Environ Dev Sustain (2016) 18:617–644
DOI 10.1007/s10668-015-9673-7

REVIEW

Payments for ecosystem services (PES): a flexible, participatory, and integrated approach for improved conservation and equity outcomes

Giulia Irene Wegner[1]

Received: 26 June 2014 / Accepted: 20 May 2015 / Published online: 3 June 2015

Abstract Over the past 20 years, payments for ecosystem services (PES) has become increasingly popular as a mechanism to promote environmentally sustainable land-use practices, and a burgeoning literature has been produced on this policy approach. The goal of this paper is to offer a comprehensive review of this literature, and to focus on four major aspects of PES: (1) its efficiency in delivering environmental conservation, (2) its impacts on the well-being of local land users, (3) its interaction with local norms of distributive justice and environmental stewardship, and (4) its interplay with broader national policies and socio-economic trends. Two major insights are drawn from this review of the literature. First, the conceptualisation of PES according to the neoclassical economic theory of efficient market transactions and utilitarian human behaviour may be unrealistic and counterproductive. In terms of efficient financial transactions, the physical properties of public ecosystem services obstruct the voluntary establishment of PES schemes by direct beneficiaries, practical constraints exist on the enforcement of outcome-based conditionality, and efficiency goals may need to be partly sacrificed to prevent the exacerbation of social inequalities. In terms of human behaviour, land users' actions are shaped not only by personal utility calculations, but also by intrinsic norms of distributive justice and environmental stewardship; the interaction of PES with these intrinsic norms can negatively impact on its local legitimacy and even 'crowd out' existing motivations for the conservation of nature. The second insight is that land users' capacity to shift to sustainable land practices, while influenced by the direct payments, remains strongly determined by broader socio-economic trends and by national strategies for rural development and institutional reform. On the basis of these insights, a flexible, participatory, and integrated conceptualisation of PES that can better account for this range of physical, socio-economic, and normative factors is proposed here as more capable of delivering efficient, equitable, and resilient conservation outcomes.

✉ Giulia Irene Wegner
giulia_wegner@yahoo.com

[1] Wildlife Conservation Research Unit (WildCRU), Department of Zoology, University of Oxford, Tubney House, Abingdon Road, Tubney, Abingdon OX13 5QL, UK

Keywords Payments for ecosystem services · Government and governance · Economic efficiency · Distributive equity · Motivation crowding-in/crowding-out effects · Deliberative participation · Rural development · Regulatory pluralism

1 Introduction

Since environmental degradation problems became apparent in the nineteenth century, the state and civil society have engaged in various ways in the search for solutions. One proposed solution that has become increasingly popular in the past 20 years is so-called payments for ecosystem services (PES), which consists of direct financial transfers to land users for the adoption of land-use practices that are environmentally sustainable in the interest of specific beneficiary groups or the general public. PES is one of a set of market-based policy approaches that have been developed since the 1980s to encourage producers to adopt environmentally sustainable production practices through supportive and restrictive economic incentives. What distinguishes PES from other types of supportive economic incentives for environmental sustainability, such as state subsidies and certification, is that instead of subsidising sustainably produced products within existing commodity markets, PES creates new separate markets for the direct purchase of so-called ecosystem services. Ecosystem services consist of life-supporting services (e.g. the recycle of nutrients; the assimilation of waste; and the regulation of climate, watershed, and pests/diseases) and provisioning services (e.g. water flow, domestic crops and livestock, and wild plants and animals). The combination of these services with human-related assets (e.g. man-built infrastructures, knowledge, and networks) leads to the realisation of those tangible and intangible benefits that are directly experienced and valued by humans, such as food, hydroelectric power, a stable climate, the intrinsic values of nature, and psychophysical equilibrium (MA 2003; Wegner and Pascual 2011).

Following its increasing popularity, a burgeoning literature has been produced on PES. The goal of this paper is to offer a comprehensive review of this literature, and to focus on four major aspects of PES: (1) its efficiency in delivering ecosystem services conservation, (2) its impacts on the well-being of local land users, (3) its interaction with local norms of distributive justice and environmental stewardship, and (4) its interplay with broader national policies and socio-economic trends.

Through the analysis of these four themes, the paper identifies two major obstacles to the design and implementation of effective PES schemes. The first obstacle is the neoclassical economics conceptualisation of PES as a policy tool in which negotiations should be geared towards the maximisation of economic efficiency, and in which individuals are expected to respond to financial incentives uniquely on the basis of personal utility calculations. This efficiency and utilitarian framing of PES remains mainstream within several environmental conservation organisations (e.g. Ingram et al. 2014), influencing how scientists, policy-makers, and field practitioners understand and operationalise PES (Pascual et al. 2014). This paper highlights how this conceptualisation of PES fails to account for the physical properties of ecosystem services, and a range of socio-economic and normative factors that strongly influence the capacity of land users and ecosystem service beneficiaries to participate in PES and derive benefits from it.

The second obstacle to the design and implementation of effective PES schemes is the fact that these schemes tend to be formulated as disjoint from broader national strategies for rural development and from wider socio-economic trends that strongly determine the capacity of land users to participate in PES schemes and shift to sustainable land-use practices.

I begin Sect. 2 of this paper by considering what distinguishes PES from other forms of supportive incentives for environmental conservation, and why it has come to occupy a prominent role in the environmental policy arena of less-developed countries. In Sect. 3, I consider the well-known debate on the extent to which trade-offs between environmental efficiency and distributive equity outcomes should be allowed, and argue that divergences within this debate rest on the endorsement of contrasting principles of distributive equity.

Beyond the debate on efficiency/equity trade-offs, fundamental questions remain to be answered about how to tackle major existing constraints on both these goals. Section 4 discusses constraints on environmental efficiency, including physical constraints on the administration of PES by the direct beneficiaries, and practical constraints on the enforcement of strict conditionality, and suggests that alternative efficiency-enhancing criteria such as geographical additionality and land users' participation in PES design may be more effective. Section 5 then analyses constraints on distributive equity, which consist of eligibility and ability filters to the participation of poorer land users in PES schemes, and points to the need of coordinating PES initiatives with broader national strategies of poverty alleviation.

In Sect. 6, I consider how further constraints on environmental efficiency and equity goals may be encountered when these goals are framed in ways that clash with norms of distributive justice held by local land users. Then, in Sect. 7, I draw attention to the potential interaction of PES with local values of environmental stewardship, and the possibility of designing PES so as to entice the 'crowding-in' and prevent the 'crowding-out' of these values. The importance of land users' participation in designing PES schemes that harmonise with these local norms of justice and environmental stewardship is discussed in Sect. 8.

In Sect. 9, the focus of analysis shifts onto the interplay of PES with broader socio-economic trends and national policies that affect rural livelihoods and can either encourage or constrain land users' successful participation in the programmes.

On the basis of this set of analyses, in Sect. 10, I conclude by encouraging the endorsement of a flexible, participatory, and integrated conceptualisation of PES that can better account for the complex set of physical, socio-economic, and normative factors that affect the implementation of this policy tool, and highlight the importance of a dialectic complementarity between the state and local institutions for the accomplishment of PES goals.

2 From government to governance to PES

The growing interest in PES observed in recent years is part of what has been described as a gradual move from government to governance, that is, from rigid forms of state regulation and property rights towards more flexible policy approaches that grant actors in civil society increased flexibility in setting and pursuing environmental goals (Lemos and Agrawal 2006; Gunningham 2009). This shift started in the 1980s, and it was motivated partly by the observed inefficacy of state regulations and property rights, and partly by the emergence and consolidation of neoliberal and people-centred theories of development that, although for different ideological reasons, both encouraged a shift of decisional power away from the state and into civil society (Akram-Lodhi et al. 2009; Roe and Nelson 2009).

In industrialised market economies, a range of state-managed market-based economic incentives came to be proposed, including direct incentives, such as state taxation and subsidies, and indirect incentives, such as state-managed frameworks for cap-and-trade, ecolabelling, and certification (Gatzweiler 2006; Gunningham 2009).[1] In the developing world, the early phase of this swing from government to governance was largely characterised by reforms for community-based natural resources management (CBNRM), the devolution of property rights over natural resources from the state to local communities being advocated as crucial for the conservation of local ecosystem services that rural dwellers heavily rely upon (Ostrom 2000; Roe et al. 2009). However, it soon became evident that CBNRM alone is inadequate to deal with regional and global environmental externalities that affect a large number of people at high scales of geographical aggregation, such as the erosion of common-pool goods with low levels of excludability (e.g. dryland pastures) and of public ecosystem services that by definition are non-excludable (e.g. climate regulation). Similarly, CBNRM alone was observed to be inadequate to address the erosion of ecosystem services valued by distant groups, but whose conservation clashes with the economic interests and values of local land users, such as charismatic wild species that pose a threat to rural livelihoods (e.g. large predators).

