Monitoring payments for watershed services schemes in developing countries

Ina, Porras and Bruce, Alyward and Jeff, Dengel

IIED

March 2013

Sustainable Markets

Monitoring payments for watershed services schemes in developing countries

2013

Ina Porras, Bruce Aylward and Jeff Dengel

www.iied.org

This paper was produced by IIED's Sustainable Markets Group. The Sustainable Markets Group drives IIED's efforts to ensure that markets contribute to positive social, environmental and economic outcomes. The group brings together IIED's work on market governance, business models, market failure, consumption, investment and the economics of climate change.

Published by IIED, 2013

Citation: Porras, I., Alyward, B. and Dengel, J. 2013. Monitoring payments for watershed services schemes in developing countries, IIED, London

Front cover © Ina Porras

Sustainable Markets Group

International Institute for Environment and Development

80–86 Gray's Inn Road, London WC1X 8NH, UK

Tel: +44 (0)20 3463 7399

Fax: +44 (0)20 3514 9055

email: info@iied.org

www.iied.org

ISBN: 978-1-84369-903-3

This publication can be downloaded from http://pubs.iied.org/16525IIED.html

Monitoring payments for watershed services schemes in developing countries

Ina Porras, Bruce Aylward and Jeff Dengel

Payments for watershed services (PWS) are schemes that use funds from water users (including governments) as an incentive for landholders to improve their land management practices. They are increasingly seen as a viable policy alternative to watershed management issues, and a means of addressing chronic problems such as declining water flows, deteriorating water quality and flooding. In some places, local governments, donor agencies and NGOs are actively trying to upscale and replicate PWS schemes across the area. While their apparent success and progress in launching new initiatives is encouraging, there is still much to be learned from formative experiences in this field, especially with regard to monitoring and evaluation.

In this paper we discuss the monitoring and evaluation criteria behind compliance or transactional monitoring, which ensures that contracts are followed, and effectiveness conditionality, which looks at how schemes manage to achieve their environmental objectives regardless of the degree of compliance. Although the two are usually linked, a high degree of compliance does not necessarily ensure that a scheme is effective. This is because a poorly designed scheme may target the wrong land managers and land that is at least risk, meaning that payments do not generate the desired hydro-ecological or conservation benefits. As the levering capacity to demand payments for better watershed management increases, so does the need to understand the dynamics of such activities and demonstrate their impacts. While the growing interest in such schemes shows that participants believe in the principle of land management, evidence of their impact is needed to determine which initiatives genuinely add value and are worth pursuing.

Keywords: Payments for watershed services; environmental impacts; monitoring and evaluation; land use; low- and middle-income countries.

Authors

Dr Ina Porras (PI) is a Costa Rican environmental economist. Her current work focuses on designing market mechanisms for environmental services to improve livelihoods, valuation of ecosystem services and accounting for social impacts of environmental policies, particularly PES in Latin America and Africa.

Bruce Aylward is the Director of Ecosystem Economics LLC, a firm specialising in the application of economics to ecosystem management and restoration. He has 19 years of U.S. and overseas experience in research, policy analysis and project implementation across a range of ecosystems, topical areas and socio-economic conditions. He provides strategic advice and capacity-building on water rights acquisition for ecosystem purposes to foundation, non-profit and agency initiatives in the western U.S. He also carries out policy research, economic valuation and market design with clients and partners in Oregon and abroad on water resources management and water-related ecosystem services.

Jeff Dengel is a senior analyst at Ecosystem Economics LLC. His expertise lies in water and water rights, and transactions to reallocate water from agricultural to environmental use.

Acknowledgements

The authors would like to thank Julia Martín-Ortega from the Hutton Institute for her insightful and valuable comments.

Contents

Tables and boxes

1. Introduction

Sustainable land and water management can provide multiple environmental services (including watershed services) by conserving existing natural ecosystems, managing agricultural and agroforestry land, and restoring degraded ecosystems. These efforts are often driven by a range of instruments, from direct regulation through prohibitions and zoning, 'soft' approaches such as information and capacity building, market-based instruments such as taxes and fines, and hybrid approaches that use regulatory authorities and market mechanisms such as cap-and-trade measures.

The use of payment for ecosystems services (PES) to promote sound ecosystem and watershed management is an idea that has moved from economic theory to policy debate and finally into practice over the last 30 years or so. In watersheds, economic incentives are used to promote upstream land management practices that are expected to help protect or improve the quantity and quality of water downstream. Like carbon taxes or carbon cap and trade schemes that 'put a price' on carbon, these payment for watershed service (PWS) schemes aim to incorporate the externalities of land use into land managers' production processes in order to achieve greater economic efficiency.

PWS schemes are increasingly regarded as a viable policy alternative for resolving watershed management problems. In some places local governments, donor agencies and NGOs are actively trying to upscale and replicate PWS schemes across the area.

1.1 The problem, objectives and methodology

The problem. Actions associated with payment for watershed services must constitute a change in past or future behaviour and practices. They only generate real 'additional' benefits if they secure actions that would not have happened if the payment had not been made. The greatest difficulty in monitoring PWS schemes is knowing with any certainty what would have happened without them, since the PWS scheme is all that can be observed and measured (Ferraro and Pattanayak, 2006). While some regard this as a major issue in biodiversity conservation, it also applies to any kind of incentive that seeks to influence future behaviour. Such actions are inherently uncertain, but nonetheless necessary.

Box 1. Economic benefits of watershed protection

The keen interest in PWS schemes can be ascribed to their economic impacts and the way that they affect water filtration and purification, regulate seasonal flows, control erosion and sediment and preserve habitats. While scientists such as Bruijnzeel (2004b) and Calder (2005) highlight their capacity to regulate relatively intact soils, they also generate spiritual and other benefits unrelated to land and water use.

The proportion of natural forests is often seen as an indicator of a watershed's health (Kaimowitz, 2004). Indeed, economic analysis from the United States shows an inverse relationship between average treatment costs and forest cover, with the former declining as the latter increases (Postel and Thompson, 2005). However, exotic tree species with high water requirements reduce water flows: it is estimated that alien tree species in South Africa would account for more than 2,720 million m^3 of water or 16 per cent of total registered water use if left unchecked (Turpie *et al.*, 2008).

Other ecosystems like wetlands and páramos (alpine tundra ecosystems) naturally absorb, clean and release water flows to major cities in South America, meaning that they have little need for purification apart from chlorine (Buytaert *et al.*, 2006; Crespo *et al.*, 2009). Highly degraded slopes increase siltation and turbidity, while compaction and poor cover reduce the soil's ability to infiltrate water, increasing losses through evaporation and the risk of flash floods. Farm-level management plans have been successful in controlling non-point source pollution from agriculture in New York (Appleton, 2002); and a combination of mixed cropping, terracing and agro-forestry have been suggested as feasible techniques that will enable small-scale farmers to retain moisture for rain-fed crops in the Upper Tana in Kenya. It is estimated that this would cost between US$2 million and US$20 million per year, and generate possible benefits of between US$12million and US$95 million (IFAD, 2012).

The objective. The objective of this paper is to take stock of how we understand the monitoring and evaluation of PWS schemes. We argue that the environmental performance of PWS schemes is best assessed in terms of their conditionality and additionality, and by focusing on the issue of monitoring and evaluation we hope to stimulate activity in this area and ultimately help improve the design and implementation of future programmes.

Methodology. The apparent success and progress made in launching new PWS schemes should not distract us from the fact that we still have much to learn from experiences in this field. Various efforts have been made to review global progress with PWS, but there has been no systematic, widespread or rigorous review of what these schemes have achieved. This paper mainly focuses on low- and middle-income countries, but also draws on well-known experiences in the United States, Europe and Australia, looking at the evidence collected to date and published in journals and reports. The main sources of information include:

- IIED's PWS profiles (available online at www.watershedmarkets.org) These were prepared for a review of PWS which identified 50 ongoing and 45 proposed schemes (Porras *et al.*, 2008) up from 41 proposed and ongoing schemes in low- and middle-income countries identified in an earlier review (Landell-Mills and Porras, 2002). The profiles were updated in 2012 and new schemes and proposals identified.
- Forest Trends State of the Water Markets (Stanton *et al.*, 2010). Here, the emphasis was mostly on monetary payments and other economic instruments for managing water.

Table 1. Examples of land practices that include target watershed services

Promoted land use	Ecosystem service	Example
Conservation and protection of existing ecosystems	Bundled	• In Costa Rica most PES is allocated per hectare of land protected each year (Porras, 2010; Sánchez-Azofeifa *et al.*, 2007). In 2012, a set of criteria for allocating contracts for protection assigned extra points and increased payments by 25 per cent in water protection areas. • A compensation programme in Finland offers incentives to create new nature reserves that provide habitats for threatened species or protect areas of great natural beauty (Tikka, 2003). • Voluntary forest conservation contracts in Norway (Barton, 2010; Skjeggedal *et al.*, 2010). • Swedish nature conservation agreements (Naturvårdsavtal) are normally signed for 50 years (EEA, 2010; Mayer and Tikka, 2006). • The Austrian Natural Forest Reserve Programme (launched in 1995) compensates farmers for not harvesting over a period of 20 years.
Agricultural practices aimed at providing environmental services and on-site economic returns for farmers	Usually biodiversity and water	• Silvo-pastoral projects in Colombia, Nicaragua and Costa Rica (Casasola *et al.*, 2009; Ibrahim *et al.*, 2010; Montagnini and Finney, 2010). • Organic agriculture in Costa Rica: National Electricity Institute (ICE) Project in La Angostura Dam. • Agroforestry contracts in the PSA Programme in Costa Rica, Sumberjaya in Indonesia, and Jesus de Otoro in Honduras. • Best management contracts in the Catskill-Delaware Watershed in New York. • Most European countries use subsidies co-financed by the EU Common Agricultural Policy (CAP) to fund the conservation of agricultural ecosystems, such as the High Nature Value areas promoted in Europe (EEA, 2010).
Reforestation for commercial purposes (medium- to long-term schemes with timber as main objective)	Usually carbon but also for watershed protection	• Six national PES schemes and approximately 11 small local watershed schemes promote reforestation (Porras *et al.*, 2008). • Community reforestation contracts through Plan Vivo in Mexico, Uganda, Mozambique and other countries (www.planvivo.org). • REDD projects (Bond *et al.*, 2009).
Rehabilitation of degraded ecosystems for protection	Biodiversity and water	• Removal of alien tree species, Working for Water in South Africa. • PCJ in Brazil to restore riparian forests (Porras *et al.*, 2008).

