 45° diagonal separates practices whose total value to society is positive (above) from those where it's negative (below). The goal of PES programs is to make privately unprofitable but socially-desirable practices become profitable to individual land users, thus leading them to adopt them. This is illustrated in case A. Various types of inefficiency that a PES program might experience can be identified in Fig. 2:

(a) Offering payments that are insufficient to induce adoption of socially-desirable land uses, thus causing socially-undesirable land uses to remain in use (case B).
(b) Inducing the adoption of socially-undesirable land uses, that supply environmental services, but at a cost higher than the value of the services (case C).
(c) Paying for adoption of practices that would have been adopted anyway (case D).

Social inefficiency. The first two are problems that clearly result in social inefficiency: in either the failure to adopt practices whose social benefits exceed their costs, or in the adoption of practices whose benefits are smaller than their costs. In both cases, social welfare is reduced over what it might have been. In practice, judging whether these problems are experienced is frequently not possible, as valuing environmental services in monetary terms is often very difficult or costly. The type and size of payments provided by a PES program affect the likelihood of these problems arising. Costa Rica's PSA program, for example, offers a relatively low, undifferentiated, and mostly un-targeted payment (Pagiola, 2008-this issue). Thus it will only tend to attract participants whose opportunity cost of participation is low, or negative. Such a program is very likely to experience the first type of problem, in which socially-desirable land-use practices are not adopted because the payment offered is insufficient. The relatively low payments mean, however, that the program is unlikely to induce the adoption of socially-inefficient land uses on a significant scale (second problem).

Lack of additionality. The third problem of paying for adoption of practices that would have been adopted anyway is known as a lack of additionality, or "money for nothing" (Ferraro and Pattanayak, 2006). It is not a problem of social inefficiency *sensu strictu*: the practices adopted are in fact socially efficient. Rather, this problem is one of financial efficiency for the program, which is generating less ES per dollar spent than if the problem was avoided. It can result in social inefficiency, however, in cases where funds for PES are limited: payments to land uses that would have been adopted anyway reduce funds available to induce socially-efficient land-use change elsewhere. It is also inefficient in that the transaction costs involved are 'wasted'.[24] PES programs that offer low, undifferentiated, and un-targeted payments are particularly likely to experience this problem.

Leakage. Leakage (sometimes also called *spillage*) refers to the inadvertent displacement of activities damaging environmental service provision to areas outside the geographical zone of PES intervention (Robertson and Wunder, 2005). If

[23] Practices in the bottom-right quadrant would in principle be better addressed by taxes that discourage the privately profitable, but socially damaging activity. As discussed above, however, PES may also be implemented in these cases due to distributional concerns or lobbying by powerful actors causing the damages. On a more fundamental level, whether an activity induces a positive or a negative externality is an issue of definition of property rights.

[24] In government programs, transaction costs include the distortionary cost of raising revenue through taxes that are not directly linked to ES use.

leakage occurs, the environmental benefits obtained from PES may be overestimated. Leakage may occur directly, e.g., if landholders protecting forests under PES shift destructive activities to other forest areas. It may also occur more indirectly through market mechanisms. For example, land enrollment in PES for forest conservation may lead to increased prices of forest products or agricultural crops, thus encouraging extractive activities or agricultural conversion in other forest areas (Chomitz, 2002).

Lack of permanence. Permanence refers to the ability of PES to achieve long-run improvements in environmental service provision, including beyond the period of the payments proper when payment horizons are finite. Critics of PES (e.g., Swart, 2003) have stressed that permanence may be hindered by changes in external conditions (e.g., increases in market prices of agricultural crops competing with forest conservation) or by lack of long-run funding for PES (e.g., due to limited project durations). Pagiola and Platais (2007) note, however, that one of the attractions of PES is precisely that it should be able to adapt to changing conditions. As long as participation is voluntary for both buyers and sellers, both have the option to walk away at any point if conditions change. What may seem as the epitome of impermanence, however, is in fact the means by which permanence is assured: by giving both parties the ability to require that contracts be re-negotiated to cater for the new conditions.[25] Should conditions change so much that there is no longer room for a deal between ES buyers and sellers, then it is actually desirable that the program stop working, as continuing would be socially inefficient. More generally, the basic logic of PES of compensating ES providers for the externalities they generate means that it is not very useful to talk of permanence 'after payments end' — there cannot be any expectation of permanence in the absence of payments.[26] This makes the permanence of benefits of a PES program dependent on the continued flow of financing. Lack of long-run funding may be a problem in government-financed PES programs, where funding is subject to project durations or policy cycles; it is less likely to be an issue in user-financed programs, as long as the programs are delivering the ES that the users are paying for.