When dealing with these regional and global externalities, internal coordination through CBNRM is not possible, as it is difficult for those causing the externality and those affected by it to be bound by a shared set of property rights and rules. In these cases, the provision of forms of incentives by external actors becomes therefore necessary. Accordingly, CBNRM reforms in developing countries came to be accompanied by two financial mechanisms for tackling regional and global externalities. One of these mechanisms is known as integrated conservation and development projects (ICDP), which consists of the enforcement of environmental regulations and the establishment of livelihood enterprises that are either centred on the sustainable use of threatened resources (e.g. certified forestry and ecotourism), or meant to act as alternatives to their unsustainable use (e.g. tree planting and livestock rearing). The other financial mechanism consists of direct transfers of money to land users to incentivise the adoption of environmentally sustainable land-use practices, that is, PES.

In general, in this gradual move from government to governance, manufacturing producers have tended to be targeted through restrictive forms of incentives such as state taxation, cap-and-trade systems, and boycotts. On the contrary, agricultural producers have generally been targeted through supportive forms of economic incentives such as state subsidies, certification, and PES (FAO 2007; Iftikhar et al. 2007). This is partly a consequence of how equity concerns inform the allocation of rights between those who generate environmental externalities and those who are impacted by them. Manufacturers

[1] In their broad definition, incentives are the positive and negative changes in outcomes that individuals perceive as likely to result from particular actions (Gatzweiler 2006). With direct instruments like subsidies, taxes, and state-administered PES the incentive is provided directly by the state to those who generate the externality. Under cap-and-trade permit systems, habitat mitigation schemes and state-mandated PES the state creates indirect incentives by establishing a maximum overall level of externality (the cap), and then giving actors the possibility of trading permits among each other or of offsetting their impacts on ecosystem services at a certain location by financing the conservation of ecosystem services elsewhere. With indirect instruments like certification, eco-labelling, and voluntary firm-administered PES the state creates legal frameworks whereby the generator of the externality is put in a position to receive economic incentives from consumers and ecosystem service beneficiaries for the adoption of sustainable production practices, through a price premium, a share increase in existing markets, or access to new environmental markets.

are generally believed to be relatively wealthy and therefore to have no right to cause negative environmental impacts upon society, and as such they are held responsible for bearing the costs of pollution abatement (the 'polluter pays' principle). On the contrary, farmers are generally believed to be worse off than other groups in society, and consequently they are entrusted with rights while consumers are expected to bear the costs of environmental conservation (the 'beneficiary pays' principle) (FAO 2007; Iftikhar et al. 2007).

Although state-administered PES may resemble other forms of supportive incentives for environmental sustainability, such as state subsidies, certification programmes, and ICDPs, in fact PES differs from these policy approaches (Van Hecken and Bastiaensen 2010). State subsidy and certification programmes, such as the European and US agri-environmental schemes, promote the conservation of ecosystem services by subsidising sustainably produced products within existing agricultural commodity markets. Similarly, ICDPs that promote wildlife-based commercial enterprises (e.g. bee-keeping, non-timber forest products, and ecotourism) operate within existing agricultural and recreational commodity markets.[2] On the contrary, PES programmes create new separate markets for the direct purchase of ecosystem services. This may explain why subsidies and certification have been predominant in capitalist market economies where agricultural production is largely commoditised and consumers are willing to pay for environmentally certified agricultural products, while PES tends to be adopted in transition economies where subsistence and petty commodity agriculture are prevalent and consumers' willingness to pay for environmental sustainability is still low.[3]

PES is believed to carry some important advantages in comparison with other forms of supportive incentives. When compared to state subsidies and ICDP, PES is believed to tackle environmental externalities more efficiently because it tries to generate a stream of financial incentives that is sufficient to offset the opportunity costs[4] of abandoning unsustainable land-use practices, and conditional on the actual maintenance of ecosystem services (Swallow et al. 2007; Noordwijk and Leimona 2010; Van Hecken and Bastiaensen 2010). The off-setting of opportunity costs and the conditionality of payments are considered the key advantages of PES: without them, revenues generated through sustainable land uses may be used as a complement, rather than a substitute, to revenues generated through destructive uses.

3 Efficiency/equity trade-offs

One of the major debates in the literature on PES regards the management of potential trade-offs between environmental efficiency and distributive equity goals. This debate has largely occurred between scholars from mainstream environmental economics on one side,

[2] However, some authors refer to projects that promote wildlife-based recreational enterprises as a form of PES projects (e.g. Ingram et al. 2014).

[3] This explains why the majority of PES initiatives cited in this paper, and in the literature in general, are from developing countries. Nonetheless, many of the insights and conclusions reached in this study apply to PES programmes in both developing and developed countries.

[4] The opportunity cost of a choice is the cost of forgoing the best alternative use of a resource (Arrow 1969). In the case of PES, opportunity costs can be incurred from offsetting natural resources from alternative potential uses, and from adopting sustainable but less profitable land-use practices.

and from institutional and ecological economics on the other (Farley and Costanza 2010), and it is rooted in the way PES is conceptualised in the field of economics.

In economics, PES is conceived as an attempt to put into practice the Coase theorem (Wunder 2005; Engel et al. 2008), and it is framed according to the neoclassical Paretian precept of economic efficiency. According to the Coase theorem, a negative environmental impact (e.g. the degradation of an ecosystem service) can be eliminated by allocating enforceable user rights between those who cause the impact (e.g. land users) and those who experience it (e.g. ecosystem service beneficiaries), because such allocation of rights stimulates private negotiations between the parties (e.g. payments for sustainable land-use practices) for the elimination of the impact (Coase 1960). The theorem states that under specific conditions, these negotiations between the parties can result in a Pareto efficiency outcome, whereby the total net benefits accrued to land users and ecosystem service beneficiaries from these transactions are maximised, and whereby some of these actors are made better off and none of them are made worse off (Coase 1960).

The conditions necessary for Coasean negotiations between parties to result in a Pareto efficiency outcome are that they should be as close as possible to market transactions of demand and supply. In the case of PES, this implies that on the demand side, payments should be made voluntarily by direct ecosystem service beneficiaries such as private firms and households (Wunder et al. 2008; Fisher 2012). On the supply side, it implies that payments should target the most effective ecosystem service providers, which are those who can deliver the highest level of environmental additionality[5] and incur the lowest level of opportunity and transaction costs,[6] so as to maximise the amount of ecosystem services that can be bought with a given budget (Wunder et al. 2008; Kroeger 2013). The debate on PES efficiency/equity trade-offs revolves around this efficiency requirement that payments should be strictly targeted, and its implications in terms of distributive equity.

The concept of distributive equity relates to the fair or just distribution of socio-economic factors, including the benefits and costs of policy interventions, in society (Dobson 1998).[7] Theories of distributive justice fall into two broad categories: consequentialist and deontological (McDermott et al. 2013). Consequentialist theories of equity focus on the maximisation of benefits for the greatest number of individuals in society and are embedded in the Paretian concept of efficiency endorsed by modern welfare economics (Wegner and Pascual 2011). On the contrary, deontological theories of equity focus on the

5 Additionality refers to the amount of ecosystem services generated under a PES scheme that is additional to what would be generated if the scheme was not implemented. In most cases, the level of additionality can only be postulated, because its scientific measurement is strongly limited by the difficulty of obtaining context-specific information on the causal relationship between land-use practices and ecosystem service provision (Corbera et al. 2007).

6 Transaction costs are the costs of carrying out market negotiations (Arrow 1969). In the case of PES, transaction costs are related to the definition of the ecosystem service to be maintained, the identification of potential sellers and buyers, the development of mutual trust between them, the bargaining over the service's price, the transfer of payments, the monitoring of contractual obligations and conservation outcomes, and the enforcement of contracts (Vatn 2010).

7 The terms justice, fairness, and equity all refer to the concept of fair treatment, although the concept of justice is generally defined in terms of conformity (of an action or thing) to a moral right, while the concept of equity is comparative, being concerned with the relative circumstances of particular groups in society (McDermott et al. 2013). The concept of equity is composed of two core dimensions: distributive and procedural (Corbera et al. 2007; McDermott et al. 2013). In this review, I focus only on the economic aspect of distributive equity, this being, until now, the most used in analyses of PES. A comprehensive analysis of distributive equity would focus also on the distribution of the non-economic outcomes of PES, which relate to the psychological, social, political, and cultural dimensions of human well-being (see Sen 1999).

relative distribution of benefits among individuals in society according to specific normative principles or rules. A number of deontological principles of distributive equity exist, known as 'liberty rule', 'equality rule', 'opportunity cost rule', 'needs-based rule', and 'merit-based rule' (Konow 2001; Pascual et al. 2010; McDermott et al. 2013). The endorsement of specific consequentialist and deontological principles of equity can vary across individuals and groups, as well as across contexts: for example, the equality rule might be considered the most adequate approach in the context of votes distribution, the needs-based rule in the context of aid allocation, and the merit-based rule in the context of job appointments (Sen 2009).