Source: Porras *et al.* (2011). For many other examples see: Baylis *et al.* (2008); Ferraro (2009); Landell-Mills and Porras (2002); Porras *et al.* (2008); TEEB (2011); Waage and Steward (2007); and Wätzold *et al.*(2010).

- Websites of the main regional PES-hubs. Specifically RUPES in Asia and PRESA in Eastern Africa.

Schemes are not implemented in a linear manner, and many proposals are abandoned or absorbed into other projects or type of instrument. In Africa, for example, approximately half of proposals never make it to the pilot or implementation stages (Berttram, 2011).

1.2 A working definition of PWS

The idea of PES is underpinned by the notion that ecosystems become degraded because ecosystem services are not properly valued. PES schemes try to create an economic mechanism (usually to complement regulation) that will internalise the positive environmental externalities associated with the production of particular ecosystem services, including watershed protection, biodiversity and carbon sequestration (Kosoy *et al.*, 2007; Porras *et al.*, 2008; Wunder, 2005). The three main characteristics of PES schemes identified by Porras *et al.* (2008) include:

- suppliers or sellers of ecosystem services responding to an offer of compensation made by one or more beneficiaries (NGOs, private parties, local or central government entities, and/or a separate beneficiary that is not a central government entity);
- compensation being tied to land management practices specified by the programme;
- only the supply-side of the transaction is voluntary, in that providers enter in to the contract of their own free will.

Payments are made in cash or in kind. Ideally, they will fall somewhere between the opportunity costs of managing the land and the value (measured or perceived) that the user or bundle of users place on the ecosystem service (see section 3.2.2). Payments for watershed services generally encourage the adoption of land practices that are expected to influence the biophysical attributes of an ecosystem and affect the provision of freshwater supplies (see section 3.2.1).

PES and PWS schemes in low- and middle-income countries have paid more attention to different types of land practice that are expected to provide a particular environmental service (see Table 1 below) than to measuring and rewarding the changes observed at ecosystem level. Activities that support the provision of bundled environmental services are sometimes expected to complement each other. For example, forest conservation is expected to maintain existing water quality and quantity, protect biodiversity and safeguard the beauty of the landscape, and is therefore often 'marketed' as a 'bundle'. Conversely, there may be trade-offs between watershed services and other ecosystem services (like carbon), as carbon projects that promote large-scale reforestation may potentially reduce other ecosystem services like biodiversity and water flows, especially in water-strained environments and when using fast-growing, single exotic species (see Calder, 2005).

Interest in PWS-type projects rose sharply between 2002 and 2008. A review of emerging markets for environmental services conducted by IIED in 2002 found 41 proposed and ongoing schemes in low- and middle-income countries (Landell-Mills and Porras, 2002). By 2008, this had risen to at least 50 ongoing schemes and nearly as many proposals at different stages, despite major setbacks in some cases (Porras *et al.*, 2008). A 2010 review by Forest Trends looking at PWS and water quality trading indicates that these schemes have not been implemented in a linear manner, and that many proposals are abandoned or absorbed into other projects or type of instrument (Stanton et al, 2010). In Africa, for example, approximately half of proposals never make it to the pilot or implementation stages (Bond, 2008).

2. Conditionality and additionality

Conditionality refers to the conditions attached to a PES contract. Although the theory suggests that payments are only made if ecosystem services are provided (Wunder, 2005), in practice there seems to be a link between the payments or incentives and the actions contracted under PWS schemes.

Additionality refers to the net positive impact in the provision of ecosystem services created by the payment, compared with the baseline scenario or hypothetical situation where no such scheme is in place (Pascual *et al.*, 2009). In other words, it is the change in land use generated by the payment, which can be compared with what would have happened without the programme ('business as usual', see Wunder *et al.*, 2008).

For the purpose of this paper, conditionality is seen as the way that incentives change behaviour; and additionality as the manner and extent to which actions on the ground lead to net changes in watershed services. Both are affected by the nature of the initiative concerned, the thought given to its design, the account taken of reliable scientific information and the care with which it is implemented.

In terms of monitoring and evaluation, conditionality is monitored for compliance, and additionality for effectiveness.

It can generally be expected that PWS schemes will need to demonstrate a high level of compliance in order to be effective. The conditionality implicit in PWS contracts is typically a necessary step in generating additional benefits to those that would occur without the PWS scheme. However, a high degree of compliance does not necessarily equate to a high degree of effectiveness, as a poorly designed scheme may target the wrong land managers or land that is least at risk, meaning that the payments may not generate the desired hydro-ecological or conservation benefits. A case in point is the lack of additionality in the early part of the Costa Rican PES scheme (Sánchez-Azofeifa *et al.*, 2007).

2.1 Monitoring compliance

It is necessary to monitor compliance to ensure that the land manager and 'buyer' stick to the terms of the contract. With input-based systems, the focus is on the transaction. Assumptions are made about the impacts of predetermined land-use activities, and these changes are monitored accordingly. Output-based systems concentrate on changes in the ecosystem service, leaving the land manager to make the internal adjustments needed to provide the service.

Compliance monitoring usually is linked to:

- the suitability of the land uses selected to provide the required ecosystem service
- the type and level of payment as an indicator of whether it constitutes a realistic incentive
- strategies to monitor contract compliance (self-monitoring, group monitoring, remote monitoring)
- the type and credibility of the sanctions.

In simple terms, compliance monitoring seeks to ensure that the conditionality inherent in a PWS scheme is put into practice, and that the project is implemented effectively. Only then is the conditionality in place to drive additionality.

2.2 Monitoring environmental effectiveness

Effectiveness is monitored in order to ascertain the extent to which a PWS scheme has achieved its overall objectives. This kind of monitoring tracks how the reward offered by a scheme leads to improved watershed services and ecological outcomes. If the same ecological outcomes would have occurred without the PWS scheme, then it is not effective, and there is no additionality.

Effectiveness monitoring is linked to:

- the existence of an appropriate baseline
- attribution
- targeting strategies
- leakage and spillovers.

3. Evidence from application

This paper expands on the framework presented by Wunder *et al.* (2008), using the theory and evidence available from existing schemes to examine evaluation criteria and consider the outcomes and lessons to be learned from monitoring and evaluation efforts to date. Although some schemes have been going for several years, the 'market' is relatively new and loosely defined, and evidence of impact is still thin on the ground (Porras *et al.*, 2008; Wunder, 2005).

The PWS schemes reviewed in this paper are presented in the Annex.

3.1 The reported impacts of ongoing schemes

Most existing schemes in low- and middle-income countries are intended to change or improve land management decisions, regardless of the size of the payments made. It seems that well-designed and monitored contracts can encourage high levels of compliance, thereby impacting on land use and (for some) on water quantity and quality. The following sections discuss the factors that affect these results, and how they can be monitored. Some of the main reported impacts are listed below.

Impacts on land use

- The conditional land tenure scheme in Sumberjaya, Indonesia, now covers about 70 per cent of the protection forest (compared with only 7 per cent in 2004), building on the Indonesian law for community-based forest management. Conditional land tenure is now in place for 6,400 farmers that use practices such as multi-storey coffee gardens (Suyanto, 2010).
- By February 2010, the Costa Rica national programme covered nearly 730,000 hectares of forest and had planted nearly 3 million trees in agroforestry plantations.
- Between 2000 and 2007 the PSAH (National Programme for Hydrological Environmental Services) programme in Mexico had reduced the rate of deforestation from 1.6 per cent to 0.6 per cent (Muñoz-Piña *et al.*, 2011).
- The total plantation area in China amounts to 53 million hectares. This includes 28 million hectares planted in six years by the Sloping Lands Conversion Programme, and 8.8 million hectares of crops that have been converted to plantations through the Cropland to Forest Programme (Sun and Chen, 2006; Xu *et al.*, 2010).

Box 2. The perceived impacts of certain schemes

It seems to have become common practice to ask users about perceived changes in water services since the introduction of a PWS scheme. Kosoy *et al.* (2005) report that the majority of water users in Jesus de Otoro in Honduras and Heredia in Costa Rica see water provision as the most important benefit from forests, and believe that increased forest cover will lead to better water quality and greater water availability. They also note that 64 per cent of the 100 users interviewed in Jesus de Otoro and 39 per cent of the 100 interviewees in Heredia thought that water availability had improved in the past two years – although it is hard to see how any changes could be attributed to the PWS schemes, given the short time frame and the small geographical area covered in both cases. This difficulty in attributing benefits to PWS applies to other cases as well. It has been suggested that a perceived reduction in pollution from coffee-processing waste in Campamento, Honduras, had more to do with falling coffee prices and activity levels, and that pollution would increase again as coffee prices recovered (Ardón and Barrantes, 2003).

The case of Meijiang in China illustrates the difficulty of drawing conclusions about the impact of land management practices introduced by PWS, when farmer perceptions vary and other factors such as the extraction of river sand may help reduce sedimentation. Leshan *et al.* (2005) interviewed several farmers involved in the scheme. When asked about water quality, they reported an initial increase in soil erosion while the new activities were under way (building terraced strips, level ditches, bamboo ditches and planting), but did not see environmental pollution as a problem despite the common use of fertilisers and pesticides. There was no consensus on the impact of orchard development on water flow, as half of the interviewees said that it had increased and half that it had decreased.

Source: Porras *et al.* 2008

- Small-scale projects like San Pedro Norte in Nicaragua have now expanded to other areas under the Fondo Nacional de Desarrollo Forestal (FONADEFO). Other examples include small PWS schemes in the Murciélago y Corcuera watersheds in Rivas, El Sauce in Leon, and Cerro San Rafael in Chinandega (see http://www.fonadefo.org/proyectos.php for more information).
- In Ecuador, FONAG had more than 65,000 hectares under land management by 2008. However, the plantations that have been established are still too young to have their impact evaluated, and thresholds are regarded as very small, with about 2.5 per cent of the total area reforested (Cannon *et al.*, 2010).

Impacts on water quantity

- Statistics for local water availability upstream in Colombia indicate that long-term trends for annual rainfall and irregular fluctuations are due to large-scale geographical causes rather than the status of local forests (Kosoy *et al.*, 2005).
- Protecting cloud forests has little economic impact on water. Hydrological studies in Monteverde show that cloud forests have a modest impact on water budgets in the mountains, and very little effect on dry-season flows because of the relatively regular rainfall in the area. Most of the economic value of marginal flows is absorbed by the intra-annual reservoir downstream (Porras, 2008). A hydrological study in Los Negros, Bolivia, found no relationship between forest cover and streamflow (Le Tellier *et al.*, 2009) and studies near the area covered by Fidecoagua in Mexico confirm this trend, although they also do show that old secondary forests have roughly the same hydrological behaviour as established natural cloud forest (Muñoz-Villers *et al.*, 2010).
- Studies of páramo near Cuenca in Ecuador show that pine plantations decrease annual water yields, that livestock grazing does not affect soil compaction because of low herd density, and that cultivation mainly affects regulation by increasing the magnitude of peak flows and reducing base flows (Crespo *et al.*, 2009).
- Salas (2004) noted that a severe reduction in water volume was reported following reforestation activities with fast-growing exotic species (mainly mahogany and gmelina).
- Models used to calculate the impacts of the Working for Water programme on water flows in South Africa estimate that it generates 48 to 56 million cubic metres of additional water per annum (DWAF, 2006; Swallow *et al.*, 2009).