The role of targeting. When the number of applications to participate in the PES program exceeds available financing,[27] service buyers can use targeting to select among applicant sites to maximize the program's financial efficiency. Targeting approaches for conservation programs may be based on benefit considerations, cost considerations, or a combination of both (Babcock et al., 1997). Several authors have discussed targeting in the context of PES (Barton et al., 2003; Ferraro, 2003, 2004; Alix-Garcia et al., 2005; Wünscher et al., 2006, 2008-this issue; Engel et al., 2007).

Benefit targeting of PES would be based on actual ES (and possibly achievements of side objectives) delivered by a given site. Targeting could also be based on threats, and hence the likelihood of additionality.[28] By explicitly considering both ES levels and threats (*i.e.*, the probability that ES would not have been provided in the absence of PES) in selecting among PES applicants, the real benefits of the program can be enhanced (see, e.g., Muñoz-Piña et al., 2008-this issue).[29] Cost targeting is related to making payments flexible. Fixed payments give high production rents to land owners with low costs of ES provision, while those with high costs of ES provision are likely to not participate in the program. Thus, flexible payments equal to (or just above) the individual costs of ES provision would allow larger areas to be included in a PES program for a given budget. The challenge in cost targeting lies in estimating site-specific costs of ES provision, particularly opportunity costs. Ferraro (2008-this issue) discusses the strengths and weaknesses of alternative methods for estimating opportunity costs in the light of information asymmetries between ES providers and ES buyers. The study by Wünscher et al. (2008-this issue) provides an example of a targeting instrument that combines benefit, threat, and cost considerations.

In practice, the benefits from improved targeting have to be compared to the transaction costs associated with factors such as additional data needs and changes in administrative procedures (Wünscher et al., 2006; Engel et al., 2007). Whether the consideration of a particular targeting criterion (ES provision, threat, or provision costs) is worthwhile will also depend on the spatial variation among applicant sites with respect to this criterion (Wätzold and Drechsler, 2005).

6. Distributional implications of PES

As use of PES approaches grows, there is a need to understand how they affect the poor. The PES approach was conceptua-

[25] In this view, the typical five-year time span of PES contracts (for example, in Costa Rica and Mexico) is a good compromise between the need to periodically re-negotiate contracts and the need to keep transaction costs low by not re-negotiating every year (Pagiola and Platais, 2007). Changes in external conditions could in principle also be addressed by designing flexible contracts; for example, payments might be made dependent on the market prices of alternative products (Benítez et al., 2006).

[26] A possible exception to this statement is the case in which a short-term payment is sufficient to 'tip the balance' between environmentally-damaging and environmentally beneficial activities. For example, Roberts and Lubowski (2007) find that temporary cropland retirement payments under the U.S. Conservation Reserve Program generate land-use changes that often extend beyond contract periods. Some PES programs have been explicitly predicated on this hypothesis (Pagiola et al., 2007a). But in many such cases where the environmentally beneficial activity is privately profitable, approaches such as providing credit or technical assistance may well be sufficient.

[27] This situation frequently arises because free riding by some ES buyers reduces total offered payments below the value of the ES (in user-financed programs) or because government funding decisions are unrelated to the value of the ES (in government-financed programs). Moreover, it is very common in activity-reducing programs because there are always strong incentives for applications by ES providers with zero or negative opportunity costs.

[28] For example, forest sites may have high environmental value, but may be at low or no threat to be deforested. Similarly, if PES are made for adoption of silvopastoral practices, threat would refer to the probability that such practices would have been adopted on applicant sites anyhow.