Most debates on distributive equity in the context of PES are informed by the needs-based rule. This principle advocates the equal satisfaction of basic needs for all individuals and consequently focuses on the distribution of socio-economic factors among the most disadvantaged members of society (Konow 2001). According to this equity principle, then, PES programmes promote distributive equity when they are pro-poor, that is, when they preserve and/or promote the well-being of the poor (Iftikhar et al. 2007).

In the remainder of this section, I review the empirical literature on the impact of PES on the well-being of poor land users, and then use findings from this literature to analyse the debate on PES efficiency/equity trade-offs.

3.1 The equity impacts of PES: empirical findings

Studies from developing countries indicate that in absolute terms, i.e. in relation to an initial baseline, PES has often had beneficial socio-economic effects on poor land users who are able to participate in the programmes as ecosystem service suppliers, through both cash benefits and non-cash benefits such as enabling the transition to more profitable production practices, although per capita gains are seldom large (Wunder et al. 2008; Bond and Mayers 2010; Noordwijk and Leimona 2010; Van Hecken and Bastiaensen 2010; Richards 2012).

There are also cases in which PES schemes have negatively affected poor participants. For example, reviews of China's Sloping Land Conservation Program (SLCP) (Bennett 2008) and of a range of tree-based PES schemes in Africa (Namirembe et al. 2014) indicate that in a number of cases the absence of pre-programme estimations of participants' opportunity costs has resulted in payments that are too low to compensate the costs they incurred from farmland retirement, resource-use restrictions, and increased human–wildlife conflict, and that in some instances payment agreements have not been fully honoured by the buyers of ecosystem services. Another study in southern Mexico suggests that the joint establishment of a community-conserved area and a PES scheme for watershed protection on ancestral land may have been the cause of a decrease in cultivated land, pest control, meat procurement, and environmental knowledge, and that these have had a negative impact on local food security, households now spending most of their annual PES income in purchasing food from external sources (Ibarra et al. 2011). However, despite these negative experiences, in many cases PES has had some degree of beneficial socio-economic impacts on poor participants.

On the contrary, studies from developing countries point out that in absolute terms PES may have negative impacts on poor non-participant land users. To start with, like all market opportunities, PES can encourage the legitimate or illegitimate consolidation of exclusive property rights over land and other natural resources in the hands of few wealthier and powerful individuals, at the expense of larger groups of poorer land users who may have only customary rights to the same resources (O'Neill 2001; FAO 2007;

Asquith et al. 2008). In addition, PES can have negative impacts on poor non-participants, especially the landless and unemployed, through negative derived effects on local wages and commodity prices. These impacts are more likely to occur in 'use-restricting' schemes that demand the cessation of economic activities than in 'asset-building' schemes that require a shift to sustainable production practices (Coomes et al. 2008; Wunder 2008; Adhikari and Boag 2013).[8] Comprehensive macroeconomic studies are still lacking, but it is generally believed that in areas with limited links to external commodity and labour markets, these negative derived impacts may be substantial (Asquith et al. 2002; FAO 2007; Wunder 2008).

Moving on to equity outcomes in relative terms, studies from developing countries indicate that PES has the potential to exacerbate existing inequalities within communities, i.e. it can widen the gap between wealthier and poorer land users, even when the condition of the latter may not necessarily worsen relative to an initial baseline (e.g. Corbera et al. 2007; Adhikari and Agrawal 2013). PES can increase local inequalities through participation filters, or selective barriers, that favour relatively wealthier land users and exclude poorer ones from participating in PES schemes as sellers. There are three types of such participation filters: eligibility, targeting, and ability filters.

The main type of eligibility filter is the ownership of land, which is essential to any PES scheme[9]: in many areas, statutory and customary property rights over land and natural resources are unclear or insecure (Mwangi and Markelova 2009), with the result that the poorest members of rural communities tend to be excluded from PES (Rosa et al. 2004; Corbera et al. 2007). Targeting filters originate from the fact that in order to contain transaction costs, PES schemes tend to target a small number of large (wealthier) landowners who can sell a greater volume of ecosystem services per transaction, rather than a large number of small (poorer) landholders (Engel et al. 2008; Pagiola et al. 2008; Namirembe et al. 2014).

Ability filters refer to structural and cultural constraints in the adoption of sustainable land-use practices by the poorer land users, such as the lack of information, financial and human capital at the household level, and weak institutional frameworks and administrative capacity at the community level (Barrett et al. 2001; Landell-Mills and Porras 2002; Milder et al. 2010). For example, poorer households have struggled to participate in a PES programme for water provision in Tanzania due to their lack of financial capital to invest in new land-use practices and spread the risk of failure, and the need to use their limited human capital off-farm to earn wage cash and cover immediate needs (Blomley 2013). Gender inequalities in the participation in PES have also been observed to result from the presence of ability filters. For example, to date women's participation in schemes for watershed services has been significantly lower than men's, due to their often limited land rights and access to credit, and to socio-economic and cultural factors that constrain their participation in community decision-making (Richards 2012).

In general, the incidence of participation filters to the participation of land users in PES programmes depends on the initial distribution of wealth (land, financial and human capital) and power within the communities in which PES is implemented—a more even

[8] Contrariwise, asset-building PES programmes that promote a shift to sustainable land-use practices may have the effect of increasing demand and wages for local labourers (Adhikari and Boag 2013).

[9] Clear and secure land tenure is necessary for the functioning of PES schemes: on the supplier side, secure land tenure is important considering that participation in such schemes often involves a significant initial investment in the modification of land-use practices (Adhikari and Agrawal 2013) and the ability to de facto prevent third parties from using natural resources without consent; on the buyer side, secure tenure is important because it decreases the risk of non-delivery of the ecosystem service (Richards 2012).

distribution of these factors is likely to result in a more equal sharing of PES benefits (FAO 2007). For example, inequalities in rural land distribution in China are significantly lower than in other countries, and consequently the participation of small landholders in the SLCP has not been an issue (Bennett 2008). Contrariwise, in Costa Rica's PES scheme for forest conservation, payments have been observed to disproportionately benefit farmers with higher levels of education, wealth, and farm size (Porras 2010). Therefore, where large inequalities in the distribution of resources exist, unless specific attention is granted to poor and disadvantaged land users, PES interventions carry the risk of deepening existing social and gender inequalities.

3.2 Contrasting interpretations of PES efficiency/equity trade-offs

Scholars in the environmental economics tradition maintain that PES schemes can target the most effective providers of ecosystem services without necessarily resulting in undesired equity outcomes, and that therefore it is neither necessary nor advisable to sacrifice efficiency goals in the name of equity concerns (e.g. Pagiola and Platais 2007; Engel et al. 2008; Wunder et al. 2008; Richards 2012; Kinzig et al. 2011). Scholars in the ecological economics tradition, on the contrary, argue that targeting providers of ecosystem services according to efficiency requirements can often result in undesired equity outcomes, and that to prevent these outcomes efficiency goals need to be partly sacrificed (e.g. Corbera et al. 2007; Proctor et al. 2009; Milder et al. 2010). I suggest that this divergence of views on PES efficiency/equity trade-offs rests on two different interpretations of the need-based or pro-poor principle of distributive equity.

Environmental economists are likely to interpret the pro-poor principle of equity in absolute terms, meaning that PES schemes are considered to be equitable if the well-being of the poor does not decline in relation to an initial baseline.[10] Since, as we have seen in the previous section, the empirical literature indicates that PES tends to have a positive impact on participant land users, the only equity-related action deemed necessary by environmental economists is to prevent negative impacts of PES on non-participant land users, by compensating them for their exclusion from land and other natural resources (Kerr 2002; Pereira 2010), and for negative derived effects on wages and commodity prices (Engel et al. 2008; Wunder et al. 2008). On the contrary, further attempts to use PES to improve the economic condition of the poorer are considered by environmental economists to be superfluous and to be avoided, since these attempts may unnecessarily reduce the environmental efficiency of the schemes. Equity and poverty reduction goals, they argue, should instead be pursued through separate policy instruments (e.g. agricultural, education, and health policies) (Pagiola and Platais 2007; Wunder et al. 2008; Kinzig et al. 2011; Richards 2012). For example, since 2004 the Mexican PSAH programme for watershed services has achieved progressively higher poverty reduction levels by targeting poorer land users, but at the same time it has diverted a substantial share of funding to areas at lower risk of forest and watershed degradation, thereby losing in terms of additionality (Wunder 2008).