Impacts on water quality

- Community monitoring in Valle del Bravo, Mexico, indicates that water quality is good in forested areas receiving payments, but that it declines in lower areas of the watershed. This suggests that there may be potential problems with fish farms and domestic or industrial wastewaters (Manson, 2008). Fidecoagua implemented communal payments alongside the PSAH and reported a decline in sediments, although it is hard to judge the validity of this claim (Blanco and Rojo, 2005).
- Forest protection in El Triunfo has helped reduce the accumulation of sediment and lower heat levels. This has had positive effects on shrimp catches and the local fishing industry (Stem, 2005), although it is difficult to attribute these impacts to a reduction in the risk of deforestation without a baseline.
- San Pedro del Norte (Chinandega, Nicaragua) reported that water sources have become permanent since the introduction of the scheme (Obando, 2007), although the lack of an in-depth hydrological study and relatively small size of the scheme make it difficult to unequivocally attribute this development to it. Nevertheless, a study by Kosoy *et al.* (2005) suggests that this scheme (alongside Jesus de Otoro and Puerto Barrios) can demonstrate with a degree of confidence that the agreed environmental service is provided, and specify the threshold areas required to reach specific quality targets.
- A successful lesson from the Hunter River Salinity Trading in New South Wales, Australia, shows that salinity targets for a total area of 2.2 million hectares were met at all times in the upper sector of the river when discharges were allowed under the scheme (NSW, 2009).

3.2 Lessons from compliance monitoring

In this paper, we concentrate on monitoring compliance and evaluating the efficacy of programmes. The implication is that monitoring and evaluation involves providing assurances that the agreed land management or production practices have been put in place. Compliance and the resulting implementation of PWS are crucial for environmental effectiveness, and must be considered a necessary condition for successful schemes. Compliance and enforcement mechanisms are included in PWS schemes more often than hydrological monitoring.

3.2.1 Indicators and activities promoted under PWS

Kroeger and Casey (2007) highlight the potential loss of ecosystem services due to the difficulty of identifying and defining particular service units, and the resulting ability (or inability) to measure changes in services (this point is discussed more in depth in section 3.3.1). Thus, determining how a particular service is to be measured also helps identify the potential constraints to monitoring and evaluation. Clear biophysical interconnections between the land use activities prescribed by a PWS scheme and the resulting hydrological effects need to be identified, understood and communicated to all programme participants, particularly providers and beneficiaries (Landell-Mills and Porras, 2002).

Land practices often serve as a proxy for watershed services when PWS schemes are monitored and evaluated, due to the assumption that particular land management techniques will increase the probability of downstream recipients receiving the desired services (Asquith and Wunder, 2008; Ortiz *et al.*, 2003; Porras *et al.*, 2008). However, one cannot be certain that this will happen because of the external factors or confounders (see section 3.3.2) that influence hydrological conditions in watersheds (Asquith and Wunder, 2008).

The use of proxy actions to represent the production of desired watershed services is often based on local assumptions and perceived biophysical interconnections that tend to oversimplify complex ecosystem functions (Echavarria *et al.*, 2004; Gutrich *et al.*, 2005; Kaimowitz, 2004). Thus, it is widely assumed that there is a positive correlation between forest cover and the quality and quantity of water provided by a watershed (Echavarria *et al.*, 2004; Rojas and Aylward, 2003b; Saberwal, 1998), and that changes in the condition of vegetative cover indicate changes in hydrological functions (Asquith and Wunder, 2008). Some proponents of this view base their programme assessments on surveys of stakeholder perceptions rather than on-site hydrological studies (see Box 3 below), and rapid assessment methodologies such as the toolkit introduced by RUPES in Asia (Jeanes *et al.*, 2006) and the broader toolkit explored by UNEP (CCI and BirdLife International, 2011) are increasingly promoted as a means of bridging the gap between science and local perceptions.

Box 3. Stretching the complentarity of carbon and water?

In the Piura region of Peru a pilot group of the UK-based Cafédirect AdapCC project and the Peru-based Cepicafe (one of Cafédirect's suppliers) is involved in a combined initiative to engage with the carbon market, provide protection against flash floods and support sustainable local farming. Farmers are using pine trees and native quinal species to reforest degraded grasslands at higher elevations (70 per cent pine and 30 per cent quinal). Until now Cafédirect has pre-paid funds to purchase credits in order to help get the project up and running, but it is expected that 10 per cent of the carbon credit sales from saplings will go to Cepicafe, under the CarbonFix standard.

The pilot has planted 224 hectares so far, and ultimately aims to cover 5,000 hectares (Lee, 2012, in The Ecologist). Although the project has been widely applauded, there are some questions as to how it will be scaled up and what effect this will have on the wider ecosystem, especially if it continues to use pine as the main species. Piura is located in the northern coastal desert of Peru where extreme weather conditions combine very low average annual precipitation with marked rainfall during the wet season. Hydrologists have long warned that the high water requirements of fast-growing forest species in large plantations can significantly reduce water flows downstream (Calder and Aylward, 2006). This is a particular problem in areas like the desert of Peru, where water is very scarce.

Given the complexity of watershed ecosystem functions and the fact that common myths often prove to be unfounded, assuming that traditional actions such as reforestation will deliver certain services (expected increased in water flows) seems a questionable extrapolation (Aylward, 2005; Bruijnzeel, 2004a; Bruijnzeel *et al.*, 2010; Calder, 2005).

3.2.2 Monitoring the adequacy and timing of payments

Payment levels. Schemes need to cover the opportunity costs of their actions in order to avoid compromising long-term participation. Failure to do so may make alternative land uses more attractive, and pass costs on to farmers (possibly unjustly). The overall PWS payment ideally sits somewhere between the value of the service to users (or their representative, such as the government) and the cost of implementing the land use that is expected to provide the service.

The way that opportunity costs are calculated should be taken into account when assessing the adequacy of the compensation allocated to service providers (Hoffman, 2009). In order to provide effective or adequate compensation, payments to suppliers (landowners) should exceed or at the very least meet the opportunity costs of intended land uses outside those specified by a PWS scheme (Engel *et al.*, 2008; White and Minang, 2011).

Placing a value on the opportunity costs not only helps determine whether payments are adequate, but can also enhance the environmental efficiency of a scheme by differentiating between payments, especially in cases where there are budgetary constraints or excessive demand. For example, it was found that too much compensation was being paid for approximately 75 per cent of the land enrolled in the Fundacion Natura scheme in Bolivia, and not nearly enough for other areas (Hoffman, 2009).

Most opportunity cost analyses tend to be unduly simplistic. There are a number of reasons for this:

- Opportunity cost calculations are often based on fairly basic cash flow analysis that fails to account for risks and uncertainties or the different timescales to which rural land managers work.
- Changing behaviour involves giving up previous practices or 'losing' gains that used to be made through traditional land management. In behavioural economics, this is associated with an 'endowment effect' (also known as 'divestiture aversion'), which reflects people's tendency to focus on what they lose or give up in a transaction rather than what they gain from it.
- Simple cash flow analyses do not take account of the seller's possible inner motivation for developing and implementing a PWS transaction.

This suggests that a basic opportunity cost analysis (which tends to provide a static picture) will understate the incentives needed to motivate land managers to engage in a transaction and change their behaviour. Even when opportunity costs are judged accurately, the decision to participate may be influenced by non-monetary issues like seller remorse or group pressure, which impact on participation in the programme as word spreads through the community of potential sellers.

Therefore, keeping an eye on opportunity costs is a necessary but not sufficient condition for successful PWS.

Apart from monitoring the adequacy of payments, there is the much larger question of whether PWS schemes deliver economic value, and whether there is an associated increase in environmental services (see next section). There are methods for valuing hydrological services, as for any non-marketed good or service (Freeman, 1993), but they can be difficult and costly to apply. Until now, this type of valuation has usually been done in the context of academic studies that are not linked to PWS, or as preliminary efforts to justify PWS schemes in which the opportunity costs of previous production or replacement costs are used as the measure of value. This approach is largely meaningless in terms of environmental services (Rojas and Aylward, 2003a) as the impact on producers' net profits from farming reveals nothing about the value of watershed services that would be derived from the land under alternative scenarios. Behavioural economics based on multi-stakeholder, multi-criteria analysis and hydrological studies is increasingly used to provide a better assessment of the non-financial values and costs for ecosystem services (Spangenberg and Settele, 2010).

Type and timing of payments. As the name implies, PWS schemes involve some form of compensation for agreed activities. This can be made in cash or in kind, as one-off or continuous payments. According to economic theory, cash payments are superior to payments in kind because of the way that they affect the recipient's utility and expenditure behaviour (Currie and Gahvari, 2007). In-kind transfers are often seen as paternalistic, but may be useful when cash is seen as culturally inappropriate (Wunder and Vargas, 2005) or when large numbers of suppliers render per capita cash payments too small to be meaningful.

The type of payment will have implications for programme compliance. For example, in-kind, one-off compensation may provide immediate benefits but is very difficult to withdraw in the case of non-compliance. Smaller, continuous payments may encourage long-term compliance, but will be insufficient to encourage people to make expensive initial investments in their farm if they are too low.

3.2.3 Field monitoring strategies

It should be noted that this discussion focuses on field monitoring rather than overall project monitoring, which can involve multi-stakeholder boards and independent audits (such as the FONAG trust fund in Ecuador). Here, compliance monitoring is concerned with the enforcement of contracts with individuals, associations or local facilitators. Stakeholders and participants may be involved if there is adequate capacity on both sides. This helps reduce transaction costs (Reis *et al.*, 2007), but this type of 'monitoring' generally refers to data collection (readings from staff gauges or water quality tests) rather than the assessment of environmental additionality.

Delegating monitoring and enforcement to programme participants is common practice among institutions that are responsible for managing sustainable common pool resources (Ostrom, 1990). Intermediaries such as non-governmental organisations (NGOs) can also help reduce costs, as with the technical assistance to Costa Rica's national PSA provided by FUNDECOR and CODEFORSA in exchange for a proportion of the final payment received by farmers (Porras *et al.*, 2012). What this means in effect is that many transaction costs are shifted from the buyer (government) to the seller (farmer).