[29] In programs of environmental restoration (e.g. reforestation or natural regeneration), the concept corresponding to 'threat' is 'opportunity', *i.e.* the probability that this positive land-use change would have happened without the PES intervention.

lized and undertaken as a mechanism to improve the efficiency of natural resource management, and not as a mechanism for poverty reduction. Nevertheless, many have assumed that PES will contribute to poverty reduction by making payments to poor land users, while others have warned of potential dangers (Landell-Mills and Porras, 2002; Pagiola et al., 2002; Kerr, 2002; Pagiola et al., 2005; Grieg-Gran et al., 2005; Ravnborg et al., 2007; Wunder, 2008). There has been little empirical verification to date, however. A review of the potential linkages between PES and poverty (Pagiola et al., 2005) raised three key questions: (1) Who are the actual and potential participants in PES programs, and how many of them are poor? (2) Are poorer households able to participate in PES programs? And (3) are poor households affected indirectly by PES programs?

Because most land users in upper watersheds and other marginal areas are thought to be poor (CGIAR, 1997; Heath and Binswanger, 1996), and because most ES are thought to come from such areas (Nelson and Chomitz, 2007), many have assumed that most potential PES recipients are poor. Whether this holds true in practice depends on the specific ES being sought and the degree to which PES programs are spatially targeted. Wunder (2008) notes that government-financed PES programs tend to be much less targeted than user-financed ones.[30] User-financed PES programs are thus likely to be much more sensitive to variations in the spatial distribution of poverty. In an analysis of highland Guatemala, Pagiola et al. (2007c) find that the assumed close spatial correlation between poverty and ES provision does not always hold. Indeed, they find no correlation between the importance of an area for water ES provision and either the incidence or the density of poverty.

The potential impacts of PES programs will only be realized by those who participate. Pagiola et al. (2005) group the factors that might affect a household's decision to participate in a PES program into three categories: factors that affect eligibility to participate, which depend on the program's targeting; factors that affect a household's desire to participate; and factors that affect their ability to participate. The three categories form a logical sequence (ability to participate only becomes an issue for households that wish to do so, and that in turn is only relevant for households that are eligible to participate). Wunder (2008) adds a fourth 'filter': whether households are competitive in terms of transaction costs.

The available evidence to date on participation of the poor in PES programs is mixed. Some studies in Costa Rica have found that many participants in the PSA program are well off (Miranda et al., 2003; Zbinden and Lee, 2005), while others have found substantial participation by poor households (Muñoz, 2004). Most such studies tend to simply observe patterns of participation, without attempting to determine which of the various factors at play caused this. Pagiola et al. (2007b, 2008) examine the extent to which poorer households are able to participate in a pilot PES program in Colombia and Nicaragua, and find either no statistical difference between the degree of participation of poorer and better-off households or that poorer households actually participate to a greater extent than better-off households. In both cases, they find that higher transaction costs are likely to be much greater obstacles to the participation of poorer households than the households' own limitations. Wunder (2008) comes to a similar conclusion.

The extent to which poorer households actually benefit from participation has also been little documented to date. To the extent that participation in a PES program is voluntary, there is a *prima facie* presumption that participants are at least no worse off than they would be without the PES program.[31] Were this not the case, they could simply decline to participate (Pagiola et al., 2005; Wunder, 2008). In cases where participation is not voluntary (deviating from our first PES criterion), on the other hand, no such presumption can be made. Even assuming that the impact is positive, the question remains as to how large it is. The proper measure to use would be the net benefits of participation (PES received minus the costs of ES provision). As these costs are usually unobserved, however, the few studies that have attempted to quantify benefits generally use indicators such as PES received as a share of income (e.g., Miranda et al., 2003).

7. Payments for environmental services in practice

The case studies in this Special Issue examine what happens when the theoretical elegance of PES meets the messiness of the real world.

