Payments for Ecosystem Services: Program Design and Participation

Natasha James and Erin Sills

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Summary and Keywords

Payments for ecosystem or environmental services (PES) are broadly defined as payments (in kind or in cash) to participants (often landowners) who volunteer to provide the services either to a specific user or to society at large. Payments are typically conditional on agreed rules of natural resource management rather than on delivery of the services. The rules range from protection of native ecosystems to installation of conservation practices. The earliest proponents of PES were economists who argued that they are a cost-effective way to conserve forests, manage watersheds, and protect biodiversity. Political support for PES rests on the claim that these programs can alleviate poverty among participants as well as protect the environment. More recent literature and experience with PES reveals barriers to achieving cost-effectiveness and poverty alleviation, including many related to the distribution of participation. The Costa Rican experience illustrates the choices that must be made and the potential for innovation in the design of PES programs.

Keywords: ecosystem services, PES, Costa Rica, cost-effectiveness, equity

Introduction

Natural ecosystems provide a variety of services (Braat, 2016). The Millennium Ecosystem Assessment (MEA, 2005, p. v) defines these as “the benefits people obtain from ecosystems” and proposed the now widely accepted categorization into (1) provisioning services, e.g., food and medicine; (2) regulating services, e.g., climate regulation, pollination, and biological control; (3) cultural services, e.g., aesthetics, tourism and recreation; and (4) supporting services, e.g., habitat for species and nutrient cycling. The MEA contributed to the growing recognition of both the value and the threats to these services, leading to reinforced efforts to protect ecosystems and reframing of those efforts as measures to secure the supply of ecosystem services.

Most ecosystem services are positive externalities supplied by nature that have no traditional market and thus no market value. Economic theory suggests that the degradation of ecosystem services is a result of this market failure (Bulte, Lipper, Stringer, & Zilberman, 2008; Pattanayak, Wunder, & Ferraro, 2010). Specifically, the concern is that the public value of services lost is often greater than the private returns generated by activities that degrade or completely replace natural ecosystems. One way to correct this market failure is to create a mechanism that will internalize the cost of providing ecosystem services by having the users of ecosystem services pay the providers. This mechanism has been deemed “payment for ecosystem services” (PES) and has been widely advocated for the conservation of tropical forests (Engel, Pagiola, & Wunder, 2008).

This article is presented in three sections. The first section discusses the theory of PES, including how the definition of PES has evolved and the core components of PES programs. The second section considers the challenges of translating theory into practice, with a focus on developing countries, where PES has been widely advocated for conserving tropical forests. In this context, PES programs typically have dual objectives of environmental conservation and poverty alleviation (see Muradian, Corbera, Pascual, Kosoy, & May, 2010; Pattanayak et al., 2010). To date, there is more evidence of impacts on ecosystem conservation than on socioeconomic outcomes. The final section presents a case study of Costa Rica’s Pago de Servicios Ambientales (PSA) program, the first national policy proposed and implemented explicitly as payments for ecosystem services. By detailing the institutional structure and evolution of the PSA forest protection program, this case study illustrates some of the design choices that must be made when putting PES into practice.

What Is PES?

The early literature on PES was grounded in the Coase Theorem, which suggests that negative environmental externalities can be reduced through voluntary, market-like transactions, as long as transaction costs are low and property rights are clearly defined (Coase, 1960; Pascual, Muradian, Rodriquez, & Duraiappah, 2010; Pattanayak et al., 2010). In the context of PES, the Coase Theorem suggests that payments negotiated directly between beneficiaries and producers of ecosystem services should result in an equilibrium price and quantity that maximize welfare (Martin, Gross-Camp, Kebede, & McGuire, 2014).The earliest and most cited definition of PES (Tacconi, 2012) is by Wunder (2005, p. 3), who defines PES as “a voluntary transaction where a well-defined ecosystem service (or a land-use likely to secure that service) is being ‘bought’ by a (minimum one) ecosystem service buyer from a (minimum one) ecosystem service provider, if and only if the ecosystem service provider secures ecosystem service provision (conditionality).” This definition highlights five specific characteristics of a PES program. The first characteristic is that participation must be voluntary, clearly distinguishing PES from the command-and-control approach to conservation. Second, the ecosystem service provided must be well defined, in the sense that it should be either directly measurable (e.g., low turbidity of water) or directly generated by a specific and measurable land management practice (e.g., retiring agricultural land on steep slopes). Third, there must be at least one buyer and one seller, who agree that payments are delivered based on a conditionality agreement. In other words, payments will be awarded if and only if the ecosystem service is provided (Wunder, 2005). Although transaction costs are not explicitly addressed in this definition of PES, the Coasian perspective does recognize that low transaction costs are necessary for buyers and sellers of ecosystem services to negotiate and reach an agreement that is economically efficient and hence maximizes welfare.

As PES programs were implemented and existing programs (e.g., agri-environmental payments) were reinterpreted as PES, it became clear that they often did not meet Wunder's definition of PES. This led to a reconceptualization of PES in the scientific literature, especially in the journal *Ecological Economics* (James, 2018). One innovation was to expand the definition of ecosystem services themselves to include services generated from stock-flow resources and thus labeled "fund services" (Farley & Costanza, 2010). For example, reducing soil erosion may require a stock of trees but does not result in the physical transformation of those trees. This is consistent with payments for maintenance of the ecosystem funds (e.g., trees) rather than for the ecosystem services themselves (e.g., reduced soil erosion).

Along with this new conceptualization of ecosystem services, a new broader definition of PES was proposed. Muradian et al. (2010, p. 1205) suggest that PES are a "transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources." This definition encompasses a range of programs that vary along three axes. The first axis is how funds for the economic incentives are generated and determined. All PES programs offer economic incentives, but they may be generated through market-like transactions or through other structures, such as public subsidies. If the funds for those subsidies are raised through mandatory fees or taxes, the users of ecosystem services (i.e., the public) are not participating voluntarily (Tacconi, 2012). The second axis is the directness of transfer between the buyer and seller of ecosystem services. Essentially, this measures the degree of mediation between buyers and sellers of ecosystem services (Muradian et al., 2010; Tacconi, 2012). Muradian et al. (2010, p. 1206) note that "the most indirect situation would be then when the State represents buyers, there is one intermediary between the State and providers and the latter do not receive individual payments for their individual environmental protection efforts." Ideally, PES programs should ensure a direct transfer of benefits to those providing ecosystem services, to ensure that they are compensated for their efforts or their opportunity costs of participation. The final axis is commodification, or the degree to which the ecosystem service can be measured. While from the Coasian perspective this is a requirement, the expanded definition of PES recognizes that the ability to measure ecosystem services varies along a spectrum from services that can easily be measured (like carbon) to those that are much more subjective and therefore difficult to measure (like aesthetic beauty) (Muradian et al., 2010; Tacconi, 2012). Thus, PES is often not based on "commoditisation" of an ecosystem service but rather compensation or co-investment in particular land uses that generate multiple unquantified services (Namirembe, Leimona, van Noordwijk, Bernard, & Bacwayo, 2014).

At most points on these three axes, PES will not necessarily result in the most efficient level of ecosystem service provision, because the beneficiaries are not directly negotiating with the providers of well-defined ecosystem services. However, programs can still aim to deliver the maximum ecosystem services possible with a given budget, that is, they can maximize cost-effectiveness. Among the early proponents of PES, Paul Ferraro made this argument most forcefully, characterizing PES as a “direct” approach to conservation that was by definition more cost-effective and institutionally simpler than “indirect” approaches that promote alternative livelihoods, which he characterized as “conservation by distraction” (Ferraro, 2001; Ferraro & Kiss, 2002; Ferraro & Simpson, 2002). While economists conceived of and define PES as an instrument for environmental conservation, it is also widely expected to address poverty and rural development needs (Tallis, Kareiva, Marvier, & Chang, 2008). There are several motivations for this expanded mandate for PES. First, there is a high spatial correlation between areas that could supply ecosystem services and areas with high poverty rates (Pagiola, Arcenas, & Platais, 2005; Sunderlin et al., 2008). Thus, PES programs are likely to involve relatively rich beneficiaries of ecosystem services making payments to relatively poor providers of those services. This could serve to alleviate poverty among the providers, lead to a more equal distribution of income, and increase total welfare if there is decreasing marginal utility of income. Further, if an underlying principle of PES is that those who provide environmental services should be compensated for doing so and those providers of ecosystem services are poor or living in poverty, then it is only just for them to reap the financial benefits of providing ecosystem services to society. Additionally, in communities where poverty is a driver of ecosystem degradation, paying poor people to conserve their environment could increase the provision of ecosystem services indirectly (by increasing income) as well as directly (by increasing the value of ecosystem services) (Bulte et al., 2008). Thus, poverty alleviation could be both a mechanism and a positive externality of PES. Finally, maintaining ecosystem services that are vital to the livelihoods of the poor may help prevent them from falling deeper into poverty. These arguments lead to the conclusion that PES programs can be a “win-win,” simultaneously reducing negative environmental externalities and helping the poor (Engel et al., 2008; Muradian et al., 2010).[1]

Components of PES

While the definition and goals of PES have expanded, most economists agree that PES requires three components: buyers, sellers, and a defined level of management to generate the desired ecosystem service.

Buyers

The buyers of ecosystem services fall along a spectrum defined by the financing of the PES program. In a user-financed PES program, the buyer of the ecosystem service is the actual user of the ecosystem service (Engel et al., 2008). An example of this would be a group of farmers who live downstream paying the farmers who live upstream to reduce the runoff of fertilizer into the water. At the other extreme, a third-party buyer acts on be-half of service users. The third party can be a government agency, non-governmental organization, international donor, or some combination thereof (Engel et al., 2008; Tacconi, 2012).