The most common types of compliance monitoring found in existing schemes include:

- self-monitoring
- participatory monitoring by a group or groups of providers, or by interested parties
- expert monitoring using measurements and remote sensing (usually complemented by an independent audit).

Insights from Principal Agent Models1 suggest that the most cost-effective schemes are likely to have a structure where the provider has an incentive to monitor and truthfully report on their own actions (Laffont and Martimort, 2002). For example, the collective action scheme promoted by Bio-Rights2 (van Eijk and Kumar, 2009) provides community members with micro-credit for sustainable projects. If they deliver the agreed changes in land use they receive a payment equivalent to the

Box 4. Multi-stakeholder monitoring in Los Negros, Bolivia

This scheme did not establish baseline water flows and bird species diversity before activities began, as it concentrated on vegetative cover and an assumed relationship with environmental services. Basic hydrological relationships were not explored, and remain unknown and assumed. Forest conservation plots are measured and demarcated with hand-held GPS receivers and plotted onto a land-use map based on satellite images from 2001. Fields are demarcated according to natural boundaries or trails, signs and wire fencing. The various forest types in the parcel are then mapped and their areas calculated, and farmers are given a copy of 'their' map along with their contract.

The project control team, which consists of one member from upstream and one from downstream environmental committees, a nature field technician and the landowner, visits each enrolled property once a year. They receive US$20 per diem to monitor changes in land use, using GPS and maps, and submit a report to the Enforcement Directorate (the President of Natura and presidents of the upstream and downstream environmental committees). The Directorate makes recommendations on how to respond to infractions, if necessary.

Monitoring was recently extended to measuring water flows in several tributaries, and an avifaunal survey was conducted in 2005. Recent results show that the scheme has not had an impact on additionality as the land set aside for conservation is the least threatened by agricultural clearance (Robertson and Wunder, 2005), and a rapid hydrological assessment showed no initial impact on water flows (Le Tellier *et al.*, 2009). Several universities, including Vu University, Amsterdam, are now involved in a larger study in Rio Grande (2010–2014). See http://www.naturabolivia.org/eva2.html

Source: Asquith *et al.* 2008

original loan (which by then would have been repaid). With this type of initiative, and for group monitoring (see below), participants such as forest owners and households need clearly specified measurement protocols to follow when collecting data for monitoring. The Myrada project, which works in 32 micro-watersheds in India, reports that loans (rather than contributions) improve people's motivation to better monitor and manage investments (Prakash Fernandez, 2003).

Participatory monitoring can take two forms: monitoring by providers, and monitoring through multi-stakeholder groups that may include service buyers. In the first case, service providers monitor each other while the buyer (intermediary, government, direct user) monitors the group and holds the group accountable for the inputs and outcomes. Monitoring by peers is useful in systems where service delivery is contingent upon multiple resource-management units, and has proved successful in community forest management contracts in India. Social disapproval can be an important control tool, and has been suggested as an effective means of ensuring transparency and participation in REDD+ projects (Skutsch *et al.*, 2009). Group pressure is useful in group contracts where an entire group of service providers has to bear the cost of a single individual's non-compliance.

The second type of monitoring, by multi-stakeholder groups, is more common in small schemes where representatives from the local communities, municipalities and/or NGOs concerned form a voluntary commission and make field visits and recommendations. This method has been used in Los Negros in Bolivia (Asquith *et al.*, 2008b), San Pedro Norte, Nicaragua (Marín *et al.*, 2006; Porras *et al.*, 2008) and Fidecoagua, Coatepec in Mexico. However, it can be ineffective if the groups are weak and have limited options to enforce cooperative behaviour and deal with free-riders, like the short-lived Group Contracts in the PSA in Costa Rica (Porras, 2010).

Remote sensing can be a useful monitoring tool when checking for rough changes in forest cover. Some national schemes, like the PSAH in Mexico, the PSA in Costa Rica and BioBosque in Ecuador, use it in tandem with independent audits by forest regents. Several models are also available free of charge on the internet, although most have been developed for use within the United States (see RedLAC, 2010, for examples). However, remote sensing is not useful for monitoring actual changes in watershed services, and can only provide a blunt measure of whether land cover has changed or not.

3.2.4 Enforcement, penalties and gaps

Compliance and enforcement mechanisms should be implemented in the early stages of a scheme, although some have been set up as 'learning by doing' exercises, as in Pimampiro (Echavarria *et al.*, 2004). It has to be said that compliance monitoring is typically less than optimal, as inadequate funding, lack of institutional capacity and capabilities (shortages of enforcement staff) and poor communication between stakeholders, intermediaries and regulatory officials all serve to weaken compliance monitoring and enforcement (Echavarria *et al.*, 2004; Wunder and Albán, 2008).

Sanctions for non-compliance usually involve a period of exclusion from the scheme, the cancellation or suspension of payments (Claassen *et al.*, 2008; Echavarria *et al.*, 2004; Wunder and Albán, 2008) or the threat of civil legal action if contracts are breached, as in the PSA Programme in Costa Rica. Observed and anticipated penalties, the likelihood of enforcement and the perceived threat of sanctions can help reduce the costs associated with monitoring (as with self-enforced monitoring), particularly in smaller communities where social pressure and learning have the potential to encourage cooperation (Wunder and Albán, 2008; Wunder *et al.*, 2008). However, information asymmetries (hidden actions or moral hazard) can affect conditionality if conservation agents find it expensive to monitor contract compliance and politically costly to sanction non-compliance (especially in contracts with poor groups) – in which case they will fail to enforce the contract. High fines are often used as a deterrent to non-compliance, but the voluntary nature of PWS limits the range of sanctions that can be applied, meaning that landowners have an incentive to breach their contractual responsibilities (Pattanayak *et al.*, 2010; Wunder, 2005).

Nevertheless, the payments themselves can act as a mechanism to encourage compliance. When participation is properly monitored and payments are truly conditional, providers 'learn' to comply, particularly when the level of payment properly compensates them for the services rendered (Claassen *et al.*, 2008; Kosoy *et al.*, 2007).

In short, compliance monitoring and enforcement are necessary to demonstrate that watershed actions are delivered according to the terms of the contract. Being clear about what action is required is one thing; what is much harder is establishing that taking such action actually leads to changes in land management practices, as these might have occurred without the scheme. This leaves us with the question of whether examining the efficacy of contracted-for actions is sufficient to demonstrate service delivery, or whether hydrological monitoring is also necessary.

3.3 Demonstrating the effectiveness or additionality of a scheme

The land management or production practices that are incentivised through PWS schemes are intended to cause an effect: to change watershed or hydrological functions. In the context of PES, additionality can be defined as a scheme's overall ability to instigate the intended outcomes, which would not have happened if it had not been implemented (Claassen *et al.*, 2008; Muñoz *et al.*, 2005; Shrestha and Timilsnia, 2002). This change in function is presumed to have consequences for human wellbeing in terms of economic costs and benefits or welfare, particularly for the poor. Here we focus on monitoring to determine whether or not a PWS scheme can be said to be environmentally effective.

3.3.1 Indicators and baselines

When looking at watershed functions, we need to measure one or more components of water quality and quantity in order to show an effect. This often involves measuring temperature and reductions in sediment, nutrient and chemical loads to assess water quality, and hydrographic features such as base, peak or low flows for a quantitative assessment. A substantial baseline may be required to demonstrate both qualitative and quantitative environmental additionality – but even when such data are available, it can be difficult to compare sources. The hydrological records for water flow that do exist are from gauges that have been operating for a long time, while technologies for monitoring remote and continuous water quality are much more recent.

Asquith *et al.* (2008) highlight the importance of composing a baseline for monitoring, noting that sound measurement and evaluation of PES schemes requires a preliminary understanding of the biophysical interconnections in targeted areas. Most baselines have focused on measuring forest cover in properties, usually using ground-truth (information collected on location) GIS-based technology supplemented with some ground measurement, rather than measuring the quality or quantity of the watershed service. The Working for Water programme in South Africa is an exception to this trend, in that it established a hydrological baseline before commencing activities (Turpie *et al.*, 2008). This is not to say that projects do not monitor changes in watershed services; rather, that it tends to be done with less vigour than for changes in land use. For example, FONAG in Ecuador has installed fairly sophisticated weather stations and soil moisture monitoring stations (Cannon *et al.*, 2010), while Los Negros in Bolivia undertakes hydrological monitoring to look at changes in water quantity and biota (see box above). The Working for Water programme in South Africa monitors the number of alien species removed from river edges, but only estimates changes in water quantity through modelling (Ferraro, 2009). All monitoring efforts have to deal with the aforementioned problem of attribution and the difficulty of establishing causal links with any degree of certainty, due to patchy or short-term data collection, stochastic events (such as weather patterns) masking local impacts, and uncertainty about groundwater movements.

Schemes in rich countries have been more successful in using baselines to monitor outputs (the level of ecosystem services provided). It is not clear why this practice is not more widespread in low- and middle-income countries, but it may be linked to existing capacities, access to technology and increasing demand for proof of how such programmes work. The Hunter River Salinity Trading scheme in New South Wales, Australia, has been successful in managing salinity levels along the river. This scheme monitors and caps discharges from 23 coalmining and power generation facilities, and allows credits to be traded depending on the flow of the river (NSW, 2009). The Vittel project in France monitors water quality extensively in all participating farm sites (Perrot-Maitre, 2006); while the conditional land tenure scheme in Sumberjaya is an early example of payments for reduced sediments that had the added value of creating a market use for captured sediments (Harto Widodo *et al.*, 2006).

Apart from these isolated examples, very few schemes have developed measuring programmes built on a sound baseline for ecosystem services (Agarwal *et al.*, 2007; Porras *et al.*, 2008). The fact that some early schemes failed to establish a baseline for prior land management or practices also constitutes a fundamental obstacle to monitoring their basic additionality.