On the contrary, ecological economists are likely to interpret the pro-poor principle of equity in relative terms, meaning that PES schemes are considered to be equitable if they ensure not only that the well-being of the poor does not decline from an initial baseline, but also that the improvement in the well-being of the poor is equal to or higher than the

[10] This interpretation of the pro-poor principle of equity in absolute terms is compatible with the Paretian efficiency precept, which requires that welfare increases for at least some without decreasing for others.

average improvement in the well-being of a whole society, i.e. that existing inequalities between wealthy and poor land users do not increase and are possibly reduced. This stance is based on the assumption that a person's well-being is intrinsically linked to social equality, since it depends not only on basic functionings such as nutrition, health, and education, but also on her capability to interact with others in society on an equal basis, i.e. on her capability to fully participate in the society in which she lives (Townsend 1979; Sen 2006). Since the empirical literature indicates that PES can exacerbate existing inequalities through a range of participation filters, ecological economists consider deliberate efforts to tackle these filters and prevent inequitable outcomes necessary.

Overall, some main conclusions can be drawn from the above analysis. Financial budgets available to governments, NGOs, firms, and households are often limited, and the design of efficient PES programmes that maximise conservation outcomes with a given budget is therefore important. At the same time, social equality is an important dimension of human well-being, and since PES has a tendency to exclude poorer land users, deliberate efforts should be made to facilitate their participation, and in the process efficiency outcomes may be partially sacrificed.

Finally, it should be noted that in some cases, efficiency and equity goals may not necessarily clash with each other: when poor households constitute a large portion of the population inhabiting a landscape, or when their actions have a substantial impact on local ecosystems, their failure to participate in a PES programme and shift to sustainable land-use practices may undermine the overall environmental effectiveness of the programme (Van Hecken and Bastiaensen 2010).

Unfortunately, at present major constraints exist in relation to the pursuit of both efficiency and equity goals in PES. Sections 4–6 analyse these constraints and explore possible mechanisms to overcome them.

4 Constraints on efficiency

Constraints on efficiency goals exist in the form of physical constraints on the administration of PES by direct beneficiaries, and of practical constraints on the enforcement of strict conditionality. This section reviews the sources of these constraints and considers potential alternative mechanisms for the enhancement of PES efficiency.

4.1 Constraints on administration by direct beneficiaries

According to economic theory, for PES to deliver an efficient outcome, payments should be administered voluntarily by the direct beneficiaries of ecosystem services, such as private firms and households. This is because, it is argued, the investment made by direct beneficiaries is usually considerable, and therefore, they have strong incentives to ensure that their investment delivers the highest possible returns in ecosystem service flow (Engel et al. 2008; Pagiola and Platais 2007; Ingram et al. 2014). On the contrary, it is argued, when PES is administered by governments or NGOs, the beneficiaries (i.e. tax-payers and NGO subscribers) generally make only small individual contributions to the schemes through taxation and membership fees, and most of the time they are not aware of or have no authority over these schemes (Muradian et al. 2010; Vatn 2010), so that they have little incentive to monitor the outcome of their contribution (Pagiola and Platais 2007).

However, relatively few PES initiatives are initiated by private firms or groups of households. This is because, by definition, public ecosystem services such as climate regulation, pest/disease control, and biodiversity maintenance are characterised by the physical property of non-excludability, meaning that private rights cannot be established over them, and that therefore potential beneficiaries cannot be prevented from benefitting from them.[11] Since they can free-ride on the conservation efforts of others and others can also free-ride on their own efforts, households and firms lack strong incentives to invest in negotiations with the suppliers of these services (Engel et al. 2008).

It is for this reason that voluntary firm-administered PES schemes for the conservation of public ecosystem services tend to originate only when a firm or household's direct benefits are so large that it pays her to support the provision of the service even though there will be free-riders (Stiglitz 2000). For example, the bottler company Vittel/Nestlé Waters in France, the hydroelectricity company HEDASA in Guatemala, and a local group of 125 private households in Nicaragua are the largest beneficiaries of water regulation services in their water catchments (Perrot-Maître 2006; Corbera et al. 2007)—even though other users may free-ride on their conservation efforts, these companies and households have a strong incentive to finance watershed farmers for the maintenance of these services. Similarly, a community-based ecotourism enterprise in Cambodia and tourism companies in Maasai areas of Africa pay land users for ceasing activities that clash with wildlife conservation (Schomers and Matzdorf 2013), these companies being the main direct beneficiaries of wildlife-based recreational services.

However, for most public ecosystem services, the involvement of firms and households as buyers of ecosystem services needs to be either administered or mandated by governments and NGOs. In the case of PES programmes that are administered directly by state agencies and NGOs, these coordinate the payments on behalf of their constituencies. Examples include Costa Rican and Mexican programmes for public ecosystem services such as water regulation, soil regeneration, and biodiversity maintenance (Wunder et al. 2008; Schomers and Matzdorf 2013). In the case of mandated PES, private firms act as buyers of ecosystem services, but rather than engaging in these transactions voluntarily and as direct beneficiaries, they do so in order to offset their own impacts on public ecosystem services elsewhere, either under legal obligations by the state or under pressure by civil society groups. Examples of state-mandated PES schemes include those established under cap-and-trade programmes for the reduction of atmospheric greenhouse gases (Rubio Alvarado and Wertz-Kanounnikoff 2009) and various habitat mitigation schemes (Milder et al. 2010).

Another reason why private firms and households are unlikely to engage voluntarily in PES is the fact that public ecosystem services generate from extensive landscapes that are impacted upon by a large number of land users, so that negotiations tend to involve transaction costs high enough to outweigh the benefits of their conservation (Vatn 2005). This explains why in the majority of cases, when dealing with public ecosystem services,

[11] In this regard, it is useful to distinguish between waste absorption services, which are rival and excludable and over which property rights may be established, and regulation services, which are non-rival and non-excludable and over which property rights cannot be established. For example, carbon sequestration, as a type of waste absorption service, is rival and excludable: if rights over the carbon sequestration service of a forest are bought by a firm in order to offset its carbon emissions, right over the same forest cannot be bought by another firm as well. On the other hand, climate regulation, as a type of regulation service generated by carbon sequestration, is non-rival and non-excludable: enjoyment of a stable climate by one person does not diminish the enjoyment of the same service by another person, and nobody can be physically excluded from benefitting from such service (Farley and Costanza 2010).

state agencies and NGOs, with their higher capacity to coordinate financial transfers between multiple parties, constitute the only feasible option for PES (Wunder et al. 2008; Vatn 2010).

4.2 Constraints on strict conditionality

One of the postulated advantages of PES over other incentive-based policies such as ICDP and state subsidies is that payments are made conditional on the actual abandonment of destructive land-use practices (practice-based conditionality) and the verified maintenance or improvement of ecosystem services (outcome-based conditionality). The enforcement of outcome-based 'strict' conditionality is considered to be particularly effective at promoting environmental efficiency (Engel et al. 2008), for example by triggering land users' formulation of innovative and locally tailored land-use practices that maximise ecosystem services flow at lower costs (Zabel and Roe 2009).

In reality, the conditionality of many existing PES schemes is practice-based rather than outcome-based. Often, this lack of 'strict' conditionality is explained in terms of insufficient financial capacity and/or willingness of beneficiaries and their intermediary institutions to enforce it (Pagiola and Platais 2007). However, a practical constraint on true conditionality is also posed by the long period of time required before land-use changes deliver noticeable and verifiable improvements in ecosystem service flow. Due to this time lag, the adoption of an outcome-based approach would penalise early participants in the schemes and discourage them from getting involved, as the impact of their actions would be negligible until later periods (Luttrell et al. 2012; Blomley 2013). This is especially the case if a minimum number of land users are needed to join a PES scheme for changes in ecosystem services flow to be realised. An outcome-based approach also neglects the fact that some groups of land users might put as much effort as others in conservation practices but achieve lesser outcomes, due to locally different ecological and socio-economic conditions (Luttrell et al. 2012).