The Special Issue is structured as follows. We present a range of case studies of user-financed and government-financed PES programs, from both developed and developing countries. Specifically, the cases examined include three user-financed programs: payments for watershed services in Pimampiro, Ecuador (Wunder and Albán); combined payments for watershed and biodiversity services in Los Negros, Bolivia (Asquith et al.); and payments for carbon sequestration by the PROFAFOR program in Ecuador (Wunder and Albán). The government-financed programs analyzed include the Sloping Land Conversion Program (SLCP) in China (Bennett); the Payments for Environmental Services (PSA) program in Costa Rica (Pagiola); the Payments for Hydrological Environmental Services (PSAH) program in Mexico (Muñoz et al.); the Working for Water (WfW) program in South Africa (Turpie et al.); the Conservation Reserve Program (CRP) and the Environmental Quality Incentives Program (EQIP) in the USA (Claassen et al.; Baylis et al.); the Environmentally Sensitive Area (ESA) and Countryside Stewardship Scheme (CSS) in the United Kingdom (Dobbs and Pretty); and the CAMPFIRE program in Zimbabwe (Frost and Bond). Case-study authors were asked to address the following issues: (i) services covered and actors involved, (ii) program evolution, spatial and temporal scale, transaction costs, (iii) additionality and baseline establishment, (iv) permanence, accounting and leakage, (v) payment structure

[30] Moreover, some government-financed PES programs use poverty as a targeting criterion. Part of the eligible areas for Costa Rica's PSA program, for example, are based on poverty rather than importance for ES provision (Pagiola, 2008-this issue).

[31] Where participation in PES requires undertaking long-lived investments (as in the reforestation contract in Costa Rica's PSA program (Pagiola, 2008-this issue), there is a potential for the long-term benefits of these investments to be misestimated (e.g., future timber prices or tree growth might be overestimated). Under these conditions, it is possible for participants to be worse off even when participation is voluntary.

and targeting, (vi) distributional effects, and (vii) future plans and challenges.

The case studies are followed by three conceptual papers, each of which discusses a specific aspect of PES design: (i) estimation of ES provision costs (Ferraro), (ii) targeting (Wünscher et al.), and (iii) PES in a context of weak property rights (Engel and Palmer). Finally, the conclusions chapter provides a comparative analysis of all case studies presented as well as three further case studies that were presented at the Titisee workshop. The conclusions chapter also summarizes the main results from the conceptual papers presented in this volume and draws conclusions on several of the overall issues raised above.

Acknowledgments

Funding for this research and the corresponding workshop was provided in part by the Robert Bosch Foundation, the Center for International Forestry Research (CIFOR), the European Union (EU), and the International Institute for Environment and Development (IIED). We would like to thank Paul Ferraro and John Dixon for helpful comments on an earlier draft. The opinions expressed are those of the authors and do not necessarily represent those of the World Bank Group, CIFOR, or IIED.

REFERENCES

Alix-Garcia, J., de Janvry, A., Sadoulet, E., 2005. The Role of Risk in Targeting Payments for Environmental Services. University of Montana, Missoula.

Asquith, N.M., Vargas, M.T., Wunder, S., 2008. Selling two environmental services: in-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. Ecological Economics 65, 675–684 (this issue). doi:10.1016/j.ecolecon.2007.12.014.

Babcock, B.A., Lakshminarayan, P.G., Wu, J., Zilberman, D., 1997. Targeting tools for the purchase of environmental amenities. Land Economics 73 (3), 325–339.

Baland, J.M., Platteau, J.-P., 1996. Halting Degradation of Natural Resources: Is There a Role for Rural Communities? Clarendon Press, Oxford.

Barton, D.N., Faith, D., Rusch, G., Gjershaug, J.O., Castro, M., Vega, M., Vega, E., 2003. Spatial Prioritisation of Environmental Service Payments for Biodiversity Protection. Report SNR 4746/2003. NIVA, Oslo.

Baumol, W.J., Oates, W.E., 1988. The Theory of Environmental Policy, Second edition. Cambridge University Press, Cambridge.

Baylis, K., Peplow, S., Rausser, G., Simon, L., 2008. Agri-environmental policies in the EU and United States: a comparison. Ecological Economics 65, 753–764 (this issue). doi:10.1016/j.ecolecon.2007.07.034.

Behera, B., Engel, S., 2006. The four levels of institutional analysis of the evolution of joint forest management in India: a new institutional economics approach. Forest Policy and Economics 8 (4), 350–362.