Sellers

Engel et al. (2008, p. 667) define the sellers of ecosystem services as those who “are in a position to safeguard the delivery of the ecosystem service.” Generally, sellers in PES programs are either private landholders or communities with communal property rights to land, including exclusion rights that allow them to safeguard delivery of the service.

Generating Ecosystem Services

In PES programs, the buyer and seller of ecosystem services agree on a payment (cash or in kind) that will be made when and if certain conditions are met. The “commoditisation” paradigm for PES suggests that this condition should be the generation of the ecosystem service of concern. However, the most often cited definition of PES suggests that the condition may also be adoption of “a land-use likely to secure that service.” For example, a PES program could require land management that reduces pollution, such as the reduction of animal waste in streams. Other PES programs pay for land management that generates environmental amenities that are public goods, for example, planting trees to sequester carbon or generate aesthetic beauty. In either case, the PES program may require forgoing the current land use (e.g., eliminating livestock production, reforestation of agricultural land), adopting new management practices in the existing land use (e.g., fencing cattle out of streams, planting trees on contours), or forgoing conversion of existing natural resources such as avoided deforestation. Zilberman, Lipper, and McCarthy (2008) refer to the first two options as “diversion” and “working lands” programs, respectively. The third option (“conservation”) was adopted by Costa Rica in its program of Pago de Servicios Ambientales (PSA).

Cost-Effectiveness Concerns

Over the past 25 years, PES has diffused across developing countries, as documented in reviews by Bulte et al. (2008), Landell-Mills and Porras (2002), Neef and Thomas (2009), Rebelo (2009), Robalino and Pfaff (2013), and Salzman, Bennett, Carroll, Goldstein, and Jenkins (2018). However, these authors and others (e.g., Pattanayak et al., 2010) have questioned whether these programs are living up to promise of PES as a cost-effective way to ensure the provision of ecosystem services. These concerns stem from a variety of potential problems, including (1) spillover effects on areas and resources that are not considered in the conditionality agreements (called leakages if they dampen the conservation gains from the program), (2) failure of the contracted land use to secure the service, (3) non-compliance by participants, and (4) participants who would have implemented the required land use even without payments. All of these problems are shaped by the so-called “principal-agent problem” (Platteau, 2000) that is inherent to PES programs.

Principal-Agent Problem

Applying principal-agent theory to PES, the agent is the landowner or seller of ecosystem services and the principal is the buyer or the institution implementing the PES program. The principal's goal is to maximize ecosystem services (i.e., hectares of forest conserved to sequester carbon) for a given budget (Platteau, 2000). The agent's goal is to maximize her own well-being, which may be approximated by profits from land management including any direct payments. Persson and Alpizar (2013) propose a typology of the agents who could voluntarily apply to enter a PES program that pays for some type of "conservation":

A: Those who apply for payments, but would conserve even without payments
B: Those who apply for payments and would not conserve without payments
C: Those who do not apply for payments but will conserve regardless (due to the fact that they were already or were planning to conserve)
D: Those who do not apply for payments and will not conserve

There could also be agents who apply for payments but who will not or cannot fulfill the conditions of the contract. To reflect this moral hazard problem, we add a fifth agent:

E: Those who apply for payments, but will not conserve

These five types of agents are summarized in Table 1.

Table 1. Agent Types Defined by Behavior With and Without PES Program

	Behavior	
Type	With PES Program	Without PES Program
A	Apply and conserve	Conserve
B	Apply and conserve	Do not conserve
C	Do not apply and conserve	Conserve
D	Do not apply and do not conserve	Do not conserve
E	Apply and do not conserve	Do not conserve

For any given PES program (with specified conditions and payment level), the agents know their own types, but this is private information. Gathering the information required to determine each agent's type would be costly (in both time and money) for the principal, and thus typically agents are allowed to self-identify when they apply for the program and are subject to only limited monitoring once they enroll. This information asymmetry and the interactions between the types of agents can decrease the cost-effectiveness of PES programs through compliance failures, lack of additionality, and spillover effects. All three of these can potentially be addressed through program design, but with increased transaction costs (Alston et al., 2013).

Compliance

Economists emphasize the importance of conditionality for PES: payments should only be rendered to an ecosystem services provider if the management plan is implemented and the ecosystem services are generated. In many PES programs, the participating agents are bound by contract to implement management plans that are expected to generate ecosystem services. To determine if the participating agents are in compliance with their contracts and the agreed management plans, the program administrator must allocate resources to detection, enforcement, and litigation of violations. For cost-effective conservation, agents should not receive payments if they do not comply with the conditionality requirements. Yet Wunder (2007) found that PES programs in developing countries often fail to enforce conditionality requirements. In some programs, payments are made upfront and in good faith. One possible reason is that program implementers recognize that monitoring and withholding payments would strain their relationships with landowners (often poor and rural) that participate in the program (Wunder, 2006). These problems could be avoided if Type E agents were screened out at the application stage, but this is complicated by information asymmetries: Type E agents could self-identify (or incorrectly expect to behave) as Type B agents.

Additionality

While compliance means that the agent is delivering ecosystem services (or at least the agreed upon land use), it does not necessarily mean that those are additional services above and beyond what would have been generated without payments (i.e., additionality). In fact, compliance failure may be more likely when there is additionality, because that implies that some profitable activity is being displaced. Additionality is considered a fundamental requirement for cost-effective PES.[2] From an economic standpoint, it is inefficient to pay to conserve land that would have been conserved in the absence of payments (Bennett, 2010; Alpizar, Norden, Pfaff, & Robalino, 2013). To maximize additionality, only Type B agents can be included in PES programs (Norden, Persson, & Alpízar, 2013).

Adverse self-selection occurs when Type A agents also enter the program, possibly by “pretending” to be Type B agents. Information asymmetries again explain this problem, which is particularly severe when agents are not currently meeting the PES conditions but plan to. If the principal erroneously accepts too many Type A agents into the program, additionality can be substantially diluted (Moon, Marshall, & Cocklin, 2012; Norden et al., 2013).

Price and Behavioral Spillovers

Most analyses of PES focus on the behavior of the agents and the use of the land in the areas participating in the program. Those who participate are under contract to either conduct or forgo some activity on the land under contract. Agents who are generating ecosystem services but are not participants are not obliged directly by the PES principal to implement any particular land management regime. However, especially when there is additionality, PES may have spillover effects on both land areas not under contract and agents not under contract (Alix-Garcia, Shapiro, & Sims, 2012).

Consider first land that is not under contract but is owned by an agent who has other land under contract (on the same or other properties). Placing part of a property under contract may change the availability of fixed inputs (such as family labor or equipment) for use on other parts of the property, if those inputs would otherwise have been employed (or not employed) in the area under contract. Second, the payment itself may relax cash flow or credit constraints on productive or conservation activities on land not under contract. Third, the payment may either displace or reinforce intrinsic conservation motivations that help shape decisions about management of land not under contract. Experimental evidence offers support for both crowding out and crowding in of pro-environmental motivations (Rode, Gómez-Baggethun, & Krause, 2015). In programs that pay for conservation, payments in one area may demonstrate the option value of conserving other areas that could be enrolled in the program later (Arriagada, Ferraro, Sills, Pattanayak, & Cordero-Sancho, 2012).

Spillover effects on agents not under contract can have implications for conservation across the entire landscape. PES programs can introduce new dynamics between participants and non-participants. In addition to agents who do not apply for the program (types C and D), there are almost always some agents who apply for, but are not awarded PES contracts because of budgetary limitations. These non-participants may be subject to spillover effects operating through two channels: prices and behavior.

Price spillover occurs when the introduction of a PES program changes the prices of outputs such as agricultural products or inputs such as labor. To illustrate how this can occur, consider a PES program for forest preservation implemented in a closed economy with Type B agents (those who apply for payments and would not conserve without payments) and Type D agents (those who do not apply and will not conserve). The Type B agents apply and some are selected to receive payments to set aside and not convert forestland to agriculture. By definition, PES has “additional” impacts on these agents. As illustrated in Figure 1, this additionality could lead to a decrease in supply (assuming an isolated economy and no change in the demand schedule for agricultural products) and a resulting increase in agricultural prices. The rejected Type B agents and the Type D agents may increase their agricultural production in response to the higher prices. Thus, it is possible that in areas isolated from the broader market, the introduction of PES will cause a shift in production to other properties rather than just reducing production, resulting in a lower-than-expected net gain in conservation (see Alpizar et al., 2013; Norden et al., 2013).[3]

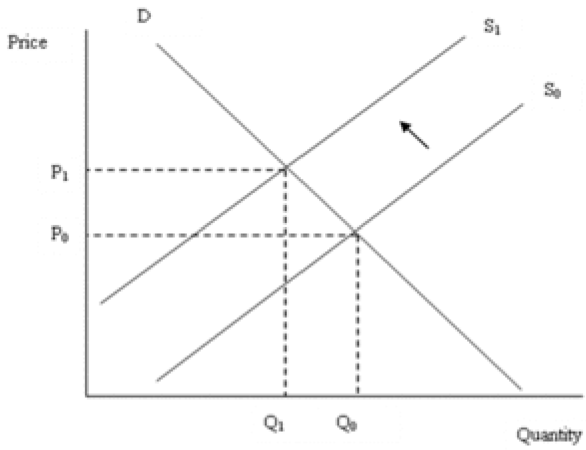

Figure 1. Increased conservation can restrict supply (e.g., of pasture), thus increasing the price if the demand scheduled does not also shift down.