Moreover, while potentially promoting effective ecosystem services conservation, an outcome-based approach may involve other costs that increase the payment size and therefore reduce efficiency. To start with, high transaction costs are involved in the development and use of reliable indicators of ecosystem services flow and third-party verification (Schomers and Matzdorf 2013). This is especially the case when ecosystem services are affected by confounding factors that are difficult to analytically control, such as the stochastic variability of key ecological processes (e.g. fluctuations in rainfall and disease outbreaks) (Binot et al. 2009), the impact of global environmental trends (e.g. climate change), and the land-use practices of actors other than the ones targeted by the PES scheme (e.g. small-scale gold mining and boats cutting bank edges along water courses) (Blomley 2013; Leimona et al. 2015). In addition, an outcome-based approach shifts the risk of service provision to the land users (Leimona et al. 2015), who might therefore charge a risk premium (Schomers and Matzdorf 2013).

Finally, the unqualified enforcement of conditionality may inhibit the formulation of PES mechanisms that have other important qualities, such as being financially viable in the long term, and resistant to market fluctuations and political instabilities. For example, a mechanism to guarantee the financial sustainability of a PES programme is the development of a trust fund, whereby only the interests that flow from the principal in the fund are used to finance the payments. Trust funds violate conditionality, because in case buyers are unsatisfied with the quality of the service delivered, they can stop supporting the fund, but they cannot withdraw their initial contributions to it (Goldman-Benner et al. 2012).

4.3 Alternative efficiency-enhancing mechanisms: payment differentiation, geographical additionality, and participation

An alternative mechanism to pursue economic efficiency in PES is the differentiation of payments according to the environmental threat levels posed by individual households, and the opportunity costs of their abandonment of unsustainable practices. The reason why PES programmes rarely adopt payments differentiation is the high transaction costs involved in collecting and processing household-level information on environmental threat levels and opportunity costs. Practical experiments are underway in Europe and in the USA for the development of innovative PES schemes that minimise the transaction costs involved in the design of differentiated payments. These include payments formulated on the basis of typical plot types for multiple administrative blocks (Claassen et al. 2008) and auctions that invite land users to disclose their opportunity costs (Ferraro 2008). However, little is known about the feasibility of transferring these contractual designs to less-developed countries characterised by weaker legal and administrative frameworks (Schomers and Matzdorf 2013; Yin et al. 2013).

A more straightforward mechanism to improve the environmental efficiency of PES programmes may be for governments and NGOs to focus on the additionality criterion when selecting the geographical areas of PES intervention. In fact, several governmental PES programmes have been criticised for focusing on geographical areas of lower conservation priority, where targeted ecosystem services are likely to be preserved without introducing the payments, while neglecting other severely threatened areas. For example, Mexico's PSAH does not give explicit priority to overexploited aquifers (Schomers and Matzdorf 2013), and REDD+ programmes such as the Bolsa Floresta in Brazil and the Noel Kempff in Bolivia often cover geographical areas that have already been under protection for many years, either as state reserves or through a logging ban, and where local communities do not pose large-scale deforestation threats (Pereira 2010; Lin et al. 2012).

Another mechanism to increase the efficiency of PES schemes may be the participation of land users in the design of payment types and mechanisms, since tailoring payments to local needs and priorities can increase their effectiveness in eliciting the desired behaviour. For example, in Brazil's Bolsa Forest programme, whether payments are perceived as adequate to promote behavioural change appears to depend on whether local households are given the opportunity to define the types of payments received (Gebara 2013).

Moreover, a participatory approach can increase the practical and financial feasibility of PES programmes by tailoring sustainable land-use and conservation practices to local knowledge and capacity. In this regard, Leimona et al. (2015) encourage the adoption of a 'multiple ecological knowledge approach' in which scientific assessments clarify the source of ecosystem services deterioration, while local ecological knowledge informs the design of landscape management practices tailored to local capacity. The importance of scientific assessments is illustrated at a RUPES site in West Sumatra, Indonesia, where scientific data revealed that the reduction in water quantity of a lake used for hydropower electricity was influenced by variations in the amount and frequency of rainfall, and that the reduction in fish in the lake was due to overharvesting by downstream fishermen, rather than both being the result of siltation caused by deforestation by upper stream farmers. Meanwhile, at another RUPES site in Lampung, Indonesia, the farmers' construction of a simple sedimentation retainer for the maintenance of watershed functions along a riparian zone illustrates how locally tailored low-cost solutions to ecosystem services deterioration may be as effective as reforestation practices commonly promoted by PES beneficiaries.

Finally, PES schemes are more likely to promote the maintenance of ecosystem services over time, and especially after payments may cease, so achieving long-term efficiency, when they are used to support sustainable production activities than when they are used to encourage the simple interruption of a certain activity. PES may therefore be used to support sustainable traditional land uses that are under threat, or to enable a shift from destructive to sustainable production practices, or a shift to alternative livelihood strategies. The Plan Vivo PES project for carbon sequestration in Malawi, for example, invested in the development of additional sustainable income sources, such as bee-keeping and the sale of non-timber forest products, in order to prevent participants from becoming reliant on the annual carbon payments, a problem that had occurred at other project sites (Dougill et al. 2012). Likewise, the success of a PES programme for wildlife conservation in the Maasai Steppe of Tanzania has been attributed to its support for traditional seasonal grazing practices that are compatible with wildlife migration patterns, and which prevent the conversion of the plains to crop-based agriculture, rather than an utter restriction upon their use (Ingram et al. 2014).

5 Constraints on equity

Several mechanisms have been suggested to facilitate the participation of poorer land users in PES schemes. First of all, before establishing PES schemes, or before encouraging their establishment by the private sector, governments and NGOs would need to invest in the clarification and enforcement of the tenure rights of the poor over land and other natural resources, so as to circumvent the eligibility filter (Milder et al. 2010). The Plan Vivo community-based PES project for carbon sequestration in sub-Sahara African countries (Dougill et al. 2012), the Rewarding Upland Poor for Environmental Services (RUPES) programme for watershed services in Indonesia (Adhikari and Boag 2013), and pilot REDD+ initiatives in Brazil (Duchelle et al. 2014) indicate that the participation of poorer land users in PES schemes can be significantly bolstered by deliberate efforts to link them with land tenure regularisation efforts. In addition, community-based PES programmes in which payments are given to a community as a whole, rather than to individual landowners, may be used in places where few inhabitants have formal land tenure (Sommerville et al. 2010a) and to make payments accessible also by the landless (Rosa et al. 2004; García-Amado et al. 2011).

Second, governments and NGOs can tackle targeting filters through collective participation schemes and monitoring techniques that can reduce the transaction costs of dealing with many smallholders (Milder et al. 2010; García-Amado et al. 2011; Adhikari and Agrawal 2013).

Third, in order to facilitate the participation of poorer land users in PES schemes, governments and NGOs need to invest in the elimination of ability filters at the household and community level. For example, they may assist disadvantaged households with access to microcredit programmes, extension services for the adoption of new land-use practices, and the formation of farmer groups for pooling labour. They may then assist disadvantaged communities with the strengthening of local governing institutions, and capacity building for the participation of women and other marginalised groups in decision-making (Milder et al. 2010; Richards 2012; Blomley 2013; Adhikari and Agrawal 2013).

Some experiments for the elimination of eligibility and ability filters have been carried out, for example, in so-called pro-poor PES programmes for silvopastoral management in Nicaragua (Pagiola et al. 2008), and for water provision in Mexico (Muñoz-Piña et al. 2008) and Tanzania (Blomley 2013). Unfortunately, the implementation of these pro-poor PES programmes has proven to be problematic. For example, the Equitable Payment for Watershed Services (EPWS) project in Tanzania adopted a number of measures to facilitate the participation of the poorest households in the scheme, such as negotiating their use of unoccupied village land and forming farmer groups for pooling labour, but despite these innovative measures their participation remained disproportionately low (Blomley 2013).

Pro-poor PES programmes face difficulties in tackling eligibility and ability filters to the participation of poor land users largely because the lack of land, financial and human capital at the household level, and the weakness of governing institutions at the community level constitute deep structural constraints that demand substantial resources and multi-layered efforts to be tackled, and these are rarely within the scope of PES programmes. This highlights the important fact that in developing countries pro-poor PES programmes may have few chances of being successful unless they are integrated in and coordinate with more comprehensive national strategies for rural poverty alleviation. For example, a study of Ecuador's SocioPáramo programme for water, carbon, and biodiversity services in the highland Andean grasslands indicates that a significant percentage of Andean land users are small landowners who have no access to alternative productive land at lower altitudes, nor to off-farm income opportunities, so that they are unable to set land in the highland grasslands aside for PES. The authors therefore suggest that in order to enable the participation of small landholders, the programme should be preceded by structural land reforms that give them access to more and better land where they can concentrate agricultural production, and by the promotion of alternative off-farm livelihood strategies (Bremer et al. 2014).