3.3.2 Attribution

Demonstrating environmental additionality involves much more than having a prior record measuring the desired hydrological variable. It is also necessary to understand the relationship between the action and that variable – which can create further obstacles even where baseline data on water quantity or quality do exist. The key issue here is attribution: understanding whether factors other than the target actions affect water quality and quantity over time, and how they do so. This implies the ability to consult data on potentially confounding drivers as well as on the principle action concerned. While this can be quite difficult and time-consuming with regard to PWS, failure to take this crucial step may lead to faulty conclusions about cause and effect, with PWS programmes deemed to be successful when

Box 5. Collecting information

So how much research is needed before and during the implementation of PWS schemes? Information is very valuable, but it does come at a cost. The amount of research needed to implement these schemes varies according to the nature and scale of the problem, and the level of funding available for this type of study. The Bellagio Conversations (Asquith and Wunder, 2008) provide a useful rule of thumb for starting a PWS scheme. The simplest schemes involve conserving existing ecosystems that are under threat in order to maintain water quality or quantity. Mechanisms for such schemes can be based on the precautionary principle, and studies can follow in time, as in the case of Los Negros in Bolivia (Asquith, Vargas and Wunder, 2008). Restoring habitat is more complicated, as research will be required to demonstrate bio-geochemical linkages with water quality, threshold levels and cost-effectiveness; while research on water quantity is even more complicated if no site-specific information is already available. The wisest course of action in such cases is to undertake a series of inexpensive 'no-regret' actions until more in-depth research has been conducted to determine whether or not to implement a full-scale PWS scheme.

Van Noordwijk (2005) provides a useful description of the types of watershed services, key indicators, ways of measuring them (or proxies if needed) and how these indicators can be monitored locally. In their book about forests, water and people in the tropics, Bonell and Bruijnzeel (2005) present several papers on collecting information that cover remote sensing technologies, statistical methods and selecting models. Although they have their limitations, these methods currently provide the best information about the impacts of land-use changes on water variables.

they actually had little or no impact on watershed services. Conversely, a lack of evident outcomes may be linked to external factors other than programme design.

The Maasin watershed programme in the Philippines clearly illustrates the problems that can arise with attribution. There was a marked reduction in water flows and significant siltation in the area three years after this programme started. While some blamed the fast-growing species used by the scheme (mainly mahogany and gmelina), other factors also needed to be taken into account, such as the fact that the water utilities were using more water than before, and that siltation had increased as farmers displaced by the programme tilled new land. In the event, none of these hypotheses were explored, and the arguments stopped when the rains restarted (Salas, 2004).

3.3.3 Targeting strategies

Recent studies assessing the additionality of PWS schemes indicate that enrolment-based (voluntary) mechanisms may negatively influence the efficiency and effectiveness of a programme, as it may end up with lands that are at low risk of degradation or have little influence over hydrological changes (Asquith and Wunder, 2008; Hoffman, 2009; Kalacska *et al.*, 2008; Muñoz *et al.*, 2005; Pfaff *et al.*, 2008; Robertson and Wunder, 2005). For voluntary contracts like those promoted in PWS schemes, targeting and/or the use of preferential criteria for contract allocation can help increase environmental effectiveness (Sen, 1996). Monitoring the type of targeting used and the scheme's ability to enforce it can help indicate the potential impact on the watershed service.

Effective targeting entails selecting an appropriate location and choosing resources that will deliver services of the required quality. PWS schemes typically involve one buyer (usually the government) with monopsonic powers, including the ability to select and change prices, which is an important tool in targeting through price differentiation.

Critical areas can be targeted by directing payments towards watersheds that are important for human watershed services (domestic water use or hydroelectricity), areas that are prone to degradation (high slopes, river edges) or at greater risk from changes in land use (such as forest clearance for agriculture).

It is easier for local schemes to target precise areas around the headwaters of the main water supply (such as Pimampiro in Ecuador, Tacuba in El Salvador, Jesus de Otoro in Honduras, and San Pedro Norte in Nicaragua). Wunder and Albán (2008) argue that targeting the small-scale scheme in Pimampiro has helped stop deforestation and contributed to the marked recovery of native vegetation, bucking the trends in most neighbouring villages. Approximately 30 per cent of the total area had been converted to cropping and pastures before payments were introduced in 2000, compared with just 14 per cent in 2005. It is hard to determine the scheme's effect on water availability as there are no baseline studies or appropriate counterfactuals to draw on, and downstream perceptions of increased water flows may be influenced by improvements to the infrastructure introduced at the start of the PES scheme (Echavarria *et al.*, 2004). In Los Negros, Bolivia, lack of targeting and very low rates of compensation (covering only 2–10 per cent of the opportunity costs of setting aside agricultural land) meant that the areas selected by farmers for the scheme had less agricultural potential and were therefore at less risk of conversion (Asquith *et al.*, 2008b; Robertson and Wunder, 2005). Better targeting critical areas may result in fewer participants, but can make a scheme more attractive by increasing individual levels of compensation (Asquith *et al.*, 2008b; Wunder, 2008).

Lack of targeting can also dilute the potential benefits of large schemes, especially national ones. Analysis of the Sloping Land Conversion programme in China, which took a very top-down approach, shows that 38 per cent of the area converted from agriculture to forestry in Gansu Province was on low slopes, and hence at little risk of causing erosion (the same applied to 10 per cent of affected land in Shanxi and 11 per cent in Sichuan). Although these new forest areas may be of some benefit for biodiversity and landscape, it is doubtful that they would have much impact on sediments downstream (Bennett and Xu, 2008; Xu *et al.*, 2004). Using treatment analysis, Xu *et al.* (2010) demonstrate that increasing local autonomy could lead to better local targeting, more cost-effective programmes and improved outcomes.

Landowners involved in schemes in Norway and Finland determine the supply of potential targets, and some voluntary conservation contracts have been criticised for lack of targeting as they can result in a conservation network that does not cover all the focal ecological characteristics (Barton, 2010). Although early analysis of Natural Values Trading in Finland shows that the programme is meeting its ecological goals, it is still too early to assess its long-term ecological effects (Juutinen *et al.*, 2009).

The first-come-first-served approach initially used in national schemes in Costa Rica and Mexico was notorious for its minimal impact on at-risk lands. Most of the area covered by protection contracts in Costa Rica's Osa Peninsula is forestland that may not be in direct danger of conversion due to its isolated and inaccessible location (Sierra and Russman, 2006). Analysis by Sanchez-Azofeifa *et al.* (2007) found that there was no significant difference in deforestation rates from 1997 to 2000 in areas covered by the national PSA scheme and those that were not, although this can be debated given the low rates of deforestation in the late 1990s and early 2000s. Barton *et al.* (2009) showed that efficiency levels improved in the next phase of PES in Costa Rica due to better targeting in the period between 1999 and 2001. Nevertheless, the continued lack of focus on areas at greatest risk from deforestation is reflected in the programme's negligible impact on deforestation (Robalino *et al.*, 2008; Sánchez-Azofeifa *et al.*, 2007).

Mexico has gone through several stages in terms of targeting criteria. Contracts were initially distributed on an ad-hoc basis, and much of the land concerned was at little risk of being converted because of its low opportunity costs. A spatial model created by Munoz-Piña *et al.* (2005) showed that in 2003 only 11 per cent of the land covered by the scheme was classified as being at high or very high risk from deforestation, although this increased to 28 per cent in 2004. The introduction of priority eligibility criteria increased the focus on protecting forests in aquifers and areas affected by water scarcity, risk of flooding and other disasters associated with extreme weather. A combination of environmental, administrative and social eligibility criteria is now in place (rising from nine in 2006 to 26 by 2010), but secondary criteria are given more weight than primary criteria, and only one third of the land covered by the scheme is at significant risk from deforestation. Non-environmental criteria are increasingly hampering the programme's ability to be environmentally effective (Alix-Garcia *et al.*, 2005a; Muñoz-Piña *et al.*, 2008; Muñoz-Piña *et al.*, 2011).

The national Socio-Bosque PES programme in Ecuador shows that some lessons have been learned from Costa Rica and Mexico, as it includes the risk of deforestation as a criterion for participation, along with local poverty levels and the proportion of forest in protected areas.

On the whole, national programmes in Costa Rica, Mexico and China have not been very successful in targeting more at-risk land (Alix-Garcia *et al.*, 2005b; Alix-Garcia *et al.*, 2010; Bennett and Xu, 2008; Sánchez-Azofeifa *et al.*, 2007). Their additionality is significantly reduced by concentrating on conserving low-risk forests, as all the payments in Mexico and a very large proportion of those in Costa Rica are for conservation. There are many reasons for this, including political interests, pressure from various groups and the governments' tendency to 'learn on the job'. However, a new generation of programmes that take more serious account of risk is emerging, due to increasing demand from farmers, tighter funding and pressure from academic groups that have identified and publicised various inefficiencies. Mexico is moving from an ad-hoc to a more risk-based approach to contracts. It remains to be seen whether existing levels of payment will be enough to attract farmers in targeted areas, or whether a system of differentiated payments will be required – as in Costa Rica, which has introduced higher payments for hydrologically sensitive areas that are usually located near city centres and have higher opportunity costs.

3.4 Monitoring unintended impacts

As with other projects, PES schemes can have unintended impacts. For example, leakage (or slippage) may occur when the efforts of PES or conservation schemes are offset by degradation in other areas managed by members of the scheme. This is usually associated with the extension of cropland outside areas covered by the PES scheme while land within the scheme is managed according to programme specifications (Claassen *et al.*, 2008; Hoffman, 2009). It is less of an issue in local PWS schemes that target specific catchments for participation, and potentially more of a problem in bigger schemes, which are often government-funded (Ross *et al.*, 2006). Because of the largely unknown nature of inter-basin dynamics, one cannot assume that degradation caused by leakage outside the project area will not affect the hydrology within it.

Perverse incentives are another unintended consequence of PES schemes: people have been known to intentionally degrade land in order to receive compensation for restoration efforts specified by a PES programme, or stop restoration work so that they receive payments (Wunder and Albán, 2008).

The high cost of monitoring impacts within programme sites means that leakage is often only mentioned in passing, if at all. There is much to be learned from the way that this issue is treated in developing carbon markets, and there are approved methodologies for quantifying these impacts (see, for example, the Voluntary Carbon Standard, VCS). The FAO has developed an ex-ante strategy whereby monitoring mechanisms are designed to identify perverse incentives in order to determine the potential impact of their effects and the likelihood of their occurrence (FAO/REDLACH, 2004).

4. Conclusions

As the levering capacity to demand payments for better watershed management increases, so does the need to demonstrate the impact of such activities. Interest in market-based solutions like payments for watershed services continues to grow, in many cases complementing national or local regulations designed to raise funds (targeting service users) and restrict damaging practices (targeting service providers).

There is a fine line between obtaining enough scientific information to develop and monitor a project and ensuring that it remains cost-effective as costs increase. By definition, the attractiveness of PWS schemes relies on their ability to create incentives to change behaviour, and on such changes generating net positive impacts on the level of watershed services.