PES could also have spillover effects through behavioral channels. For example, non-participants may feel slighted due to their exclusion from the program. One example of behavioral spillover is the crowding out of intrinsic motivations (Rode et al., 2015). Again consider a forest preservation PES program, but in this case with Type B agents (those who apply for payments and would not conserve without payments) and Type C agents (those who do not apply for payments but will conserve regardless). When Type B agents are selected for participation in the program and receive payments, Type C agents may feel slighted, because they were already implementing land management practices for which other agents are now receiving payments. This could crowd out the motivations that led Type C agents to implement the land management practices that secured ecosystem services. As with spillover effects through prices, this could result in a lower-than-expected net gain in conservation. Alpizar et al. (2013) demonstrated this effect in a field experiment with Costa Rican landowners.[4] Together price and behavioral spillovers can reduce the effectiveness of PES across the entire landscape, even if there is perfect compliance and high additionality on the parcels under contract. In addition to shifting undesired activity to neighboring parcels, spillovers may postpone them to a future time (e.g., Pagiola, Honey-Rosés, & Freire-González, 2016). The permanence of PES impacts is an area ripe for further research (Börner et al., 2017).

The limited evidence to date shows that PES does protect natural ecosystems, usually forests, that are placed under contract (Alix-Garcia & Wolff, 2014; Börner et al., 2017; Miteva, Pattanayak, & Ferraro, 2012; Samii, Lisiecki, Kulkarni, Paler, & Chavis, 2014). As noted by Alix-Garcia and Wolff (2014), most causal evaluations of PES have considered the national systems in Costa Rica and Mexico, where the background rates of deforestation are low. In this context, PES has a larger percentage impact than absolute impact on deforestation rates. In contrast, the one randomized controlled trial (RCT) of PES for avoided deforestation to date was implemented in the context of high deforestation and low opportunity costs (Jayachandran et al., 2017). The results confirm that given the right conditions and careful implementation, PES can have substantial and statistically significant impacts on deforestation.

Equity Concerns

PES was originally advocated as a cost-effective way to achieve conservation, but part of its appeal was the possibility that it would contribute to rural poverty alleviation by offering payments rather than imposing restrictions on poor rural suppliers of ecosystem services (Ferraro & Kiss, 2002). Further, many institutions that implement PES have social as well as environmental goals. National-level PES program in developing countries such as China, Mexico, Vietnam, South Africa, and Costa Rica all employ some level of pro-poor targeting to ensure that poor providers of ecosystem services have access to payments (Wunder, 2008).

In the scientific literature there is more debate over the relevance of social equity to designing and evaluating PES. Some proponents of PES have argued that introducing social objectives will negatively impact its cost-effectiveness at achieving conservation (Wunder, Engel, & Pagiola, 2008). Others note that a PES program that does not take equity into consideration or that leads to an unfair distribution of benefits has little chance of being accepted by both ecosystem service sellers and buyers (Kinzig et al., 2011; Muradian et al., 2010). According to Pascual et al. (2014), there is growing concern that the focus on the cost-effectiveness of PES promised by theory comes at the expense of attention to its social equity dimensions. Although equity is not a central concern of many traditional and sometimes effective conservation policies, such as protected areas and other command and control measures, Pascual et al. (2014, p. 1029) note that "it is increasingly accepted that integrating social considerations into environmental management planning is instrumental to achieving more-robust ecological outcomes."

Equity can be defined as the "distribution of socioeconomic factors and goods in a society according to an agreed set of principles or criteria" (Corbera, Brown, & Adger, 2007, p. 589). Thus, the equity of a PES program could be considered a function of which ecosystem services providers receive payments. In turn, this is a function of how barriers to participation create differential access to PES (access equity) and the resulting distribution of participation, which determines who receives payments (outcome equity).[5]

Equity: Barriers to Participation

Barriers to participation in PES include high transaction costs and insecure land tenure.

Transaction Costs

When participation in PES is voluntary, landowners seek to participate when they expect that participation will make them better off (Arriagada, Sills, Ferraro, & Pattanayak, 2015). The landowner will only apply when their expected gains from participation (including the payment), covers their opportunity and transaction costs (taking into account any intrinsically motivated conservation that would occur without payment). McCann, Colby, Easter, Kasterine, and Kuperan (2005, p. 530) define transaction costs as “the resources used to define, establish, maintain, and transfer property rights.” Transaction costs for ecosystem service sellers, who are often landowners, are determined by factors including what legal paperwork is required, negotiation of the PES contract, and any costs that they must bear for assessment and monitoring of ecosystem services (Tacconi, 2012).

Transaction costs can have a significant impact on whether a landowner will participate in a PES program. In particular, large upfront transaction costs (e.g., to create a management plan) can be a barrier to participation for cash-constrained landowners. When transactions costs have a large fixed component per property or per contract, that effectively increases the per hectare cost for relatively small properties or contracts (Heimlich, 2005). Thus, transactions costs can be a larger barrier for poorer owners of smaller properties, resulting in inequitable access to PES (Wunder, 2008; Zbinden & Lee, 2005).

Land Tenure

PES programs often limit participation to landowners with formal land tenure (Grieg-Gran, Porras, & Wunder, 2005). In many developing countries, poor landowners and smallholders are more likely to have informal land tenure. Thus, PES programs that require formal land tenure tend to exclude poor landowners and smallholders (Wunder, 2008; Zbinden & Lee, 2005).

Equity: Distribution of Participation

In the context of PES, equity of outcomes depends on the distribution of payments, as they directly influence the net benefits received by each landowner (Brown & Corbera, 2003). The distribution of payments to (or participation of) disadvantaged groups, including poor and smallholder households, is of particular interest, given the expectation that PES offer fair compensation to poor ecosystem service providers. Of course, the distribution of payments is a direct function of who is able to apply to the program (i.e., equity of access). The net benefits of participation are expected to be positive (or at least non-negative) when participation in PES is voluntary. Arriagada et al. (2015) point out that these benefits may include intangibles like greater land security and the “warm glow” of being recognized for contributing to conservation. While these intangibles are difficult to measure, there is a small body of evidence on how participation affects other socioeconomic outcomes.

Participation of disadvantaged groups varies across PES programs (Calvet-Mir et al., 2015; Wunder, 2008). In a PES program in Nicaragua that provided payments to landowners to adopt silvopastoral practices, program evaluations found that there was stronger participation by poor compared to non-poor landowners in the first two years (Pagiola et al., 2007). Conversely, program evaluations of PES programs in Costa Rica and Madagascar found some evidence that poor landholders were less likely to participate than landholders with high incomes (Porras, Barton, Miranda, & Chacón-Cascante, 2013; Sommerville, Jones, Rahajaharison, & Milner-Gulland, 2010; Zbinden & Lee, 2005). To date, there is very limited evidence on whether PES effectively alleviates poverty or has other impacts on the welfare of participants (Alix-Garcia & Wolff, 2014; Pattanayak et al., 2010, Samii et al., 2014). In Mexico, Alix-Garcia et al. (2012) find that PES has the greatest potential to reduce poverty where it has the least potential to achieve additional conservation. The socioeconomic impacts of the Chinese Sloping Lands Conversion Program have also been evaluated. Uchida et al. (2009) find that participation shifted household labor allocation from on-farm to off-farm, and Liu et al. (2010) find that participation increased household income by 14% on average.

Zilberman et al. (2008) suggest that the type of PES program affects whether the poor are likely to participate and benefit from payments. Specifically, they argue that land diversion PES programs that require forgoing productive land use are likely to benefit mostly larger landowners and only include the poor who own land that is marginal for agriculture but with high potential ecosystem service provision. Diversion programs also may reduce the demand for labor, while working land programs may increase the demand, with important implications for smallholders who earn income from off-farm labor (Zilberman et al., 2008).

PES in Practice: Costa Rica Case Study

As the longest running PES program in the developing world, Costa Rica’s Pago de Servicios Ambientales (PSA) program has been the subject of numerous studies, most focusing on the first 10 years of the forest protection program. Of the various PSA programs (which have included reforestation, forest management, and agroforestry), the greatest number and area of contracts have been issued for forest protection. The initial design of the forest protection program was criticized for lacking cost-effectiveness, based on impact evaluations that showed little additionality, and being inequitable, based on participation biased toward relatively wealthy landowners with large properties. In the last 10 years, the program administrator, the Fondo Nacional de Financiamiento Forestal (FONAFIFO), has modified the program to target both high-priority areas for ecosystem services and disadvantaged landowners.

This case study begins with an overview of the first decade of PSA, focusing on the forest protection program, which is the subject of most of the published literature on PSA. The criticism of this program at least partly contributed to institutional changes made by FONAFIFO in the second decade of the program—specifically differentiated payments introduced in 2009 and a system for ranking applications introduced in 2011. Finally, this case study relates these criticisms and institutional changes to trends in program participation over time, considering both where and who has received contracts.

Origins of the Program

In the 1970s there was growing concern that Costa Rican timber supplies were declining. This led to tax rebates as incentives to increase the area of timber plantations. These tax incentives were expanded in 1986 to include other landowners beyond large companies. In 1995, the incentives were expanded to forest protection, contributing to over 150,000 hectares of forest being conserved (Pagiola, 2008; Zbinden & Lee, 2005).

These programs established a precedent for PES, and in 1996 PSA was created through Forestry Law 7575, which recognizes four ecosystem services made available by forests: carbon fixation, watershed protection, biodiversity protection, and the protection of scenic beauty (Castro, Tattenbach, Gamez, & Olson, 2000). Because these services have no traditional markets, the law introduced a system of direct payments to reward landowners for the ecosystem services their forests provide (Phillips, 2006). The payments are based on contracts that require either protection of existing forest or the establishment of new forests (plantations) in all or portions of properties. The exact level and distribution of payments, as well as the requirements for landowners, vary across the programs of PSA. As of 2014, there were 15 programs, with the largest budgets for various reforestation and forest protection programs.

Forest Protection Program (Prior to 2011)

In terms of the number of contracts, the forest protection program is the largest in PSA. The forest protection program establishes multi-year contracts with landowners and pays a flat, per hectare fee to conserve and protect forest resources as well as the ecosystem services they provide.