6 Clashes with local norms of distributive justice

The neoclassical conceptualisation of PES is built upon the rational actor paradigm, assuming that human beings behave on the basis of extrinsic (or instrumental) motivations geared at maximising their personal utility and respond to public policy instruments uniquely on the basis of these self-interested motivations (Van Hecken and Bastiaensen 2010). In reality, empirical research shows that human behaviour is also shaped by intrinsic (or deontological) norms of altruism, justice, and environmental stewardship (Frey and Stutzer 2006; Wegner and Pascual 2011), and public policy instruments may interplay with these intrinsic motivations.

When designing a PES scheme, it may be important to consider what norms of distributive justice are held by local land users, because these views can influence the level of local legitimacy granted to a PES scheme. For example, in a community-based PES scheme for biodiversity conservation in Madagascar, the perception of an unjust form of elite capture, whereby community leaders withheld a larger share of in-kind rewards, discouraged some community members from joining the local forest associations involved in the scheme (Sommerville et al. 2010b). In a PES project in the Central Highlands of Vietnam, through which the national state in conjunction with private hydropower plants and water supply and tourism companies paid local households for forest conservation, land users who had not participated in past land tenure allocations and were therefore excluded from the scheme, allegedly sabotaged the formally tenured coffee plantations of qualifying PES participants (To et al. 2012).

Some studies indicate that in some cases local land users may endorse a 'merit-based' or 'contribution' principle of equity, according to which PES should target land users in relation to their past and present contribution to ecosystem services maintenance, rather than their potential effectiveness or relative poverty (Pascual et al. 2010). PES programmes that are strongly structured along the additionality criterion and exclude recognition for positive land-use practices that would have been undertaken anyway may therefore be perceived as unfair by those who already engage in such practices (Proctor et al. 2009), and potentially discourage their efforts. For example, the PSA programme set-up by the Costa Rican government to encourage forest protection initially excluded landholders who already practiced agro-forestry, but this eligibility filter was eventually lifted due to pressure from small landholders and indigenous groups who perceived it as unfair (Rosa et al. 2004).

In-kind PES in the form of human capital (e.g. education and health services) and physical capital (e.g. road infrastructures) may be the most feasible form of payment when the budget available to ecosystem service beneficiaries is too small to cover the full opportunity costs of land users. Such forms of PES may be favoured by land users for their capacity to prevent elite capture and ensure that the entire community benefits from a PES scheme. For example, in the sites of the RUPES project in Indonesia, Philippines, and Nepal, in-kind rewards are often reported to be the most preferred type of payment among land users (Leimona et al. 2015). However, in other projects, such as a community-based PES scheme for biodiversity conservation in Madagascar, in-kind incentives failed to meet the expectations of land users for whom rewards should be proportional to opportunity costs, and were viewed by some as human rights that should be guaranteed by the state rather than contingent on local conservation practices (Sommerville et al. 2010b).

The members of a community tend to be characterised by multiple axes of differentiation (e.g. livelihood, wealth, political influence, and ethnicity), and these differences can give rise to divergent sets of values and interests, along which multiple lines of networking, alliance, and conflict are likely to form (Agrawal and Gibson 1999). Accordingly, we can postulate that several and often conflicting norms of distributive justice may coexist not only across communities, but also within the same community. In PES programmes, finding a mechanism that satisfies multiple norms of distributive justice can be particularly challenging. For example, in the ReDirect project for biodiversity conservation in villages buffering a national park in Rwanda, the majority of resource users appeared to favour the homogenous distribution of payments, but those with a higher degree of dependence on the resources of the park expected a higher level of payment to match their greater opportunity costs (Gross-Camp et al. 2012).

Identifying the divergent norms of distributive equity that may exist within a community and promoting dialogue among those who hold them may therefore be important for the successful design of a PES schemes. For example, it may be important to know whether PES schemes designed to target poor and 'destructive' land users may be perceived as unfair by wealthier and 'virtuous' ones who already undertake sustainable practices, and whether dialogue among the two parties may increase acceptance for pro-poor initiatives. Similarly, dialogue between governments, NGOs, and land users may be necessary to frame additionality in ways that do not clash with locally-held merit-based principles of equity.

7 The risk of a motivation crowding-out effect

In the previous section, I discussed how PES may interact with local norms of distributive justice. PES may also interact with existing intrinsic norms of environmental stewardship that may be held by local land users. In relation to this, some authors have postulated and explored the risk of a 'motivation crowding-out effect'. A motivation crowding-out effect is the mechanism whereby public financial regulations and incentives result in behaviour that is opposite to the promoted one because they discourage and undermine, rather than complement, existing intrinsic motivations that are supportive of it (Frey and Jegen 2001).[12] More specifically, according to this hypothesis, when certain behaviours are considered to be a moral obligation, introducing regulations and incentives to promote those same behaviours may actually discourage and undermine their occurrence.

The occurrence of a crowding-out effect following the introduction of financial regulations and incentives has been observed in a broad range of policy contexts, ranging from subsidies for blood donation (Titmuss 1970) and for hosting nuclear waste disposals (Frey and Oberholzer-Gee 1997), to fines for failing to make mandatory minimum contributions to public goods (Reeson and Tisdell 2008) and for late collection of children from school (Gneezy and Rustichini 2000). Some meta-analyses and literature reviews have confirmed the occurrence of motivation crowding-out effects across many domains (Frey and Jegen 2001; Bowles and Polonía-Reyes 2012).

A number of studies have been carried out also in the environmental sector, both in developing and developed countries, exploring the potential crowding-out effect of financial regulations and incentives for the conservation of common-pool resources (e.g. Cardenas et al. 2000; Vyrastekova and van Soest 2003; Lopez et al. 2009) and public ecosystem services (e.g. Gawel 2000; Reeson and Tisdell 2008). However, empirical research on the impacts of economic regulations and incentives on intrinsic motivations for ecosystem services conservation is still scarce, and results from existing studies are mostly statistically non-significant, so that overall no conclusive evidence exists about the specific conditions under which PES may result in a crowding-out effect (Rode et al. 2015). Moreover, overall the incidence of crowding-out effects in public environmental policy is often likely to be negligible, considering that people's existing levels of intrinsic motivation for environmental stewardship may often be low (Martin et al. 2014; Rode et al. 2015). Therefore, given the lack of empirical evidence, in this paper we make only tentative suggestions about the occurrence and consequences of a crowding-out effect in PES, and about what mechanisms may be available to prevent it.

On the basis of existing research, it has been hypothesised that if a PES programme is introduced to support existing, although underperforming, sustainable land-use practices, and it is not framed so as to acknowledge the intrinsic motivations that underlie those practices, it may result in a motivation crowding-out effect (Kosoy and Corbera 2010; Van Hecken and Bastiaensen 2010; Beymer-Farris and Bassett 2012; Fisher 2012; García-Amado et al. 2013). The outcome would be a reduction in the original level of provision of

[12] Intrinsic motivations are deontological reasons for doing an activity, such as the feeling of satisfaction that carrying it out may bring, or normative principles. Extrinsic motivations, on the other hand, are instrumental reasons for doing an activity, that is, the attaining of certain outcomes, whether tangible or non-tangible (Ryan and Deci 2000). Intrinsic motivations for environmental conservation can be of two types: pro-social (motivated by relationships with other people or the larger community) and pro-environmental (motivated by values attributed to, or relationships with, nature) (Rode et al. 2015).

an ecosystem service, making relatively high payments necessary to restore and get beyond that level.

The risk of a crowding-out effect in PES is particularly problematic in the light of potential fluctuations on the demand side of PES transactions; if a decline or cessation of funds were to occur in a vacuum of environmental ethics and collaborative institutions, land users may decrease or stop undertaking sustainable practices altogether (Farley and Costanza 2010; Van Hecken and Bastiaensen 2010). The likelihood of such scenario is suggested by an interview-based study among forest-adjacent communities in Uganda, which indicates that the introduction of a PES scheme for tree planting may result in conservation efforts below the original level once the PES scheme were to end (Fisher 2012).

A spillover effect would be similarly dangerous, whereby the motivation crowding-out effect of a PES intervention spreads to areas where no external incentives are planned, so that land users in those areas stop engaging in sustainable practices (Frey and Stutzer 2006). A warning sign of a spillover effect was observed in the Regional Integrated Silvopastoral Ecosystem Management Project (RISEMP) in Nicaragua, where farmers from a nature reserve neighbouring a PES-targeted area started to demand compensatory PES to continue conserving the forests on their properties (Van Hecken and Bastiaensen 2010).

Nonetheless, a crowding-out effect is not necessarily unavoidable. In fact, where intrinsic values of environmental stewardship exist, but are not able by themselves to deliver effective conservation outcomes, PES may be designed so as to not undermine and to possibly enhance them, 'crowding' them in rather than out. Towards this goal, it is important to understand the psychological mechanisms that underlie crowding-out/crowding-in effects.