This article discusses the monitoring and evaluation criteria behind compliance or transactional monitoring, which ensures that contracts are followed, and effectiveness conditionality, which looks at how a scheme has managed to achieve its environmental objectives regardless of the degree of compliance. Although the two are usually linked, a high degree of compliance does not necessarily ensure that a scheme is effective, as a poorly designed scheme may target the wrong land managers and land that is at least risk, meaning that the payments may not generate the desired hydro-ecological or conservation benefits.

At the moment most schemes in low- and middle-income countries focus on monitoring contract compliance, which could usually be improved. Inadequate funding, lack of institutional capacity and capabilities (often due to staff shortages) and poor communication between stakeholders, intermediaries and regulatory officials all serve to undermine compliance monitoring and enforcement. Although increasing monitoring does generate higher transaction costs, it can ultimately help determine whether PWS is a credible instrument that delivers more than ancillary benefits for local communities and corporate responsibility for buyers.

Demonstrating environmental additionality involves much more than having a prior record measuring the desired hydrological variable. It is also necessary to understand the relationship between the action and that variable – which can create further obstacles even where baseline data on water quantity or quality do exist – and remember that a lack of evident outcomes may be linked to external factors other than programme design.

The key issue here is attribution: understanding whether factors other than the target actions affect water quality and quantity over time, and how they do so. This implies the ability to consult data on potentially confounding drivers as well as on the principle action concerned, and a sound theory of change behind the scheme. While it can be quite difficult and time-consuming, failure to take this crucial step may lead to faulty conclusions about cause and effect, with PWS programmes deemed successful when they actually had little or no impact on watershed services. If no baseline information is available and no cause and effect relationship is established, actors will have to rely on scientific extrapolation or local knowledge to develop the PWS scheme, and it will be impossible for monitoring processes to make any meaningful links between cause and effect. All that can be done in such cases is to monitor contract compliance and the relevant hydrological variables. If hydrological conditions do not improve then it can be intuited that the PWS scheme is not having the degree of effect anticipated, but it will be impossible to establish why this is so and whether the situation might have been worse without the introduction of the scheme. If conditions do improve, the only conclusion that can be drawn is that the PWS scheme has probably not adversely affected the situation. There will be no proof that the improvement is due to the PWS scheme.

Better understanding of these relationships is vital for the long-term health of existing initiatives. The growing interest in such schemes shows that participants believe in the principle of land management, and the emergence of better-designed schemes that rely on a more scientific approach is grounds for cautious optimism about the potential of PWS. Nevertheless, evidence of impact will be required to ensure that actions really do generate added value, and to make the case for continued promotion of these schemes.

5. References

Agarwal, C., Tiwari, S., Borgoyary, M., Acharya, A. and Morrison, E. 2007. Fair deals for watershed services in India. Natural Resource Issues. International Institute for Environment and Development (IIED), London.

Alix-Garcia, J., de Janvry, A., Sadoulet, E., Torres, J.M., Braña Varela, J. and Zorilla Ramos, M. 2005a. An assessment of Mexico's payment for environmental services program, roles of agriculture project environmental services. Agricultural and Development Economics Division (ESA) of the FAO, Italy.

Alix-Garcia, J., Janvry, A.d. and Sadoulet, E. 2005b. A tale of two communities: explaining deforestation in Mexico. World Development 33(2): 219–35.

Alix-Garcia, J., Shapiro, E. and Sims, K. 2010. Forest conservation and slippage: evidence from Mexico's National Payments for Ecosystem Services Programme. Working paper, 6 August.

Alpizar, F., Blackman, A. and Pffaf, A. 2007. Payments for ecosystem services; measurements with impact. Resources 165, Resources for the Future, Washington DC.

Appleton, A.F. 2002. How New York City used an ecosystem services strategy carried out through an urban–rural partnership to preserve the pristine quality of its drinking water and save billions of dollars, and what lessons it teaches about using ecosystem services. The Katoomba Conference Tokyo, November.

Ardón, M. and Barrantes, G. 2003. Sistematización de experiencias piloto de PSA relacionadas con los recursos hídricos a nivel municipal (acción conjunta entre los Proyectos Regionales PASOLAC y CBM). Experiencias de PSA en: Jesús de Otoro, Campamento (Honduras) Tacuba, Gualabo (El Salvador) y San Pedro del Norte, Regadío (Nicaragua). Prepared for PASOLAC and the Mesoamerican Biological Corridor initiative, Central America.

Arriagada, R.A., Ferraro, P., Sills, E., Pattanayak, S. and Cordero, S. 2010. Do payments for environmental services reduce deforestation? A farm level evaluation from Costa Rica, Working Paper from Georgia State University.

Asquith, N., Vargas, M. and Wunder, S. 2008. Selling two environmental services: in-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. Ecological Economics 65(4): 675–84.

Asquith, N. and Vargas, M.T. 2007. Fair deals for watershed services in Bolivia. Natural Resource Issues, IIED, London.

Asquith, N. and Wunder, S. 2008. Payments for watershed services: the Bellagio Conversations. IIED, Blue Moon Fund, CGIAR, DFID, EcoFund Foundation.

Aylward, B. 2005. Towards watershed science that matters. Hydrological Processes 19(13):2643–647.

Barton, D. 2010. Assessment of existing and proposed policy instruments for biodiversity conservation: Norway case study. Report for the Policymix project, prepared by the Norwegian Institute for Nature Research (NINA).

Barton, D., Faith, D.P., Rusch, G., Acevedo, M. and Castro, M. 2009. Environmental service payments: evaluating biodiversity conservation trade-offs and cost-efficiency in the Osa Conservation Area, Costa Rica. Journal of Environmental Management 90(2): 901–11.

Baylis, K., Peplow, S., Rausser, G. and Simon, L. 2008. Agri-environmental policies in the EU and United States: a comparison. Ecological Economics 65(4):753–64.

Bennett, M.T. and Xu, J. 2008. China's Sloping Land conversion program: institutional innovation or business as usual? Ecological Economics 65(4) 699–711.

Berttram, D. 2011. Positioning the Kagera TAMP project in the PES landscape of East Africa, FAO (NRL), Italy.

Blackman, A. and Woodward, R.T. 2010. User financing in a national payments for environmental services program: Costa Rican hydropower. Ecological Economics 69(8):1626–638.

Blanco, R. and Rojo, P. 2005. Fideicomiso para el pago de servicios ambientales forestales Coatepec, Veracruz.

Bond, I., 2008. Payments for ecosystem services in east and southern Africa: assessing prospects and pathways forward, In: Katoomba Group (ed.), Scoping of Potential Payment for Ecosystem Service Sites. The Katoomba Group. Washington D.C

Bond, I., Grieg-Gran, M., Wertz-Kanounnikoff, S., Hazlewood, P., Wunder, S. and Angelsen, A. 2009. Incentives to sustain forest ecosystem services: a review and lessons for REDD. Natural Resource Issues. IIED, London, UK with CIFOR, Bogor, Indonesa and World Resources Institute, Washington DC, USA.

Bonell, M., Bruijnzeel, L.A., 2005. Forest, Water and People in the Humid Tropics. Cambridge University Press.

Bruijnzeel, L.A. 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? Agriculture, Ecosystems and Environment 104(1):185–228.

Bruijnzeel, L.A., Kappelle, M., Mulligan, M. and Scatena, F.N. 2010. Tropical montane cloud forests: state of knowledge and sustainability perspectives in a changing world. In: Bruijnzeel, L.A., Scatena, F.N. and Hamilton, L.S. (eds) Tropical montane cloud forests: science for conservation and management. Cambridge University Press, Cambridge, UK.

Buytaert, W., Celleri, R., De Bievre, B., Cisneros, F., Wyseure, G., Deckers, J. and Hofsteade, R. 2006. Human impact on the hydrology of the Andean páramos. Earth-Science Reviews 79, 53–72.

Calder, I. 2005. The Blue Revolution: Land use and integrated water resources management. Earthscan, London, UK.

Calder, I.R. and Aylward, B. 2006. Forest and floods: moving to an evidence-based approach to watershed and integrated flood management. Water International 31(1).

Canales, E. 2006. Cuencas y Ciudades II, un proyecto de recaudacion voluntaria. Zona sujeta a conservacion ecologica, Sierra de Zapaliname, Coahuila, Mexico. International Workshop for Ecosystem Services: Products and Market Development. Forest Trends/Katoomba group and the Sierra Gorda Ecological Group, San Juan del Rio, Querétaro. Mexico.

Cannon, P., Hill, B. and McCarthy, C. 2010. Watersheds of Quito: A consultancy to Bolster Fonag's contribution. Technical report. USAID, USA.

Casasola, F., Ibrahim, M., Sepúlveda, C., Ríos, N. and Tobar, D. 2009. 'Implementación de sistemas silvopastoriles y el pago de servicios ambientales en Esparza, Costa Rica: una herramienta para la adaptación al cambio climático en fincas ganaderas'. In: Sepúlveda C.and Ibrahim M. (eds) Políticas y sistemas de incentivos para el fomento y adopción de buenas prácticas agrícolas: como una medida para adaptación al cambio climático en América Central, Turrialba.

CCI, BirdLife International. 2011. Measuring and monitoring ecosystem services at the site scale. Cambridge Conservation Initiative and BirdLife International, Cambridge, UK.

Claassen, R., Cattaneo, A. and Johansson, R. 2008. Cost-effective design of agri-environmental payment programs: U.S. experience in theory and practice. Ecological Economics 65(4):737–52.

Crespo, P., Celleri, R., Buytaert, W., Feyen, J., Iñiguez, V., Borja, P. and de Bievre, B. 2009. 'Land use change impacts on the hydrology of wet Andean páramo ecosystems'. In: Status and Perspectives of Hydrology in Small Basins (Proceedings of the Workshop held at Goslar-Hahnenklee, Germany, 30 March–2 April 2009). IAHS Publ. 336, 2010, 71–76.

Currie, J. and Gahvari, F. 2007. Transfers in cash and in kind: theory meets the data. Journal of Economic Literature, American Economic Association 46(2):333–83.

Dasgupta, P. 2007. Report on the self-evaluation of RUPES. World Agroforestry Centre, Bogor, Indonesia.

DWAF. 2006. Working for water strategic Plan 2004/2005–2007/2008. Internal draft South African Department of Water Affairs and Forestry, Pretoria.

Echavarria, M., Vogel, J., Alban, M. and Meneses, F. 2004. The impacts of payments for watershed services in Ecuador: emerging lessons from Pimampiro and Cuenca. IIED, London.

EEA, 2010. Distribution and targeting of the CAP budget from a biodiversity perspective. EEA Technical Report, European Environment Agency, Copenhagen.