Administration, Budget, and Application Process

FONAFIFO is the administrator of all PSA programs. The central office is located in San Vincente de Moravia, and there are eight regional offices located throughout Costa Rica. The regional offices are responsible for a variety of tasks, including disseminating the rules and guidelines set forth by the central office.

FONAFIFO's budget for the forest protection program is mainly funded through revenues from a fossil fuel sales tax. From 2001 to 2005, the program also received support from a loan provided by the World Bank and a grant provided by the Global Environment Facility (GEF). Additional funding is provided by the Instituto Costarricense de Electricidad, a government-run electricity and telecommunications provider, and KfW, a German development bank (Pagiola, 2008; Sanchez, 2015). Prior to 2011, the national FONAFIFO office would establish a budget ceiling for each regional office based on the historic, regional allocation of forest protection contracts. The regional offices then dispersed contracts on a "first-come, first-served" basis. As the demand for contracts was often higher than the budget, the regional offices would accept contracts until their allocation was exhausted. If the budget in one region was exhausted before others, a reallocation of funds could be considered (Sanchez, 2015). To submit an initial application for a forest protection contract, a landowner has three options. The first option is to file an application for themselves. Although all contracts are established between private landowners[6] and FONAFIFO, the second option is for landowners to file for a contract with FONAFIFO through an intermediary organization, typically non-governmental organizations or cooperatives registered with the National Forestry Office (ONF). Intermediary organizations help landowners in all phases of the application process and in implementing the conditions of the contract (Bosselmann & Lund, 2013). The third option is to file through a registered forester (or "forest regent"), who is certified by the Agronomists Board (CIA) and approved by FONAFIFO to create management plans and to monitor their implementation on land enrolled in the program. These foresters can be independent or employed by an institution such as an intermediary organization (Bosselmann & Lund, 2013; FONAFIFO, 2018).

Once the application is screened against legal requirements, the landowner must submit further technical information, which requires the expertise of a registered forester to develop, certify, and register a forest management plan with FONAFIFO. Landowners who filed their application directly to FONAFIFO must find a registered forester. However, landowners who filed through an intermediary organization typically are provided a forester by the organization (Bosselmann & Lund, 2013). While the initial application is free, the technical assistance of the forester is often associated with a fixed fee (Chacon, 2015). The foresters and intermediary organizations that help landowners through the application process can officially receive a maximum of 18% of the contract value in payment for their services (Chacon, 2015; Matulis, 2016). While some organizations and foresters may charge less, there is some evidence foresters sometimes charge more (Matulis, 2016). At the end of the contract period, landowners are given the option to reapply for participation in the program.

Criticisms of the PSA Forest Protection Program

There have been numerous evaluations of the PSA forest protection program, including both process evaluations (e.g., Hartshorn, Ferraro, & Spergel, 2005) and empirical evaluations of causal impacts on forest cover (Arriagada et al., 2012; Daniels, Bagstad, Esposito, Moulaert, & Rodriguez, 2010; Robalino & Pfaff, 2013). In order to establish attribution, these impact evaluations must consider both selection into the program and the outcome of participation in the program. As a result, they have raised concerns about both cost-effectiveness and equity.

Impact Evaluations

At the landowner level, Zbinden and Lee (2005) surveyed landowners in northern Costa Rica found that PSA participants had 61% of their farm under forest, compared to 21% for non-participants. However, Miranda et al. (2003) reported that many PSA participants in the Virilla Watershed claimed they would have protected their forests even in the absence of payments. In the Osa Peninsula, Sierra and Russman (2006) found that participation in PSA did not affect total forest cover but did positively affect regrowth (*charral*). To account for self-selection into the program, Sills et al. (2006) and Arriagada et al. (2012) applied matching techniques to survey data from participants who signed contracts in 1997 and 1998 and non-participants in the same region. They found that issuing a PSA contract on a farm led to a 10% to 15% increase in forest cover on the entire farm.

Findings from evaluations of forest cover that use pixels or jurisdictions as the unit of analysis are more negative. Sánchez-Azofeifa, Pfaff, Robalino, and Boomhower (2007) found that deforestation rates from 1997 to 2000 were not significantly lower in areas that received payments compared to those that did. Similarly, Robalino et al. (2008) found that forest protection contracts in place from 2000 to 2005 only mitigated deforestation on about 0.4% of the parcels enrolled. Robalino and Pfaff (2013) reached a similar conclusion: during the 1997 to 2000 time period, the PSA program only had a 0 to 0.002% impact on forest cover (Robalino & Pfaff, 2013).

In sum, impact evaluations have generally found small and/or statistically insignificant impacts of PSA on forest cover. One key reason for this is that deforestation rates were already low prior to the implementation of PSA (0.06% per year in the period from 1986 to 1997), meaning that there was not much deforestation that could be reduced by the program. While PSA may have helped keep deforestation rates low, the challenge is disentangling its impacts from the impacts of prior programs (Robalino et al., 2008).

Cost-Effectiveness

The most consistent critique of the PSA program has been low additionality. It is worth noting that this critique is based on a particular definition of additionality based on forest cover, which can be measured using remote sensing. However, the stated objective of PSA is to protect ecosystem services, not forest cover. In this context, additionality means that a contract results in more ecosystem services provided by the participating property than would have been the case without the PSA contract.

One fundamental reason for low additionality in PES programs is information asymmetries, which allow for adverse self-selection. The landowner's planned land use without PES and therefore his opportunity cost of participation in PES is private information, creating an information asymmetry with the FONAFIFO. FONAFIFO cannot determine the opportunity cost of each landowner due to budget constraints and the landowner's incentive not to reveal this information (Chan, Laplagne, & Appels, 2003; Ferraro, 2008). Further, the legal context for PSA may prevent FONAFIFO from offering differentiated fees, even if they had sufficient information. The combination of a flat rate payment and adverse self-selection means that participants in the program can capture substantial informational rents. Participants whose costs of participation are lower than the payment levels, including those who would have conserved forest in the absence of payments and thus face zero opportunity cost, will self-select into the program. These landowners with opportunity and transaction costs below the payment level extract informational rents

(Deng & Xu, 2015; Ferraro, 2008; Salas, Roe, & Sohngen, 2012). Informational rents are often considered a source of inefficiency, in that the implementing agency generates fewer ecosystem services per dollar than it would if landowners were paid their true opportunity costs[7] (Deng & Xu, 2015; Ferraro, 2008).

Equity in Participation

The distribution of participation in PSA has been considered both as a first step in impact evaluations and in studies specifically on this topic. The concern typically raised is that relatively wealthy landowners with larger properties capture more of the program funds than relatively poorer landowners with smaller properties because of transaction costs and other barriers to participation (Legrand, Froger, & Le Coq, 2013; Porras et al., 2013; Zbinden & Lee, 2005). This undermines the claim that PSA is a mechanism for compensating the rural poor for providing ecosystem services benefiting the broader population.

Transaction Costs

In the context of PSA, transaction costs are most commonly incurred to obtain a land title or other necessary paperwork and to contract a registered forester who develops the management plan to be submitted after the acceptance of the initial application. The registered foresters are professionally responsible for on-the-ground implementation of PSA. Thus, they typically charge both an upfront fee for the management plan and a fraction of the contract amount (up to 18%). Independent foresters who seek out their own clients tend to prefer working on larger contracts that will generate the most revenue (Matulis, 2016). While this is not necessarily true of intermediary organizations, many of them focus on PSA programs other than forest protection in order to extend their work with existing clients and with farmers (Chacon, 2015).

Distribution of Payments

The percent of program dollars awarded to smallholders is one index of outcome equity in PSA. Figure 2 shows the percent of dollars awarded on largeholdings (using FONAFIFO’s definition of largeholdings as greater than 50 hectares). The 2014 agricultural census in Costa Rica shows that only 10% of all properties are greater than 50 hectares (INEC, 2014).[8] In stark contrast, from 2005 to 2014, more than 85% of the forest protection budget for new contracts was allocated to properties larger than 50 hectares. This intuitively makes sense: there is more land than can be enrolled on larger properties, meaning that more funds will flow to largeholdings. This distribution of funds could also indicate that largeholdings are managed less intensively, resulting in lower opportunity costs of forest conservation. On the other hand, the costs of participation may be a greater barrier for smallholders, who are also likely to be relatively poor. Wealthier landowners with larger properties can better afford the transactions costs associated with obtaining a PSA contract both because they are more capitalized and because their transactions costs are lower on a per hectare basis.

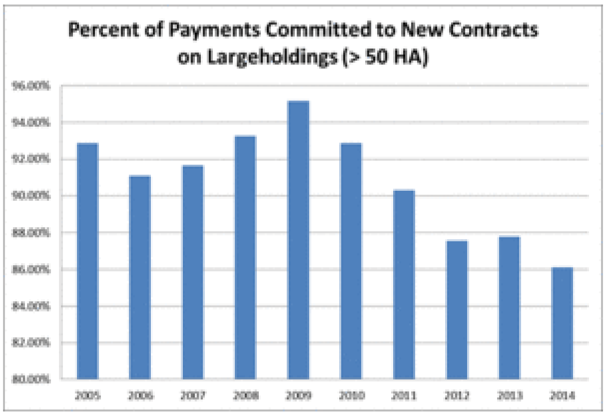

Figure 2. Percent of new dollars awarded in each year that were allocated to contracts on properties >50 hectares. (Based on data from FONAFIFO.)

Addressing Critiques

Most evaluations of PSA have called for Costa Rica to take steps to increase the environmental services provided per dollar spent. To directly address the issue of adverse selection, FONAFIO would need information about the opportunity cost of each landowner and knowledge about whether or not they would conserve without payments. However, if FONAFIFO tried to obtain this information, landowners could try to signal that they have higher opportunity costs compared to their true opportunity cost or that they have an intent to deforest when they actually do not. Given these difficulties and the cost of collecting this information, FONAFIFO has instead focused on increasing additionality by targeting contracts to properties likely to generate high-value ecosystem services and increasing social equity by encouraging participation by disadvantaged landowners.