Benítez, P.C., Kuosmanen, T., Olschewski, R., van Kooten, G.C., 2006. Conservation payments under risk: a stochastic dominance approach. American Journal of Agricultural Economics 88 (1), 1–15.

Bulte, E., Engel, S., 2006. Conservation of tropical forests: addressing market failure. In: López, R., Stiglitz, J., Toman, M. (Eds.), Sustainable Development: New Options and Policies. Oxford University Press, New York.

Buyers, B., 2007. Ecosystem Services: What Do We Know and Where Should We Go? ARD, Burlington.

CGIAR, 1997. Report of the Study on CGIAR Research Priorities for Marginal Lands. CGIAR, Rome.

Chomitz, K., 2002. Baseline, leakage and measurement issues: how do forestry and energy projects compare? Climate Policy 2, 35–49.

Claassen, R., Cattaneo, R., Johansson, R., 2008. Cost-effective design of agri-environmental payment programs: U.S. experience in theory and practice. Ecological Economics 65, 737–752 (this issue). doi:10.1016/j.ecolecon.2007.07.032.

Coase, R.H., 1960. The problem of social cost. Journal of Law and Economics 3, 1–44.

Dobbs, T.L., Pretty, J., 2008. Case study of agri-environmental payments: the United Kingdom. Ecological Economics 65, 765–775 (this issue). doi:10.1016/j.ecolecon.2007.07.030.

Engel, S., 2007. Payments for Environmental Services: Potentials and Caveats. IED Newsletter No. 1. Institute for Environmental Decisions. ETH, Zürich.

Engel, S., Palmer, C., 2008. Payments for environmental services as an alternative to logging under weak property rights: the case of Indonesia. Ecological Economics 65, 799–809 (this issue). doi:10.1016/j.ecolecon.2007.07.028.

Engel, S., Wünscher, T., Wunder, S., 2007. Increasing the efficiency of conservation spending: the case of payments for environmental services in Costa Rica. In: Schmitt, C.B., Pistorius, T., Winkel, G. (Eds.), A Global Network of Forest Protected Areas Under the CBD: Opportunities and Challenges. Freiburg Schriften zur Forst- und Umweltpolitk, vol. 16. Verlag Kessel, Remagen.

Ferraro, P.J., 2001. Global habitat protection: limitations of development interventions and a role for conservation performance payments. Conservation Biology 15 (4), 990–1000.

Ferraro, P.J., 2003. Conservation contracting in heterogeneous landscapes: an application to watershed protection with threshold constraints. Agricultural and Resource Economics Review 32 (1), 53–64.

Ferraro, P.J., 2004. Targeting conservation investments in heterogeneous landscapes: a distance function approach and application to watershed management. American Journal of Agricultural Economics 86 (4), 905–918.

Ferraro, P.J., 2008. Asymmetric information and contract design for payments for environmental services. Ecological Economics 65, 810–821 (this issue). doi:10.1016/j.ecolecon.2007.07.029.

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

Ferraro, P.J., Pattanayak, S., 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biology 4 (4), e105.

Ferraro, P.J., Simpson, R.D., 2002. The cost-effectiveness of conservation payments. Land Economics 78 (3), 339–353.

Kerr, J., 2002. Watershed development, environmental services, and poverty alleviation in India. World Development 30, 1387–1400.

Kiss, A., 2004. Making biodiversity conservation a land use priority. In: McShane, T., Wells, M. (Eds.), Getting Biodiversity Projects to Work: Towards More Effective Conservation and Development. Columbia University Press, New York.

Grieg-Gran, M., Bishop, J., 2004. How can markets for ecosystem services benefit the poor? In: Roe, D. (Ed.), The Millennium Development Goals and Conservation: Managing Nature's Wealth for Society's Health. IIED, London.

Grieg-Gran, M., Porras, I., Wunder, S., 2005. How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. World Development 33, 1511–1527.

Hardner, J., Rice, R., 2002. Rethinking Green Consumerism. Scientific American, pp. 88–95. May.