Engel, S., Pagiola, S. and Wunder, S. 2008. Designing payments for environmental services in theory and practice: an overview of the issues. Ecological Economics 65(4), 663–74.

Espinoza, A. Undated. Mecanismos de compensación relacionando bosques con agua en Centroamérica y El Caribe de habla hispana. INAFOR, FAO, p. 11.

FAO, 2010. Informe técnico Visita 1: Intercambio de experiencias Iniciativa PSAH Estelí, El Regadío. Instituto Nacional Forestal, FAO-facility, Managua.

FAO/REDLACH, 2004. Electronic forum on payment schemes for environmental services in watersheds. Final report. FAO, Italy.

Ferraro, P. 2009. Regional review of payments for watershed services: Sub-Saharan Africa. Journal of Sustainable Forestry 28, 525–50.

Ferraro, P.J. and Pattanayak, S.K. 2006. Money for nothing? A call for empirical evaluation of biodiversity conservation investments. PLoS Biology 4(4):482–88.

Freeman, A.M. 1993. The measurement of environmental and resource values. Resources for the Future. Washington DC.

Grieg-Gran, M., Porras, I. and Wunder, S. 2005. How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. World Development 33(9):1511–527.

Gutrich, J., Donovan, D., Finucane, M., Focht, W., Hitzhusen, F., Manopimoke, S., McCauley, D., Norton, B., Sabatier, P., Salzman, J. and Sasmitawidjaja, V. 2005. Science in the public process of ecosystem management: lessons from Hawaii, Southeast Asia, Africa and the US Mainland. Journal of Environmental Management 76(3):197–209.

Harto Widodo, R., Suyanto, S., Verbist, B., Lusiana, B. and Purnomosidhi, P. 2006. Rewarding communities for keeping rivers clean? First steps in a RiverCare Program in West Lampung, Indonesia. International Symposium on Community Activities for the Conservation of Water Environment. ICRAF, Bangkok, Thailand.

Hoffman, C.A. 2009. Evaluation of payments for ecosystem services in the Valley Region of Bolivia. Nicholas School of the Environment, Duke University, Durham, NC, USA.

Holwerda, F., Bruijnzeel, L.A., Muñoz-Villers, L.E., Equihua, M. and Asbjornsen, H. 2010. Rainfall and cloud water interception in mature and secondary lower montane cloud forests of central Veracruz, Mexico. Journal of Hydrology 384(1–2):84–96.

Ibrahim, M., Casasola, F., Villanueva, C., Murgueitio, E., Ramirez, E., Sáenz, J. and Sepúlveda, C. 2010. Payment for environmental services as a tool to encourage the adoption of silvo-pastoral systems and restoration of agricultural landscapes dominated by cattle in Latin America. CATIE, Costa Rica.

IFAD, 2012. Upper Tana Catchment natural resource management project. Project design report.The International Fund for Agricultural Development, Italy.

Jeanes, K., van Noordwijk, M., Joshi, L., Widayati, A., Farida and Leimona, B. 2006. Rapid hydrological appraisal in the context of environmental service rewards. World Agroforestry Centre – ICRAF, SEA Regional Office, Bogor, Indonesia, p. 56.

Juutinen, A., Mönkkönen, M. and Ylisirniö, A.-L. 2009. Does a voluntary conservation program result in a representative protected area network? The case of Finnish privately owned forests. Ecological Economics 68(12):2974–984.

Kaimowitz, D. 2004. 'Useful myths and intractable truths: the politics of the links between forests and water in Central America'. In: Bonell, M. and Bruinjzeel, L.A. (eds) Forests, Water and People in the Humid Tropics: Past, present, and future hydrological research for integrated land and water management. Cambridge University Press, Cambridge, UK, pp. 86–98.

Kalacska, M., Sanchez-Azofeifa, G.A., Rivard, B., Calvo-Alvarado, J.C. and Quesada, M. 2008. Baseline assessment for environmental services payments from satellite imagery: a case study from Costa Rica and Mexico. Journal of Environmental Management 88(2):348–59.

Kosoy, N., Martínez-Tuna, M., Muradian, R. and Martínez-Alier, J. 2005. Payment for environmental services in watersheds: insights from a comparative study of two cases in Central America. Convention on Protection and Use of Transboundary Watercourses and International Lakes, Geneva.

Kosoy, N., Martínez-Tuna, M., Muradian, R. and Martínez-Alier, J. 2007. Payments for environmental services in watersheds: insights from a comparative study of three cases in Central America. Ecological Economics 61(2–3), 446–55.

Kroeger, T., Casey, F. 2007. An assessment of market-based approaches to providing ecosystem services on agricultural lands. Ecological Economics 64(2): 321–32.

Laffont, J.-J. and Martimort, D. 2002. The Theory of Incentives. Princeton University Press, Princeton NJ, USA.

Landell-Mills, N. and Porras, I. 2002. Silver bullet or fools' gold? A global review of markets for forest environmental services and their impacts on the poor. IIED, London.

Le Maitre, D.C., Versveld, D.B. and Chapman, R.A. 2000. The impact of invading alien plants on surface water resources in South Africa: a preliminary assessment. Water SA 26, 397–408.

Le Tellier, V., Carrasco, A. and Asquith, N. 2009. Attempts to determine the effects of forest cover on stream flow by direct hydrological measurements in Los Negros, Bolivia. Forest Ecology and Management 258(9): 1881–888.

Lechuga, C. 2009. Zapalinamé: connecting cities and watersheds in Mexico. Ecosystems Marketplace, Washington DC.

Lee, M. 2012. Coffee farmers in Peru look to carbon market to fund climate adaptation. The Ecologist, May 14.

Leshan, J., Xiaoyun, L. and Ting, Z. 2005. Development contract with terms of watershed conservation: a win-win opportunity for development and environment in the Meijiang Watershed, Ningdu County, Jiangxi Province, China. China Agricultural University, Beijing.

Manson, R.H. 2008. Efectos del uso del suelo sobre la provisión de servicios ambientales hidrológicos: monitoreo del impacto del PSAH. Informe Final. Instituto de Ecología, A.C.

Marín, X., Ogier, M., Pérez, C. and Martínez, M. 2006. Elementos metodológicos para la implementación de pagos por servicios ambientales hídricos al nivel municipal en Centroamérica Programa para la Agricultura Sostenible en Laderas de América Central PASOLAC. Tegucigalpa, p. 37.

Mayer, A.L. and Tikka, P.M. 2006. Biodiversity conservation incentive programs for privately owned forests. Environmental Science & Policy 9(7–8):614–25.

Montagnini, F. and Finney, C. 2010. Payments for environmental services in Latin America as a tool for restoration and rural development. AMBIO: A Journal of the Human Environment 40(3):285–97.

Muñoz-Piña, C., Guevara, A., Torres, J.M. and Braña Varela, J. 2008. Paying for the hydrological services of Mexico's forests: analysis, negotiations and results. Ecological Economics 65(4):725–36.

Muñoz-Piña, C., Rivera, M., Cisneros, A. and García, H. 2011. Retos de la focalización del Programa de Pago por los Servicios Ambientales en México. Revista Española de Estudios Agrosociales y Pesqueros 228: 87–112.

Muñoz-Villers, L.E., Holwerda, F., Gomez-Cardenas, M., Equihua, M., Asbjornsen, H., Bruijnzeel, L.A., Marin-Castro, L. and Tobon, C. 2010. Water balances of old-growth and regenerating montane cloud forests in central Veracruz, Mexico. Journal of Hydrology draft.

Muñoz, C., Guevara, A., Bulas, J.M., Torres, J.M. and Braña Varela, J. 2005. Paying for the hydrological services of Mexico's forests.Ecological Economics, 65(4): 725–36.

NSW, 2009. Hunter River salinity trading scheme: 2008–09 performance. New South Wales Government: Environment, Climate Change and Water, Sydney.

Obando, M. 2007. Evolución de la experiencia de los PSA hídricos en Nicaragua: El caso de los Caballos, Municipio de San Pedro del Norte, Chinandega, Programa para la Agricultura Sostenible en Laderas de América Central. PASOLAC, SDC, COSUDE, Tegucigalpa.

Ortiz, E., Sage, L.F. and Borge, C. 2003. Impacto del Programa de Pago por Servicios Ambientales en Costa Rica como medio de reducción de la pobreza en los medios rurales. RUTA, San José.

Ostrom, E. 1990. Governing the commons. The evolution of institutions for collective action. Political economy of institutions and decisions. Cambridge University Press, Cambridge, UK.

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

Pattanayak, S.K., Wunder, S. and Ferraro, P. 2010. Show me the money: do payments supply environmental services in developing countries? Review of Environmental Economics and Policy 4(2):1–21.

Perrot-Maitre, D. 2006. The Vittel payments for ecosystem services: a 'perfect' PES case? IIED, London, UK.

Pfaff, A., Robalino, J. and Sánchez-Azofeifa, G.A. 2008. Payments for environmental services: empirical analysis for Costa Rica. Working Papers Series, Terry Stanford Institute of Public Policy, Duke University, Durham, NC, USA.

Porras, I. 2008. Forests, flows and markets for watershed environmental services: evidence from Costa Rica and Panama. School of Civil Engineering and Geosciences. University Of Newcastle Upon Tyne, UK.

Porras, I. 2010. Fair and green? The social impacts of payments for environmental services in Costa Rica. IIED, London, UK.

Porras, I., 2010b. ‚The case of La Esperanza Hydroelectric Power Company, Costa Rica,. In: TEEB (ed.) PES As a Strategy to Minimize Risk, The Economics of Ecosystems and Biodiversity (TTEB), Geneva.

Porras, I., Grieg-Gran, M. and Neves, N. 2008. All that glitters: a review of payments for watershed services in developing countries. IIED, London, UK.

Porras, I., Chacón-Cascante, A., Robalino, J. andOosterhuis, F., 2011. PES and other economic beasts: assessing PES within a policy mix in conservation, 9th International Conference of the European Society for Ecological Economics Istanbul.

Porras, I., Miranda, M. and Barton, D.N., Chacón-Cascante, A. 2012. De Rio a Rio+: Lecciones de 20 años de experiencia en servicios ambientales en Costa Rica. Shaping Sustainable Markets. IIED, London, UK.

Postel, S. and Thompson, B.H.J. 2005. Watershed protection: capturing the benefits of nature's water supply services. Natural Resources Forum 29(2):98–108.

Prakash Fernandez, A. 2003. Watershed management: are loans more effective in promoting participation and ownership than contribution? Rural management Systems Series 36, Myrada, Bangalore.

RedLAC 2010. Environmental funds and payments for ecosystem services. RedLAC Capacity Building Project for Environmental Funds.