Targeting has often been suggested as a way to increase the cost-effectiveness of PES programs (Hartshorn et al., 2005; Pagiola, 2008; Porras et al., 2013; Sánchez-Azofeifa et al., 2007; Wunscher, Engel, & Wunder, 2008). Targeting refers to any method of identifying and actively encouraging enrollment of prioritized areas (often areas deemed environmentally sensitive) in a voluntary conservation program (Alix-Garcia, De Janvry, & Sadoulet, 2008; Norden et al., 2013). While this approach does not directly address the issues associated with adverse self-selection, it does focus program dollars on areas where they can potentially generate a higher quality or quantity of ecosystem services. There are several possible targeting mechanisms. One is to offer higher payments to priority areas. Often referred to as differentiated payments, this allows payments to be adjusted based on any factors of concern to the program administrator, such as biodiversity, carbon, and deforestation risk (Alix-Garcia et al., 2008; Salzman, 2009). When programs are oversubscribed, another mechanism is to prioritize applications (e.g., through scoring) for areas that provide the most valuable ecosystem services.

Differentiated Payments

In theory, differentiated payments allows for landowners who provide higher-value ecosystem services to be paid a higher price per hectare. Rather than assessing the value of ecosystem services for each parcel of land offered to the forest protection program, FONAFIFO created two new programs[9] in 2009 under the forest protection umbrella that offer higher per hectare rates where forest protection is expected to generate higher-value ecosystem services compared to the general forest protection program (Table 2). The first is the protection of hydric resources, which conserves forests protecting hydrological resources in priority river basins (e.g., that generate hydroelectricity). The second is forest protection in conservation gaps, which conserves forests in buffer zones around protected areas or designated biological corridors (FONAFIFO, 2018).

Table 2. Evolution of Total Payment Per Hectare and Number of Years Paid for the Main Forest Protection Program and Two Programs for Priority Areas Established in 2009

		Payment Per Hectare		
Year	**Contract Length (years)**	**Forest Protection**	**Conservation Gaps**	**Protection of Water Resources**
2009	5	320	375	400
2010	5	320	375	400
2011	5	320	375	400
2012	10	640	750	800
2013	10	640	750	800
2014*	5	169890(~318)	188937(~354)	202000(~366)

Notes: * In 2014, the payments were defined in Costa Rican Colónes. The number in parentheses is the approximate USD value, based on an exchange rate of 1 USD = 533.5 colónes.

As with the general program, payments for each contract are distributed evenly over the length of the contract, with either 1/5 or 1/10 paid when the contract is signed. The total payment per hectare for conservation gaps was set 17% higher than the total payment for the forest protection program, while the payment level for water resources was determined by how much hydroelectric companies were willing to pay (Sanchez, 2015).

The Matrix for Evaluating Applications to PSA

Another significant innovation in the forest protection program was the centralization of the application process and the introduction of a scoring system called the *matriz*, or Matrix, in 2011. Prior to 2011, applications were accepted on a first-come, first-served basis in each region. Because there was not a sufficient budget to accept all applications, ranking applications based on their potential provision of ecosystem services was an obvious way to increase cost-effectiveness. Reflecting the dual environmental and social objectives for the program, the Matrix, which was put in place in 2011 and updated in 2012, prioritizes applications based on both environmental and social objectives. However, it does not explicitly consider additionality in the sense of providing additional ecosystems services compared to what would have been provided without payments.

The Matrix, in use since 2012 (Table 3), has three environmental criteria that assign points based on whether a forest is located in priority areas, including protected areas, conservation gaps, and biological corridors. The Matrix also awards points to applications for new contracts on properties that previously had a forest protection contract. In addition to the environmental priorities, the Matrix also addresses social objectives by awarding additional points to applications for contracts on properties that are small or located in districts of lower socioeconomic standing based on the Social Development Index (IDS). The IDS is calculated by the Ministry of National Planning and Economic Policy (MIDEPLAN) approximately every five years as a measure of the socioeconomic level of each district. It is used by the Costa Rican government for social policy and budget allocation (Porras et al., 2013). The IDS is constructed from a set of 11 socioeconomic indicators grouped into four areas: health, electoral participation, education, and economics (Figure 3). The score is standardized to range between 0 and 100, with 100 indicating the highest socioeconomic status. The median IDS[10] score of districts with new forest protection contracts was just over 40 for the decade from 2005 to 2014 (Figure 4). The rationale for prioritizing applications from districts with an IDS score lower than 40 is that this will increase participation by poor landowners without requiring data on the socioeconomic status of each applicant.

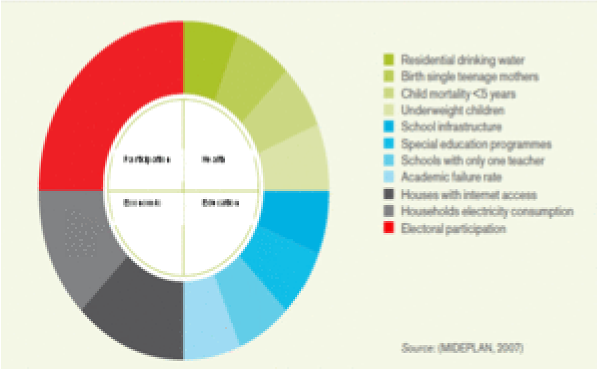

Figure 3. Factors included in the Social Development Index (IDS).

(*Source*: Porras et al., 2013.)

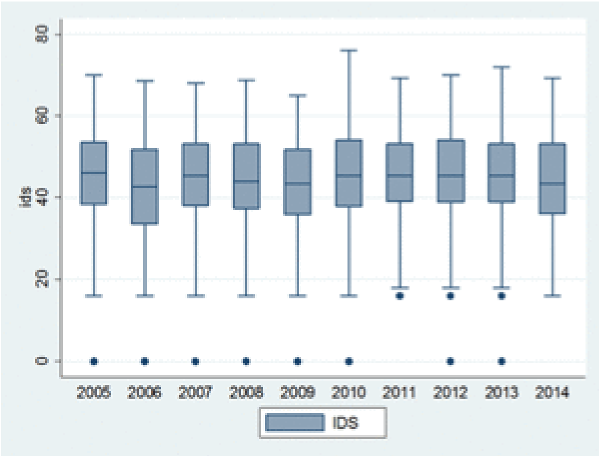

Figure 4. Boxplot of IDS scores for new Forest Protection contracts awarded in each year. The horizontal line represents the group median. The length of the box represents the interquartile range (IQR). The whiskers represent the minimum and maximum values. The dots represent outlier values (more than 1.5 times the IQR outside the IQR) (based on data from FONAFIFO).

To address concerns that smallholders and poor landowners were not receiving many PSA contracts, the Matrix awards 25 points to applications for contracts on properties that are small (<50 ha) and 10 points for properties located in districts where the IDS is less than 40.[11]

Table 3. Matrix Scoring System Used for Forest Protection Contracts Starting in 2012

Criteria	Priorities	Points
1	Forests on properties located in Conservation Gaps within Indigenous Territories	85
2	Forests on properties located within officially established Biological Corridors, or forests that have been declared important for protecting water resources	80
3	Forests on properties in Protected Areas that have not yet been purchased or expropriated	75
4	Forests outside any of the above priorities	55
I	Forests previously enrolled in the Forest Protection program, with contracts in good standing and expiring in the year of application	10 additional
II	Forests on properties in districts rated below 40 on the Índice de Desarrollo Social	10 additional
III	Forests on properties of 50 hectares or less	25 additional

Trends in PSA Participation

To address concerns about cost-effectiveness and equity, FONAFIFO has introduced both differentiated payments and ranking of applications for the forest protection program. Both changes can impact a landowner's decision to participate in the program. For a landowner with property in a prioritized area, those who own less than 50 hectares, or those with land in districts with an IDS less than 40, the higher potential payments through the specialized programs and additional points through the Matrix may increase their desire to participate and submit an application. Additionally, both institutional changes may encourage government officials, intermediary organizations, and foresters to focus their efforts to disseminate information about the program and recruit applicants who have properties in priority areas and smallholders. While there are reasons to believe that these changes in PSA introduced by FONAFIFO would increase cost-effectiveness and equity, it is an empirical question whether the cumulative effect of these institutional changes actually shifted the distribution of contracts toward higher-priority areas and disadvantaged landowners. This case study provides a starting point for this empirical analysis by describing trends in participation in PSA.

Database

Data for this case study are from FONAFIFO. FONAFIFO keeps records on all contracts, including the type of contract, the location of the property, the size of the property, and the number of hectares enrolled in PSA. This information is split across FONAFIFO's application database and Integrated Project Administration System, which they use to manage payments. These data can also be used to re-construct the Matrix score of each contract.

Cost-Effectiveness

Figure 5 shows the number of new contracts awarded within priority areas by year. In 2009, when differentiated payments were first introduced, there is a noticeable increase in the number of new contracts being awarded in priority areas, after a decreasing trend from 2007 to 2009. This increase continued through 2012, thus spanning the introduction of the Matrix in 2011. From 2013 to 2014 there was a decline in the number of new contracts awarded to landowners in priority areas. To control for variation in FONAFIFO's annual budget, Table 4 lists the percent of new contracts signed each year from 2005 to 2014 that lie inside priority areas. This shows little interannual variation. There is a low point (83%) in 2008, the year prior to the introduction of differentiated payments, and a high point (93%) in the following year. New contracts awarded for land in prioritized areas then declined from 93% in 2009 to about 85% in 2014. While it is possible that the portion of contracts awarded in prioritized areas would have declined even more without differentiated payments or the Matrix, there is no evidence suggesting that would have been the counterfactual trend.