Cognitive evaluation theory suggests that positive financial incentives such as PES may crowd out existing intrinsic motivations for pro-environmental behaviour by promoting a shift from a deontological rationality to an instrumental one, whereby agents now expect to receive financial incentives in exchange for environmentally sustainable practices (Vatn 2005; Reeson and Tisdell 2008). For example, in a Mexican Biosphere Reserve, it has been observed that the longer PES schemes have been running in a community, the more people manifest utilitaristic (extrinsic) monetary reasons for the conservation of nature, and the less they express deontological (intrinsic) motivations (García-Amado et al. 2013).

Cognitive evaluation theory indicates that the introduction of financial incentives may also be perceived as a signal that the state or society at large either distrust the agent or deny her efficacy and right to self-determination, which is perceived as unfair and therefore crowds out her internal motivation for compliance with proper behaviour (Frey and Jegen 2001; Falk and Kosfeld 2006; Reeson and Tisdell 2008; Gneezy et al. 2011).[13]

[13] Two further psychological mechanisms have been observed to underlie the motivation crowding-out/crowding-in effects of various forms of financial incentives for environmental conservation. Financial incentives may crowd out existing intrinsic motivations for pro-environmental behaviour by undermining the capacity of this behaviour to enhance a person's self-image or self-esteem, as well as her public reputation, since it will no longer be clear whether such behaviour is being performed for ethical reasons or in response to external interventions (Benabou and Tirole 2006; Lopez et al. 2009; Gneezy et al. 2011). Finally, financial incentives may crowd out intrinsic norms of reciprocity, since other people's environmentally sustainable behaviour may now be viewed as a response to financial incentives rather than as a manifestation of personal ethics (Frey and Stutzer 2006; Vollan 2008). These two psychological mechanisms are probably less likely to apply in the case of PES.

When designing PES, first of all baseline information should be gathered about intrinsic values of environmental stewardship that may exist in an area, and scoping studies should be carried out about the potential impacts of PES on these intrinsic values. Then, in order to prevent the psychological crowding-out mechanisms described above, PES initiatives may be designed so as to be perceived by local land users as a signal that their environmental values are shared by the broader society, and that their rights to self-determination and their conservation efforts are being recognised and rewarded.

For example, PES may be framed in terms of 'stewardship awards' for community conservation activities (Van Hecken and Bastiaensen 2010), or of 'co-investments in environmental stewardship' between land users and ecosystem service beneficiaries (Noordwijk and Leimona 2010), and land users may be invited to participate in the design and monitoring of the schemes (Vollan 2008; Martin et al. 2014), so as to acknowledge their past achievements and reinforce their sense of efficacy and shared responsibility. In addition, PES could be accompanied by long-term environmental education programmes, since these have the potential of encouraging people's sense of efficacy and responsibility for environmental conservation (Frey and Stutzer 2006; Bremer et al. 2014).

A useful starting point in this direction could be the analysis of those PES initiatives where hints of a crowding-in of previously existing pro-environmental norms have been observed. An example may be the RISEMP in Nicaragua, where the village where PES was most successful was the one with the highest level of pre-project local norms for environmental conservation (Van Hecken and Bastiaensen 2010). Another relevant case occurs in Chapas, Mexico, where the owners of a communal forest appear to perceive PES as society's recognition of and reward for their past conservation efforts, rather than as an incentive to change behaviour or compensation for incurred opportunity costs (García-Amado et al. 2011). A similar case is found in Ecuador's SocioPáramo programme, where most participating communities had already set aside highland grasslands prior to participation in the PES programme and see the latter as an opportunity to consolidate the persistence of pre-existing conservation efforts (Bremer et al. 2014).

Similarly, it would be useful to analyse cases in which PES may have promoted new intrinsic motivations for nature conservation where they were previously absent. For example, on land adjacent to Nyungwe National Park, Rwanda, culturally specific intrinsic values towards nature have been largely eroded by fear-based park protection and a historical process of national modernisation that tends to associate traditional land-use practices with backwardness (Martin et al. 2014). Here, an experimental PES project for forest conservation was carried out in which payments to land users were accompanied by their increased involvement in park-related decision-making and exposure to environmental education. This participation- and education-based PES design was observed to successfully reduce illegal activities within the park, and the authors tentatively suggest that this may have been through the crowding-in of new intrinsic motives for nature conservation (Martin et al. 2014).

Overall, comparable research across diverse sociocultural contexts is needed to identify broad patterns in the occurrence of motivation crowding-out effects in PES, the psychological mechanisms that underlie them, and the mechanisms through which they may be prevented (Bowles and Polonía-Reyes 2012; Rode et al. 2015). Methods and guidelines also need to be developed to enable practitioners to gather pre-project baseline data on intrinsic environmental values and social norms, and to monitor changes in motivations. Towards these goals, major challenges remain to be addressed in terms of research design and methods (Rode et al. 2015). Nonetheless, despite these challenges, research on the crowding effects of PES is likely to be valuable; shaping PES so as to crowd in rather than out intrinsic motivations for environmental stewardship carries the potential of resulting in more resilient conservation outcomes, since in case funds to sustain the payments were to end, communities would still possess some norms and institutions upon which to forge the sustainable management of their resources.

8 The importance of land users' participation in PES design

The above analyses indicate that a key node for the development and implementation of effective PES schemes may be the participation of local land users in PES design and monitoring. To start with, the participation of land users in PES design may be necessary to tailor payment methods to local needs and priorities, and to predict how payments may interact with local norms of distributive justice, so as to increase their effectiveness in eliciting the desired behaviour. For example, by engaging with each other, local land users and PES administrators can try to calibrate a balance between the contrasting goals of pursuing some level of additionality, facilitating the involvement of poorer land users, and rewarding the efforts of virtuous ones.

Second, as mentioned in the previous section, the participation of land users in PES design and monitoring may help to prevent the crowing-out, and possibly promote the crowding-in, of local norms of environmental stewardship, by promoting land users' sense of self-determination, efficacy, and shared responsibility (Vollan 2008; Martin et al. 2014).

Finally, we have seen that the participation of land users in PES design can increase the practical and financial feasibility of PES programmes by enabling the tailoring of sustainable land-use and conservation practices to local knowledge and capacity (Leimona et al. 2015).

Few studies have explored what kind of local governance bodies and institutions may be established to enable the successful participation of local land users in PES decision-making. McDermott et al. (2013) encourage affirmative efforts to ensure the inclusion of disadvantaged groups of local land users such as women, the landless, and ethnic minorities. However, by themselves such affirmative efforts are unlikely to promote truly participatory PES decision-making processes and to result in PES schemes that are broadly and successfully endorsed by local land users. This is because unequal socio-economic and power relations that tend to exist within communities of land users, and between them and the organisations responsible for administering PES, are likely to prevent equitable forms of engagement among these actors (Young 2000; Poteete and Ostrom 2004). Therefore, unless these unequal power relations are actively handled through deliberative forms of engagement, it may be difficult for participatory processes to bring to surface the full spectrum of local needs, constraints, and ethics related to landscape management (Habermas 1984; Dryzek 2000; Young 2000), and for multiple stakeholders to negotiate legitimate PES schemes. Such a risk is illustrated by the Scolel Té project in Chiapas and Oaxaca, Mexico, which is the first local farmer-driven PES scheme for forest carbon sequestration. Here, failure to address power relations within communities has resulted in politically and economically powerful landholders largely controlling decision-making for and access to the scheme, which in turn has resulted in the uneven distribution of costs and benefits and in conflicts among community members (Corbera et al. 2007).

Although investing in participatory deliberative approaches to PES can introduce transaction costs, we may presume that these costs will eventually be justified by the increased capacity of PES to deliver desired conservation and equity outcomes. New research in this direction is therefore encouraged.

9 The interplay with broader policies and socio-economic trends

In the previous sections, I have explored the potential interplay of PES with local norms of distributive justice and environmental stewardship. In order to be viable, PES programmes need to take into account also broader socio-economic and political trends that affect rural livelihoods and can either encourage or constrain the participation of buyers and sellers of ecosystem services in the programmes. For example, in Tanzania's EPWS project, the participation of upland farmers as suppliers of watershed services was influenced not only by the size of the payment, but also by their ability to access fertilisers and irrigation, and by their suspicions of the national government's intentions, given previous incidences of people's evictions from sensitive water catchment areas. On the demand side, downstream factors such as leaking pipes and illegal small-scale gold mining along the water course, which spoiled the efforts of upstream farmers to deliver noticeable downstream improvements in water quality, undermined the willingness of government and private agents to act as buyers in the programmes (Blomley 2013).