Heath, J., Binswanger, H.P., 1996. Natural resource degradation effects of poverty are largely policy-induced: the case of Colombia. Environment and Development Economics 1, 65–84.

Landell-Mills, N., Porras, I., 2002. Silver Bullet or Fool's Gold? A Global Review of Markets for Forest Environmental Services and Their Impact on the Poor. IIED, London.

MA (Millennium Ecosystem Assessment), 2003. Ecosystems and Human Well-being: a Framework for Assessment. Island Press, Washington.
MA (Millennium Ecosystem Assessment), 2005. Ecosystems and Human Well-being: Synthesis. Island Press, Washington.
Meijerink, G., 2007. The role of measurement problems and monitoring in PES schemes. In: Dellink, R.B., Ruijs, A. (Eds.), Economics of Poverty, Environment and Natural Resource Use. Wageningen UR Frontis Series No. 25. Springer, Berlin.
Miranda, M., Porras, I., Moreno, M.L., 2003. The Social Impacts of Payments for Environmental Services in Costa Rica: a Quantitative Field Survey and Analysis of the Virilla Watershed. IIED, London.
Muñoz, R., 2004. Efectos del programa de pago por servicios ambientales en las condiciones de vida de los caminos de la Península de Osa. Masters thesis. Universidad de Costa Rica, San José.
Muñoz-Piña, C., Guevara, A., Torres, J.M., Braña, J., 2008. Paying for the hydrological services of Mexico's forests: analysis, negotiations and results. Ecological Economics 65, 725–736 (this issue). doi:10.1016/j.ecolecon.2007.07.031.
Nelson, A., Chomitz, K., 2007. The forest-hydrology-poverty nexus in Central America: an heuristic analysis. Environment, Development and Sustainability 9 (4), 369–385.
Niesten, E.T., Rice, R.E., Ratay, S.M., Paratore, K. (Eds.), 2004. Commodities and Conservation: the Need for Greater Habitat Protection in the Tropics. Conservation International, Washington.
Ostrom, E., 2003. How types of goods and property rights jointly affect collective action. Journal of Theoretical Politics 15 (3), 239–270.
Pagiola, S., 2003. Farmer responses to land degradation. In: Wiebe, K.D. (Ed.), Land Quality, Agricultural Productivity, and Food Security: Biophysical Processes and Economic Choices at Local, Regional, and Global Levels. Edward Elgar, Cheltenham.
Pagiola, S., 2005. Assessing the Efficiency of Payments for Environmental Services Programs: a Framework for Analysis. World Bank, Washington.
Pagiola, S., 2008. Payments for environmental services in Costa Rica. Ecological Economics 65, 712–724 (this issue). doi:10.1016/j.ecolecon.2007.07.033.
PagiolaS., S., PlataisG., G., 2007. Payments for Environmental Services: From Theory to Practice. World Bank, Washington.
Pagiola, S., Landell-Mills, N., Bishop, J., 2002. Making market-based mechanisms work for forests and people. In: Pagiola, S., Bishop, J., Landell-Mills, N. (Eds.), Selling Forest Environmental Services: Market-based Mechanisms for Conservation and Development. Earthscan, London, pp. 261–289.
Pagiola, S., Arcenas, A., Platais, G., 2005. Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. World Development 33, 237–253.
Pagiola, S., Ramírez, E., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., Ruíz, J.P., 2007a. Paying for the environmental services of silvopastoral practices in Nicaragua. Special Issue on Ecosystem Services and Agriculture, Ecological Economics 64 (2), 374–385.
Pagiola, S., Rios, A., Arcenas, A., 2007b. Poor Household Participation in Payments for Environmental Services: Lessons from the Silvopastoral Project in Quindío, Colombia. World Bank, Washington.
Pagiola, S., Zhang, W., Colom, A., 2007c. Assessing the Potential for Payments for Watershed Services to Reduce Poverty in Highland Guatemala. Washington. World Bank, Washington.
Pagiola, S., Rios, A.R., Arcenas, A., 2008. Can the poor participate in payments for environmental services? Lessons from the Silvopastoral Project in Nicaragua. Environment and Development Economics 13 (3).