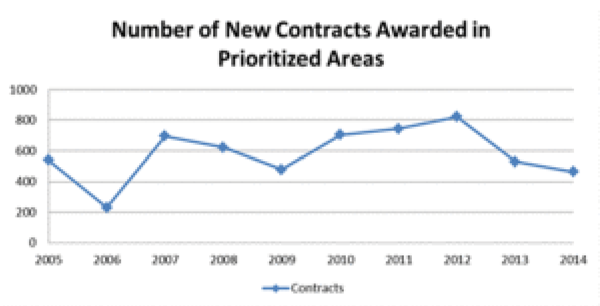

Figure 5. The number of new contracts awarded each year in priority areas (protected areas, conservation gaps, or biological corridors, including their overlapping areas). (Based on data from FONAFIFO.)

Table 4. Percent of Contracts Awarded to Landowners in Priority Areas, as Defined by 2012 Scoring Matrix

Year	Percent
2005	86%
2006	89%
2007	88%
2008	83%
2009	93%
2010	90%
2011	88%
2012	89%
2013	85%
2014	85%

Equity in Participation

The Matrix provides an advantage to smallholders and owners with land in districts with low socioeconomic status. This scoring mechanism can be seen as a means to address equity by shifting the distribution of participants toward landowners with low socioeconomic status. While it does not directly affect equity of access, it may influence where and who foresters and intermediary organizations recruit into PSA. Additionally, while differentiated payments were introduced through specialized forest protection programs intended to increase the ecosystem services generated by areas under contract (i.e., increase cost-effectiveness), they may have also affected participation by smallholders and poorer landowners. Higher per hectare payments could cover the transaction costs for landowners who would otherwise opt out of the program, thus lowering barriers to entry and potentially increasing equity in access. This is consistent with Alix-Garcia et al. (2008), who found that in the Mexican PES program, the poor receive a smaller portion of the total PES budget under a flat rate scheme compared to a differentiated payment scheme. To assess whether the differentiated payments could have had the same effect in Costa Rica, Figure 6 presents the percent of new contracts awarded on properties in four size classes: less than 25 hectares (red solid line), 25 to 50 hectares (dashed red line), 50 to 100 hectares (dashed black line), and more than 100 hectares (solid black line).

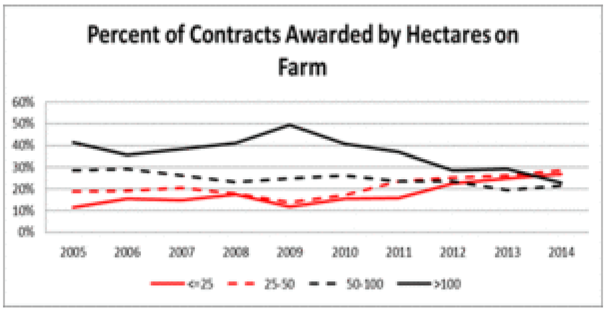

Figure 6. The percent of new contracts awarded on properties of different sizes: less than 25 hectares (solid red line), between 25 and 50 hectares (dashed red line), between 50 and 100 hectares (dashed red line), and greater than 50 hectares (solid black line). (Based on data from FONAFIFO.)

From 2005 to 2013, a larger percent of contracts were awarded on largeholdings (>50 hectares) than on smallholdings (<50 hectares). However, starting in 2011, when the Matrix was introduced, the gap between the percent of contracts awarded to largeholders and smallholders narrows. In 2013 and 2014, smallholders are awarded more contracts than largeholders. This temporal trend is consistent with the hypothesis that the introduction of the Matrix had a positive effect on smallholder participation in the overall forest protection program.

In addition to targeting small properties, the Matrix also awards additional points to landowners in districts with an IDS below 40. Figure 7 shows the percent of contracts awarded to landowners based on the district's IDS score, and Figure 8 presents a map of Costa Rica's districts, indicating their IDS scores. Slightly less than 6% of Costa Rica's districts have IDS scores less than 20, while slightly more than 20% have IDS scores greater than 60.

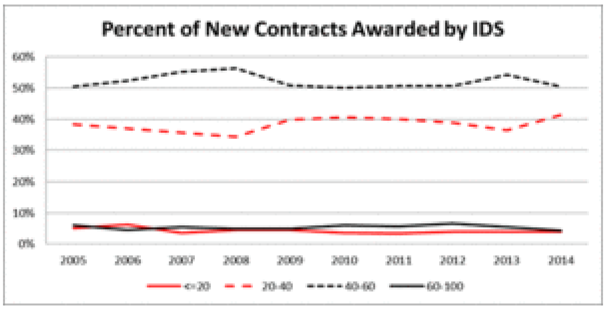

Figure 7. The percent of new contracts awarded on properties in districts at different levels of socioeconomic development: IDS less than 20 (solid red line), between 20 and 40 (dashed red line), between 40 and 60 (dashed red line), and greater than 60 (solid black line). (Based on data from FONAFIFO.)

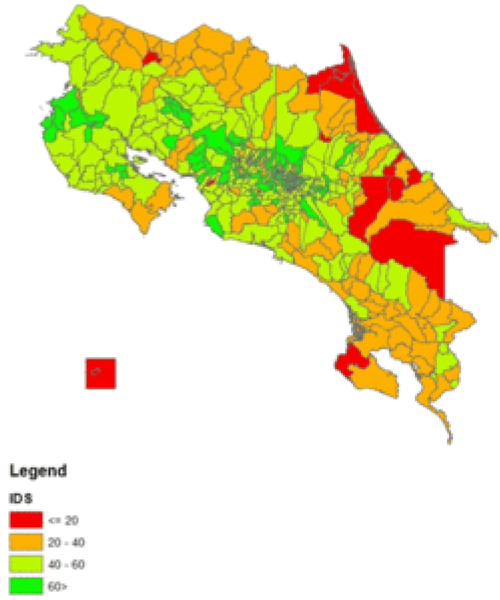

Figure 8. IDS score of districts in Costa Rica, from less than 20 (in dark red) to greater than 60 (in dark green).

Thus, it is not surprising that landowners in districts with an IDS score between 40 and 60 receive the largest percent of new contracts. Figure 7 also shows a trend (from 2009 to 2012) toward more contracts being awarded in districts at the lower end of that range, with IDS scores between 20 and 40. However, overall the trends do not suggest any obvious influence of the differentiated payments or the Matrix on participation by landowners in low-IDS districts.

Conclusions

Payments for ecosystem services have been touted as both more cost-effective and more equitable than other tools for conserving and generating ecosystem services. However, evaluations of PES in developing countries have questioned whether these advantages are actually realized when PES programs are implemented. The lack of additionality, the difficulty of imposing conditionality, as well as possible price and behavioral spillover are often discussed as impediments to cost-effectiveness. There are also concerns that significant barriers to participation, such as a large fixed transaction cost per contract, prevent the poor from reaping the potential financial benefits of participation.

The case of Costa Rica illustrates and provides context for many of the characteristics and concerns regarding PES programs, including the possibility of institutional evolution. The Costa Rican government (FONAFIFO) responded to concerns about the program within the constraints imposed by asymmetric information by targeting payments through differentiated payments introduced in 2009 and through a system for ranking applications introduced in 2011. Trends in participation in the program are consistent with the desired effect of those institutional change on participation by smallholders but provide no evidence that more contracts are being issued in priority areas for ecosystem services or districts with low levels of socioeconomic development.

While there has been a great deal of research on the PSA program and PES programs in general, there remain important questions for future work. In the realm of cost-effectiveness, key questions include: (1) What is the role of the additionality requirement in PES programs, and what mechanisms could be used to increase additionality? (2) What is the level of compliance with PES contracts, and what tools can be used to improve monitoring and enforcement? And (3) How can programs assess and manage spillover effects to maximize potential positive effects and minimize potential negative effects? In the realm of social equity, future work should identify barriers to participation that create inequitable access to PES programs. Future evaluations should consider social and environmental objectives in the same framework, e.g., compared to the same counterfactual scenario, in order to further understand the trade-offs between social and environmental objectives.

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Further Reading

Alix-Garcia, J. M., & Wolff, H. (2014). Payment for ecosystem services from forests. *Annual Review of Resource Economics*, *6*(1), 361–380.

Engel, S. (2016). The devil in the detail: A practical guide on designing payments for environmental services. *International Review of Environmental and Resource Economics*, *9*(1–2), 131–177.

Ferraro, P. J & Kiss, A. (2002). Direct payments to conserve biodiversity. *Science, 5599*(298), 1718–1719.

Jack, B. K., Kousky, C., & Sims, K. R. (2008). **Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms**. *Proceedings of the National Academy of Sciences*, *105*(28), 9465–9470.

Landell-Mills, N. & Porras, I. (2002). *Silver bullet or fools' gold?: A global review of markets for forest environmental services and their impact on the poor*. London: International Institute for Environment and Development.

Milder, J., Scherr, S., & Bracer, C. (2010). Trends and future potential of payment for ecosystem services to alleviate rural poverty in developing countries. *Ecology and Society*, *15*(2), 4.

Porras, I., Barton, D. N., Miranda, M., & Chacón-Cascante, A. (2013). *Learning from 20 years of payments for ecosystem services in Costa Rica*. London: International Institute for Environment and Development.

Schomers, S., & Matzdorf, B. (2013). Payments for ecosystem services: A review and comparison of developing and industrialized countries. *Ecosystem Services*, *6*, 16–30.

References

Alix-Garcia, J., De Janvry, A., & Sadoulet, E. (2008). The role of deforestation risk and calibrated compensation in designing payments for environmental services. *Environment and Development Economics*, *13*, 375–394.

Alix-Garcia, J. M., Shapiro, E. N. & Sims, K. R. (2012). Forest conservation and slippage: Evidence from Mexico's national payments for ecosystem services program. *Land Economics*, *88*(4), 613–638.