Similarly, in the RISEMP project in Nicaragua, factors other than the size of the payment are likely to have played a role in encouraging the adoption of improved silvopastoral practices. Among these were rising financial incentives from the national boom in the milk market, improved access to milk collection centres, and the momentum of collective learning stimulated by these trends and facilitated by the project's technical assistance component (Van Hecken and Bastiaensen 2010).

In China, the implementation of the SLCP has coincided with an unprecedented socio-economic transition of the Chinese rural economy towards off-farm employment that has the potential of limiting the risk of reconversion of the set-aside plots back to farming once the PES programme expires. At the same time, however, there have been recent reductions in land retirement compensations and increases in subsidies for grain, vegetable, and livestock production, which carry the risk of altering opportunity costs and diminishing land users' incentives to keep cropland in retirement (Yin et al. 2013).

Overall, these case studies indicate that farmers' capacity to adopt sustainable land-use practices, while influenced by direct financial transfers, remains strongly determined by broader national policies and socio-economic trends concerning land tenure, rural infrastructures, agricultural and labour markets, and the enforcement of the law, and that these should receive due consideration when designing and implementing PES programmes, so that cohesive and enduring incentive systems may be devised.

10 A flexible, participatory, and integrated approach to PES

Two major insights can be drawn from this review of the literature on PES.

The first insight is that the neoclassical economics conceptualisation of PES, according to which negotiations should be geared towards the maximisation of economic efficiency, and individuals respond to financial incentives uniquely on the basis of personal utility calculations, may be unrealistic and counterproductive, since it is unable to account for the physical, socio-economic, and normative dimensions of real-world PES. First, land users' decisions and actions are shaped also by intrinsic norms of distributive justice and environmental stewardship, and unless these are taken into account, PES may interact with them in unexpected and potentially negative ways. Second, while budgetary constraints call for the pursuit of economic efficiency, designing PES schemes that do not exacerbate and possibly diminish existing inequalities and conflicts among local land users is equally important.

The second insight that transpires from this review of the PES literature is that land users' capacity to participate in PES programmes and shift to sustainable land practices, while influenced by direct financial transfers, remains strongly determined by broader socio-economic trends and by national strategies for rural development and institutional reform. Unless due consideration is given to the broader socio-economic and political context, PES may encounter significant structural obstacles to the participation of ecosystem services providers and beneficiaries in the programmes.

Overall, these insights indicate that analysts, policy-makers, and practitioners would be better served by a more flexible, participatory, and integrated conceptualisation of PES than the one advanced in neoclassical environmental economics, so as to better account for the complex set of physical, socio-economic, and ethical factors that influence PES, and enable the design of more effective and resilient programmes.

Flexibility refers to the relaxation of the efficiency requirements. On the demand side, a more flexible conceptualisation of PES accounts for the physical excludability of public ecosystem services and envisages schemes administered by private firms, households, and communities when these are the largest direct beneficiaries of targeted ecosystem services, as well as schemes administered by the state and NGOs when the degradation of ecosystem services affects a large number of citizens and is caused by the aggregate actions of a large number of land users. A flexible approach to PES also endorses a practice-based rather than an outcome-based criterion of conditionality, so as to not penalise early participants in the schemes and land users who live in ecologically and socially disadvantaged contexts, contain verification costs, and enable the formulation of innovative long-term funding mechanisms. On the supply side, a more flexible conceptualisation of PES endorses the assumption that a person's well-being is intrinsically linked to social equality, and therefore encourages the deliberate targeting of poorer land users and active investment in tackling eligibility and ability filters to their participation.

Coming to participation, a participatory conceptualisation of PES recognises that efforts to promote efficient conservation outcomes may clash with local needs and priorities and with local norms of distributive justice, and that the engagement of local land users in PES design is therefore fundamental to devising locally legitimate PES mechanisms and to increasing their effectiveness in eliciting the desired behaviour. The participation of land users in PES design may also help to prevent the crowding-out, and possibly promote the crowding-in, of local norms of environmental stewardship, so possibly enhancing the long-term resilience of conservation outcomes. Moreover, a participatory approach can increase the practical and financial feasibility of PES programmes by tailoring sustainable land-use and conservation practices to local knowledge and capacity. However, this study suggests that in order for local PES governance bodies and institutions to be truly participatory and deliver this range of positive outcomes, unequal power relations within communities of land users, and between them and the organisations responsible for administering PES, need to be actively handled through deliberative forms of engagement.

Finally, the integrated character of an environmental policy instrument refers to its formulation as an integrated component of broader national strategies for rural development and institutional reform. This is important for two separate reasons. First, the participation of the poorer sections of the rural population in PES schemes is strongly hindered by deep structural constraints, such as unclear and insecure land tenure, limited access to credit, insufficient on-farm labour, and undemocratic local governing institutions. Since these structural constraints are hardly within the capacity of PES programmes, the coordination of these programmes with broader national strategies of institutional reform and poverty alleviation is essential, if PES programmes are to be accessible to the poor. Obviously, where national strategies for poverty alleviation are absent or weak, the implementation of pro-poor PES and the prevention of unequal outcomes can be particularly problematic.

Second, the integration of PES in national strategies for rural development is important because it enables the design of incentives that are in synergy with broader socio-economic trends and national policies that concern land tenure, rural infrastructures, and agricultural and labour markets, and which strongly affect the capacity of land users to seize PES opportunities.

The flexible and integrated conceptualisation of PES suggested here fits well within a general approach to public governance known as 'polycentric governance' (Gatzweiler 2006; Andersson and Ostrom 2008) and 'regulatory pluralism' (Gunningham 2009). The argument behind this public governance approach is that the sustainable management of complex systems such as ecosystems is better achieved through the involvement of multiple actors at different organisational levels (households, firms, communities, NGOs, and governments), and the combination of multiple tenure regimes (individual, communal, and public) and multiple policy instruments (direct regulation, economic incentives, and voluntary commitments), rather than through concentration on one of these alone (Agrawal and Gibson 1999; Barrett et al. 2001). In fact, empirical evidence suggests that no single policy instrument, property regime, or social actor works efficiently, fairly, and sustainably in relation to all environmental goods, and that therefore there is no superior locus of conservation authority (Ostrom et al. 1999; Dietz et al. 2003). What is needed instead is the flexible and effective coordination of multiple policy instruments at different organisational levels, so as to best fit the type of environmental problem tackled and the local socio-economic context.

It is important to highlight that within this flexible and integrated approach, the state maintains a fundamental steering role to harness the capacities of local governments, communities, NGOs, and private actors to accomplish environmental policy goals with increased effectiveness and legitimacy (MA 2005). In fact, as markets expand and resources get commoditised and enclosed within exclusive properties, new conflicts over resources tend to appear, and a legitimate third-party authority such as the state becomes necessary to resolve these conflicts (Polanyi 1944; Vatn 2005). This means that incentive-based approaches such as CBNRM, ICDPs, certification, and PES still rely on the central government for their effective functioning. CBNRM and ICDPs require the state to formalise and enforce common property rights over local resources, so that communities can effectively cope with internal and external claims over these resources (Ostrom et al. 1999; Murphree and Taylor 2009). Certification tends to be most widely adopted where it works in conjunction with other government environmental regulations (Cashore et al. 2007). Similarly, firm- and NGO-administered PES programmes rely on the state to establish a supportive regulatory framework that guarantees the security of land tenure, recognises the validity of payments, facilitates the flow of information, and enforces contracts (Bromley 1997; Landell-Mills and Porras 2002; Adhikari and Agrawal 2013).

To conclude, the gradual shift from government to governance described at the beginning of this paper, with direct state regulation being replaced by community-based governance and market-based incentives such as PES, is not, as supposed by some, the proof of a global process of de-regulation. On the contrary, this shift is better understood as part of an overall trend of state re-regulation, whereby the expansion of both community- and market-based institutions over public environmental goods is rendered possible by the governmental and intergovernmental formalisation of collective and individual property rights and the regulation of emerging environmental markets (Robertson 2007; Swallow et al. 2007; Gunningham 2009). In modern societies, the narratives of sustainable development may change over time, but the dialectic complementarity between the state and local institutions remains one of the central nodes of potential success.

Acknowledgments I owe sincere thanks to Arild Vatn, Krister Andersson, Tom Blomley, and three anonymous reviewers for their sharp remarks and detailed suggestions on how to improve this paper. I am also grateful to Unai Pascual for his guidance and inspiration during the initial conception of this paper. Finally, I thank Paul Johnson, Sean Duffy, and Lia Ferrario for their support with editing and proofreading.

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