Palmer, C., Engel, S., 2007. For better or for worse? Local impacts from the decentralization of Indonesia's forest sector. World Development 35 (12), 2131–2149.
Perrot-Maître, D., Davis, P., 2001. Case Studies: Developing Markets for Water Services from Forests. Forest Trends, Washington.
Ravnborg, H.M., Damsgaard, M.G., Raben, K., 2007. Payment for Ecosystem Services — Issues and Pro-poor Opportunities for Development Assistance. DIIS Report. Danish Institute for International Studies, Copenhagen.
Roberts, M.J., Lubowski, R.N., 2007. Enduring impacts of land retirement policies: evidence from the Conservation Reserve Program. Land Economics 83 (4), 516–538.
Robertson, N., Wunder, S., 2005. Fresh Tracks in the Forest: Assessing Incipient Payments for Environmental Services Initiatives in Bolivia. CIFOR, Bogor.
Rojahn, A., Engel, S., 2005. Direct payments for biodiversity conservation, watershed protection and carbon sequestration: contract theory and empirical evidence. Institute for Environmental Decisions, Chair of Environmental Policy and Economics. ETH, Zurich.
Smith, J., Scherr, S.J., 2002. Forest carbon and local livelihoods: assessment of opportunities and policy recommendations. CIFOR Occasional Paper No. 37. CIFOR, Bogor.
Sterner, T., 2003. Policy Instruments for Environmental and Natural Resource Management. Resources for the Future, Washington.
Swart, J.A.A., 2003. Will direct payments help biodiversity? Science 299, 1981.
Tietenberg, T., 2006. Environmental and Natural Resource Economics, 6th edition. Addison-Wesley, Boston.
Tomich, T.P., Thomas, D.E., van Noordwijk, M., 2004. Environmental services and land use change in Southeast Asia: from recognition to regulation or reward? Agriculture Ecosystems and Environment 104, 229–244.
Turpie, J.K., Marais, C., Blignaut, J.N., 2008. The Working for Water Programme: evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecological Economics 65, 788–798 (this issue). doi:10.1016/j.ecolecon.2007.12.024.
WätzoldF., F., DrechslerM., M., 2005. Spatially uniform versus spatially heterogeneous compensation payments for biodiversity-enhancing land-use measures. Environmental and Resource Economics 31 (1), 73–93.
Wells, M., Guggenheim, S., Khan, A., Wardojo, W., Jepson, P., 1998. Investing in Biodiversity: a Review of Indonesia's Integrated Conservation and Development Projects. World Bank, Washington, DC.
Wünscher, T., Engel, S., Wunder, S., 2006. Payments for environmental services in Costa Rica: increasing efficiency through spatial differentiation. Quarterly Journal of International Agriculture 45 (4), 319–337.
Wünscher, T., Engel, S., Wunder, S., 2008. Spatial targeting of payments for environmental services: a tool for boosting conservation benefits. Ecological Economics 65, 822–833 (this issue). doi:10.1016/j.ecolecon.2007.11.014.
WunderS., S., 2005. Payments for Environmental Services: Some Nuts and Bolts. Occasional Paper No. 42. CIFOR, Bogor.
Wunder, S., 2008. Payments for environmental services and poor: concepts and preliminary evidence. Environment and Development Economics 13 (3).
Wunder, S., Albán, M., 2008. Decentralized payments for environmental services: the cases of Pimampiro and PROFAFOR in Ecuador. Ecological Economics 65, 685–698 (this issue). doi:10.1016/j.ecolecon.2007.11.004.
Wunder, S., Wertz-Kanounnikoff, S., in press. Payments for ecosystem services: a new way of doing forest conservation. Journal of Sustainable Forestry.
Wunder, S., S.Engel, S., S.Pagiola, S., 2008. Taking stock: lessons learnt for the design of payments for environmental services programs. Ecological Economics 65, 834–852 (this issue). doi:10.1016/j.ecolecon.2008.03.010.
ZbindenS., S., LeeD.R., D.R., 2005. Paying for environmental services: an analysis of participation in Costa Rica's PSA Program. World Development 33 (2), 255–272.