Alix-Garcia, J. M., & Wolff, H. (2014). Payment for ecosystem services from forests. *Annual Review of Resource Economics*, *6*(1), 361–380.

Alpizar, F., Norden, A., Pfaff, A., & Robalino, J. (2013). *Effects of exclusion from a conservation policy: Negative behavioral spillovers from targeted incentives*. Duke Environmental and Energy Economics Working Paper Series, 1–23.

Alston, L. J., Andersson, K., & Smith, S. M. (2013). Payment for Environmental Services: Hypotheses and Evidence. *Annual Review of Resource Economics*, *5*, 139–59.

Arriagada, R., Ferraro, P., Sills, E., Pattanayak, S., & Cordero-Sancho, S. (2012). Do payments for environmental services affect forest cover? A property-level evaluation from Costa Rica. *Land Economics*, *88*(2), 382–399.

Arriagada, R. A., Sills, E. O., Ferraro, P. J., & Pattanayak, S. K. (2015). Do payments pay off? Evidence from participation in Costa Rica's PES program. *PloS one*, *10*(7), e0131544.

Bennett, K. (2010). Additionality: The next step for ecosystem service markets. *Duke Environmental Law & Policy Forum*, *20*, 417–438.

Börner, J., Baylis, K., Corbera, E., Ezzine-de-Blas, D., Honey-Rosés, J., Persson, U. M., & Wunder, S. (2017). The effectiveness of payments for environmental services. *World Development*, *96*, 359–374.

Bosselmann, A. S., & Lund, J. F. (2013). Do intermediary institutions promote inclusiveness in PES programs? The case of Costa Rica. *Geoforum*, *49*, 50–60.

Braat, L. (2016). **Ecosystem Services**. *Oxford Research Encyclopedia of Environmental Science*.

Brown, K., & Corbera, E. (2003). Exploring equity and sustainable development in the new carbon economy. *Climate Policy*, *3*(Suppl. 1), S41–S56.

Bulte, E. H., Lipper, L., Stringer, R., & Zilberman, D. (2008). Payments for ecosystem services and poverty reduction: Concepts, issues, and empirical perspectives. *Environmental and Development Economics*, *13*, 245–254.

Calvet-Mir, L., Corbera, E., Martin, A., Fisher, J., & Gross-Camp, J. (2015). Payments for ecosystem services in the tropics: A closer look at effectiveness and equity. *Current Opinion in Environmental Sustainability*, *14*, 150–162.

Castro, R., Tattenbach, F., Gamez, L., & Olson, N. (2000). The Costa Rican experience with market instruments to mitigate climate change and conserve biodiversity. *Environmental Monitoring and Assessment*, *61*(1), 75–92.

Chan, C., Laplagne, P., & Appels, D. (2003). *The role of auctions in allocating public resources*. Productivity Commission Staff Research Paper, Productivity Commission, Melbourne, 1–144.

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

Corbera, E., Brown, K., & Adger, W. N. (2007). The equity and legitimacy of markets for ecosystem services. *Development and Change*, *38*, 587–613.

Chacon, G. (2015, June 18,). Personal interview with N. James, E. Sills, and A. Salas, Turrialba, Costa Rica.

Daniels, A. E., Bagstad, K., Esposito, V., Moulaert, A., & Rodriguez, C. M. (2010). Understanding the impacts of Costa Rica's PES: Are we asking the right questions? *Ecological Economics*, *69*(11), 2116–2126.

Deng, X., & Xu, Z. (2015). Green auctions and reduction of information rents in payments for environmental services: An experimental investigation in Sunan County, Northwestern China. *PloS One*, *10*(3), 1–15.

Engel, S., Pagiola, S., & Wunder, S. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics*, *65*, 663–674.

FAO. (2004). *Payment schemes for environmental services in watersheds*. Regional Forum on Payment Schemes for Environmental Services in Watersheds, 1–95.

Farley, J., & Costanza, R. (2010). Payments for ecosystem services: From local to global. *Ecological Economics*, *69*(11), 2060–2068.

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., & Kiss, A. (2002). Direct payments to conserve biodiversity. *Science*, *5599*(298), 1718–1719.

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

Ferraro, P. J. (2008). Asymmetric information and contract design for payments for environmental services. *Ecological Economics*, *65*(4), 810–821.

Ferraro, P. J. (2011). The future of payments for environmental services. *Conservation Biology*, *25*(6), 1134–1138.

FONAFIFO. (2018). ***Methods of payments for environmental services***. San Jose: Fondo Nacional de Fianciamiento Forestal.

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*(9), 1511–1527.

Hartshorn, G., Ferraro, P., & Spergel, B. (2005). *Evaluation of the World Bank—GEF Ecomarkets Project in Costa Rica*. Report prepared for the World Bank, 1–27.

Heimlich, R. E. (2005). *The policy-related transactions costs of land conservation in the United States: Evolution over time and comparison between programmes*. OECD Workshop on Policy-Related Transaction Costs, 1–40.

INEC. (2014). ***Total de fincas en hectáreas***.

James, N. A. (2018). *Costa Rica's Payments for Ecosystem Services Program: Understanding the implications of implementing targeting mechanisms and procurement auctions on cost effectiveness and equity of participation* (Dissertation). Raleigh: North Carolina State University.

Jayachandran, S., De Laat, J., Lambin, E. F., Stanton, C. Y., Audy, R., & Thomas, N. E. (2017). Cash for carbon: A randomized trial of payments for ecosystem services to reduce deforestation. *Science*, *357*(6348), 267–273.

Kinzig A. P., Perrings, C, Chapin III, F. S., Polasky, S., Smith, V. K., Tilman, D., & Turner II, B. L. (2011). Paying for ecosystem services—Promise and peril. *Science*, *334*, 603–604.

Landell-Mills, N., & Porras, I. (2002). *Silver bullet or fools' gold?: A global review of markets for forest environmental services and their impact on the poor*. London: International Institute for Environment and Development.

Legrand, T., Froger, G., & Le Coq, J.-F. (2013). Institutional performance of payments for environmental services: An analysis of the Costa Rican Program. *Forest Policy and Economics*, *37*, 115–123.

Liu, C., Lu, J., & Yin, R. (2010). An Estimation of the Effects of China's Priority Forestry Programs on Farmers' Income. *Environmental Management*, *45*(3), 526–540.

MEA (Millennium Ecosystem Assessment). (2005). *Ecosystems and human well-being: Synthesis*. Washington, DC: Island Press.

Martin, A., Gross-Camp, N., Kebede, B., & McGuire, S. (2014). Measuring effectiveness, efficiency and equity in an experimental payments for ecosystem services trial. *Global Environmental Change*, *28*, 216–226.

Matulis, B. (2016). The coercive laws of competition in a neoliberal era: the case of forestry in Costa Rica. *Journal of Political Ecology*, *23*, 279–295.

McCann, L., Colby, B., Easter, K. W., Kasterine, A., & Kuperan, K. V. (2005). Transaction cost measurement for evaluating environmental policies. *Ecological Economics*, *52*(4), 527–542.

Miranda, M., Porras, I.T., & 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*. London: International Institute for Environment and Development.

Miranda, M., Dieperink, C., & Glasbergen, P. (2006). Costa Rican environmental service payments: The use of a financial instrument in participatory forest management. *Environmental Management*, *38*, 562–571.

Miteva, D. A., Pattanayak, S. K., & Ferraro, P. J. (2012). Evaluation of biodiversity policy instruments: What works and what doesn't? *Oxford Review of Economic Policy*, *28*(1), 69–92.

Moon, K., Marshall, N., & Cocklin, C. (2012). Personal circumstances and social characteristics as determinants of landholder participation in biodiversity conservation programs. *Journal of Environmental Management*, *113*, 292–300.

Muradian, R., Corbera, E., Pascual, U., Kosoy, N., & May, P. H. (2010). Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. *Ecological Economics*, *69*(6), 1202–1208.

Namirembe, S., Leimona, B., van Noordwijk, M., Bernard, F., & Bacwayo, K. E. (2014). Co-investment paradigms as alternatives to payments for tree-based ecosystem services in Africa. *Current Opinion in Environmental Sustainability*, *6*, 89–97.

Neef, A., & Thomas, D. (2009). Rewarding the upland poor for saving the commons? Evidence from Southeast Asia. *International Journal of the Commons*, *3*(1), 1–15

Nordén, A., Persson, U., & Alpízar, F. (2013). Incentives, impacts, and behavioral issues in the context of payment for Ecosystem Services Programmes: Lessons for REDD+. In A. Bigsten (Ed.), *Globalization and development: Rethinking interventions and governance* (pp. 147–168). London: Routledge.

Pagiola, S. (2008). Payments for environmental services in Costa Rica. *Ecological Economics*, *65*(4), 712–724.

Pagiola, S., Arcenas, A., & Platais, G. (2005). Can payments for environmental services help reduce poverty? An exploration of the issues and evidence to date from Latin America. *World Development*, *33*(2), 237–253.

Pagiola, S., Ramirez, E., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., & Ruiz, J. (2007). Paying for the environmental services of silvopastoral practices in Nicaragua. *Ecological Economics*, *64*(2), 374–385.

Pagiola, S., Honey-Rosés, J., & Freire-González, J. (2016). Evaluation of the permanence of land use change induced by payments for environmental services in Quindío, Colombia. *PLoS ONE*, *11*(3), 18p.

Pascual, U., Muradian, R., Rodriquez, L. C., & Duraiappah, A. (2010). Exploring the links between equity and efficiency in payments for environmental services: A conceptual approach. *Ecological Economics*, *69*, 1237–1244.

Pascual, U., Phelps, J., Garmenda, E., Brown, K., Corbera, E., Martin, A., Gomez-Baggethun, E., & Muradian, R. (2014). Social equity matters in payments for ecosystem services. *BioScience*, *64*(11), 1027–1036.

Pattanayak, S., Wunder, S., & Ferraro, P. (2010). Show me the money: Do payments supply environmental services in developing countries? *Review of Environmental Economics and Policy*, *4*(2), 254–274.

Persson, U. M., & Alpizar, F. (2013). Conditional cash transfers and payments for environmental services. *World Development*, *23*, 124–137.

Phillips, F. K. (2006). **Costa Rica Forest Law 7575, 1996**. *Centre for International Sustainable Development Law*.

Platteau, J. (2000). *Institutions, social norms, and economic development*. London: Routledge.

Porras, I., Barton, D. N., Miranda, M., & Chacón-Cascante, A. (2013). *Learning from 20 years of payments for ecosystem services in Costa Rica*. London: International Institute for Environment and Development.

Rebelo, C. (2009). Financing for forest conservation: Payments for ecosystem services in the tropics. *Journal of Sustainable Forestry*, *28*, 279–284.

Robalino, J., Pfaff, A., Sanchez-Azofeifa, G. A., Alpízar, F., León, C., & Rodríguez, M. (2008). **Changing the deforestation impacts of ecopayments: Evolution (2000–2005) in Costa Rica's PSA program**. Duke University.

Robalino, J., & Pfaff, A. (2013). Ecopayments and deforestation in Costa Rica: A nationwide analysis of PSA's initial years. *Land Economics*, *89*(3), 432–448.

Rode, J., Gómez-Baggethun, E., & Krause, T. (2015). Motivation crowding by economic incentives in conservation policy: A review of the empirical evidence. *Ecological Economics*, *117*, 270–282.

Salas, P., Roe, B., & Sohngen, B. (2012). *Addressing additionality in REDD contracts when formal enforcement is absent*. Selected Paper No. 712, Agricultural & Applied Economics Association's 2012 AAEA Annual Meeting.

Salzman, J. (2009). *A policy maker's guide to designing payments for ecosystem services*. Asian Development Bank.

Salzman, J., Bennett, G., Carroll, N., Goldstein, A., & Jenkins, M. (2018). The global status and trends of Payments for Ecosystem Services. *Nature Sustainability*, *1*(3), 136–144.

Samii, C., Lisiecki, M., Kulkarni, P., Paler, L., & Chavis, L. (2014). Effects of payment for environmental services (PES) on deforestation and poverty in low and middle income countries: a systematic review. *Campbell Systematic Reviews*, *10*(11), 1–95.

Sanchez, O. (2015, October 15). Personal interview by Francisco Alpizar, Tabare Capitan, and Ariana Salas, San Jose, Costa Rica.

Sánchez-Azofeifa, G. A., Pfaff, A., Robalino, J., & Boomhower, J. (2007). Costa Rica's Payment for Environmental Services Program: Intention, implementation, and impact. *Conservation Biology*, *21*(5), 1165–1173.

Sierra, R., & Russman, E. (2006). On the efficiency of environmental service payments: A forest conservation assessment in the Osa Peninsula, Costa Rica. *Ecological Economics*, *59*:131-141.

Sills, E., Arriagada, R., Ferraro, P., Pattanayak, S., Carrasco, L., Ortiz, E., . . . Andam, K. (2006). ***Impact of the PSA Program on land use***. Paper presented at the Workshop on Costa Rica's Experience with Payments for Environmental Services, San José, California, September 25–26, 2006.

Sommerville, M., Jones, J. P. G., Rahajaharison, M., & Milner-Gulland, E. J. (2010). The role of fairness and benefit distribution in community-based Payment for Environmental Services interventions: A case study from Menabe, Madagascar. *Ecological Economics*, *69*(6), 1262–1271.

Sunderlin, W., Dewi, S., Puntodewo, A., Muller, D., Angelsen, A., & Epprecht, M. (2008) .Why forests are important for global poverty alleviation: a spatial explanation. *Ecology and Society*, *13*(2), 24.

Tacconi, L. (2012). Redefining payments for environmental services. *Ecological Economics*, *73*, 29–36.

Tallis, H., Kareiva, P., Marvier, M., & Chang, A. (2008). An ecosystem services framework to support both practical conservation and economic development. *Proceedings of the National Academy of Sciences of the United States of America*, *105*(28), 9457–9464.

TEEB. (2010). *The economics of ecosystems and biodiversity: Mainstreaming the economics of nature: A synthesis of the approach, conclusions and recommendations of TEEB*. Malta: Progress Press.

Uchida, E., Rozelle, S., & Xu, J. (2009). Conservation payments, liquidity constraints and off-farm labor: impact of the Grain for Green program on rural households in China. *American Journal of Agricultural Economics*, *91*(1), 70–86.

Wegner, G. (2015). Payments for ecosystem services (PES): A flexible, participatory, and integrated approach for improved conservation and equity outcomes. *Environment Development and Sustainability*, *18*(3), 617–644.

Wunder, S. (2005). *Payments for environmental services: Some nuts and bolts*. CIFOR Occasional Paper No. 42. Center for International Forestry Research, Bogor, Indonesia.

Wunder, S. (2006). Between purity and reality: taking stock of PES schemes in the Andes. *Ecosystem Marketplace*, *1*(4), 1–2.

Wunder, S. (2007). The efficiency of payments for environmental services in tropical conservation. *Conservation Biology*, *21*(1), 48–58.

Wunder, S. (2008). Payments for environmental services and the poor: Concepts and preliminary evidence. *Environmental and Development Economics*, *13*, 279–297.

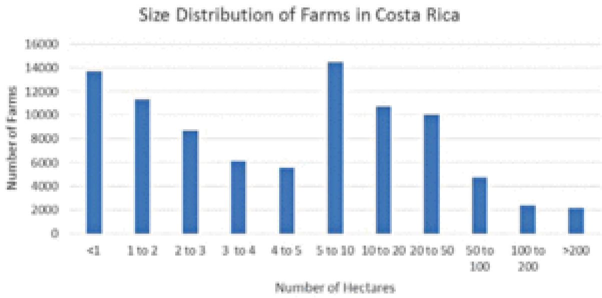
Wunder, S., Engel, S., & Pagiola, S. (2008). Taking stock: A comparative analysis of payments for environmental services programs in developed and developing countries. *Ecological Economics*, *65*, 834–852.

Wunscher, T., Engel, S., & Wunder, S. (2008). Spatial targeting of payments for environmental services: A tool for boosting conservation benefits. *Ecological Economics*, *65*(4), 822–833.

Zbinden, S., & Lee, D. R. (2005). Paying for environmental services: An analysis of participation in Costa Rica's PSA Program. *World Development*, *33*(2), 255–272.

Zilberman, D., Lipper, L., & McCarthy, N. (2008). When could payments for environmental services benefit the poor? *Environmental Economics*, *13*(3), 255–278.

Appendix: Size Distribution of Farms in Costa Rica



Size Distribution of Farms in Costa Rica

Source: Data from INEC (2014).

Notes:

(1.) One important caveat to this argument is that where ecosystem service suppliers are poor, this may change the voluntary nature of PES. The poor may not be in a position to refuse payments due to their status, turning PES into a forced trade (Muradian et al., 2010).

(2.) Additionality is also key to climate change mitigation policies, including carbon offset mechanisms that are designed to reduce the cost of meeting a global target for reductions in net carbon emissions.

(3.) This does not necessarily imply an absolute decrease in conservation but rather that actual conservation is less than expected.

(4.) In a two-round dictator game, each participant was given a total of 5,000 colones ($10), which could be donated to an environmental program to conserve forests. In the first round, all participants were given the option to donate all, some, or none of their endowment to the conservation program. In the second round, another 5,000 colones was endowed to the participants, and an incentive was introduced: half of the second contribution would be returned to the landowner. However, those who had contributed more than 1,000 colones in the first round were excluded from receiving the incentive for their second contribution. Those who were excluded due to positive behavior in the past significantly reduced their contributions. Those who received the incentive increased their contributions. This experimental evidence suggests how sorting agents into participants and non-participants can affect additionality: in this case, small contributors were participants in that they received incentives to increase their additional contributions, but this caused the large contributors (the non-participants) to decrease their contributions. Thus, the increased contributions of one group may be offset by decreased contributions of the group affected by this behavioral spillover.

(5.) A third way to judge equity (not considered here) is based on the decision-making process and whether all stakeholders have a voice (Brown & Corbera, 2003).

(6.) There are also PSA contracts for large tracts owned collectively by indigenous groups

(7.) It is worth noting that while additionality is generally considered an important measure of conservation success, FONAFIFO has argued that their objective is to compensate landowners for their production of ecosystem services, without necessarily increasing the provision of those services. It is also important to note that around the time PSA was implemented, deforestation was deemed illegal in Costa Rica. Therefore, an alternative view of PSA and the informational rents is compensation for the restrictions on the use of private forested lands.

(8.) This includes all properties. The 2014 agricultural census does not provide data on the size, class, or location (rural or other) of properties with forest cover. A description of the distribution of the farm sizes in Costa Rica is located in the Appendix.

(9.) Another program, Protection Within Protected Areas, was also added in 2009. This program works to conserve forests on private inholdings in protected areas. However, the payment level was identical to that of the forest protection program.

(10.) Based on the 2007 IDS scores.

(11.) FONAFIFO has noted that the cut-off points of 50 hectares and an IDS score of 40 were based on “common knowledge.” A smallholder is defined as one that owns less than 50 hectares of land (Sanchez, 2015). The two lowest quintiles of the IDS distribution fall below 40.

Natasha James

Southern Research Station, USDA Forest Service; Department of Forestry and Environmental Resources, North Carolina State University

Erin Sills

Department of Forestry and Environmental Resources, North Carolina State University