Reis, N., Sydness, G. and Barton, D. 2007. Feasibility of payments for watershed services in India and Asia. Norwegian Institute for Water Research, Oslo.

Robalino, J., Pffaf, A., Sánchez-Azofeifa, A., Alpízar, F., León, C. and Rodríguez, C.M. 2008. Deforestation impacts of environmental services payments. Discussion Paper Series. Environment for Development; Resources for the Future, Washington DC.

Robertson, N. and Wunder, S. 2005. Fresh tracks in the forest: assessing incipient payments for environmental services in Bolivia. CIFOR, Bogor, Indonesia.

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

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

Ross, M., Depro, B. and Pattanayak, S. 2006. Assessing the economy-wide effects of the PSA Program. Paper presented at the workshop on Costa Rica's experience with payments for environmental services, San José, 25–26 September.

Saberwal, V.K. 1998. Science and the desiccationist discourse of the 20th Century. Environment and History 4(3): 309–43.

Salas, J. 2004. Case study of the Maasin Watershed: analyzing the role of institutions in a watershed-use conflict, developing mechanisms for rewarding the upland poor in Asia for environmental services they provide. RUPES, World Agroforestry Centre, Nairobi.

Sánchez-Azofeifa, A., Pfaff, A., Robalino, J. and Boomhower, J. 2007. Costa Rica's Payment for Environmental Services Program: intention, implementation, and impact. Conservation Biology 21, 1165–173.

Sen, A. 1996. 'The political economy of targeting'. In: D. van de Walle and K. Nead (eds.) Public spending and the poor: theory and evidence. The John Hopkins University Press, Baltimore, USA.

Shrestha, R. and Timilsnia, G. 2002. The additionality criterion for identifying Clean Development Mechanism projects under the Kyoto Protocol. Energy Policy 30(1): 73–79.

Sierra, R. and Russman, E. 2006. On the efficiency of environmental service payments: a forest conservation assessment in the Osa Peninsula, Costa Rica. Ecological Economics 59(1):131–41.

Skjeggedal, T., Gundersen, V., Harvold, K.A. and Vistad, O.I. 2010. Frivillig vern av skog-evaluering av arbeidsformen (Voluntary conservation of forest-evaluation working procedure), Samarbeidsrapport. NIBR/NINA.

Skutsch, M.M., van Laake, P.E., Zahabu, E.M., Karky, B.S and Phartiyal, P. 2009. 'Community monitoring in REDD+'. In: Angelsen, A.w.B., M., Kanninen, M., Sills, E., Sunderlin, W. D. and Wertz-Kanounnikoff, S. (eds) Realising REDD+: National strategy and policy options. CIFOR, Bogor, Indonesia, pp. 101–362.

Southgate, D. and Wunder, S. 2007. Paying for watershed services in Latin America: a review of current initiatives. Working Paper No. 07-07. Sustainable Agriculture and Natural Resource Management Collaborative Research Support

Program (SANREM CRSP) and Office of International Research, Education, and Development (OIRED), Virginia Tech, Blackburg, VA.

Spangenberg, J.H., Settele, J. 2010. Precisely incorrect? Monetising the value of ecosystem services. Ecological Complexity 7(3): 327–37.

Stanton, T., Echavarria, M., Hamilton, K. and Ott, C. 2010. State of watershed payments: an emerging marketplace. Ecosystem Marketplace, Washington, DC.

Stem, C. 2005. TNC and partner experiences with watershed valuation activities in the State of Chiapas, Mexico. Final report. The Nature Conservancy, USAID, USA.

Sun, C. and Chen, L. 2006. A study of policies and legislation affecting payments for watershed services in China. Developing markets for watershed protection services and improved livelihoods. Research Center of Ecological and Environmental Economics Beijing, and IIED, London.

Suyanto, S. 2010. Site Profile: RUPES Sumberjaya, RUPES Program. ICRAF, Bogor, Indonesia.

Svarstad, H., Sletten, A., Paloniemi, R., Barton, D. and Grieg-Gran, M. 2011. 'Three types of environmental justice: from concepts to empirical studies of social impacts of policy instruments for conservation of biodiversity'. In: Tingstad, K.M.K. (ed.) Assessing the Role of Economic Instruments in Policy Mixes for Biodiversity Conservation and Ecosystem Services Provision. Report for the Policymix project, prepared by the Norwegian Institute for Nature Research (NINA).

Swallow, B., Kallesoe, M., Iftikhar, U., Van Noordwijk, M., Bracer, C., Scherr, S., Raju, K.V., Poats, S., Kumar Duraiappah, A., Ochieng, B., Mallee, H. and Rumley, R. 2009. Compensation and rewards for environmental services in the developing world: framing pantropical analysis and comparison. Ecology and Society 14(2):26.

Swallow, B., Leimona, B., Yatich, T., Velarde, S. and Puttaswamaiah, S. 2007. The conditions for effective mechanisms of compensation and rewards for environmental services. CES Scoping Study Issue Paper no.3. World Agroforestry Centre, Nairobi, Kenya.

TEEB (The Economics of Ecosystems and Biodiversity) 2011. 'Rewarding benefits through payments and markets', Chapter 5. In: ten Brink, P. (ed.) TEEB in National and International Policy Making. Earthscan, London.

Tikka, P. 2003. Conservation contracts in habitat protection in southern Finland. Environmental Science & Policy 6, 271–78.

Turpie, J.K., Marais, C. and Blignaut, J.N. 2008. The working for water programme: evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecological Economics 65(4): 788–98.

van Eijk, P., Kumar, R., 2009. Bio-rights in theory and practice: a financing mechanism for linking poverty alleviation and environmental conservation. Wetlands International, Wageningen, The Netherlands, p. 134.

Van Noordwijk, M., 2005. RUPES typology of environmental service worthy of reward In: Suyanto, S., Leimona, B., Permana, R.P. and Chandler, F. (eds.) Developing Mechanisms for Rewarding the Upland Poor in Asia for Environmental Services They Provide, World Agroforestry Centre, Bogor, Indonesia.

Veiga, F. and Gavaldão, M. 2011. Iniciativas de PSA de Conservação dos Recursos Hídricos na Mata Atlântica'. In: Guedes, I., Becker, F. (eds) Pagamento por Serviços Ambientais na Mata Atlântica. Lições aprendidas e desafios. Ministério do Meio Ambiente, Brasilia.

Villanueva, C., Ibrahim, M., Torres, K. and Torres, M. 2008. 'Planificación agroecológica de fincas ganaderas: La experiencia de la subcuenca Copán, Honduras'. In: Desarrollo, D.d.l.y. (ed.) Serie Técnica. Centro Agronómico Tropical de Investigación y Enseñanza (CATIE), Turrialba.

Waage, S., Steward, E., 2007. The new markets for environmental services: A corporate manager's guide to trading in air, climate, water and biodiversity assets. December, Business for Social Responsibility, New York.

Wätzold, F., Mewes, M., van Apeldoorn, R., Varjopuro, R., Chmielewski, T., Veeneklaas, F., Kosola and M.-L. 2010. Cost-effectiveness of managing Natura 2000 sites: an exploratory study for Finland, Germany, the Netherlands and Poland. Biodiversity and Conservation 19(7):2053–069.

White, D. and Minang, P. 2011. Estimating the opportunity costs of REDD+: a training manual. Version 1.3. The World Bank, CGIAR, Carbon Finance Assist, Forest Carbon Partnership, ASB.

Wunder, S. 2005. Payments for environmental services: some nuts and bolts. CIFOR Occasional Paper. CIFOR, Indonesia.

Wunder, S. 2008. Payments for environmental services and the poor: concepts and preliminary evidence. Environment and Development Economics 13(3):279–97.

Wunder, S. and Albán, M. 2008. Decentralized payments for environmental services: the cases of Pimampiro and PROFAFOR in Ecuador. Ecological Economics 65(4):685–98.

Wunder, S., Engel, S. and Pagiola, S. 2008. Taking stock: a comparative analysis of payments for environmental services programs in developed and developing countries. Ecological Economics 65(4): 834–52.

Wunder, S., Vargas, M.T. 2005. 'Beyond "markets": why terminology matters'. Editorial in The Ecosystem Marketplace, 22 March.The Katoomba Group.

Wünscher, T., Engel, S., Wunder, S. 2008. Spatial targeting of payments for environmental services: a tool for boosting conservation benefits. Ecological Economics 65(4):822–33.

Xu, J., Tao, R., Xu, Z. and Bennett, M.T. 2010. China's Sloping Land Conversion Program: does expansion equal success? Land Economics 86(2):219–44.

Xu, Z., Bennett, M., Tao, R. and Xu, J. 2004. China's Sloping Land Conversion Program four years on: current situation, pending issues. Special Issue: Forestry in China – Policy, Consumption and Production in Forestry's Newest Superpower. The International Forestry Review 6, 317–26.

Annexes

Annex 1: Monitoring and impacts in existing PWS schemes

Case	Extent	Design and monitoring	Reported results
Australia, Hunter River Salinity Trading, New South Wales	Local output-based scheme (NSW, 2009)	Point source dischargers trade credits in order to maintain a maximum salinity level of 900 EC (electrical conductivity) units. The scheme monitors and caps the discharges of 23 coalmining and power generation facilities, which hold a total of 1,000 salinity credits. During high flow conditions, each credit allows a discharger to release up to 0.1% of the Total Allowable Discharge or sell the credit to another participating facility through the scheme's online trading platform. No discharges are allowed during low flow conditions, and discharges are unrestricted during flood conditions.	The service area covers a total of 2.2 million hectares. The scheme achieved excellent results in 2008–2009. Salinity goals in the Upper Sector of the Hunter River were met at all times when discharges were allowed under the scheme.
Bolivia, Los Negros (Asquith *et al.*, 2008a; Asquith and Vargas, 2007; Le Tellier *et al.*, 2009)	Local input-based initiative	Fundacion Natura implements an on-site monitoring programme for participating farmlands, which is conducted every 12 months by a Project Control Team. The team is composed of one upstream and one downstream community member, a Fundacion Natura field technician and the land owner. Monitoring reports are submitted to the enforcing agency that imposes final sanctions. Due to the lack of a hydrological baseline and sufficient monitoring data, the Fundacion Natura takes bi-weekly quantitative hydrological measurements in the watershed's tributaries.	The scheme's additionality has been questioned as the land concerned is not at highest risk of deforestation, and has not been rigorously measured. A hydrological study found no relationship between forest cover and streamflow.

Annex 1: continued

Case	Extent	Design and monitoring	Reported results
China (Bennett and Xu, 2008; Sun and Chen, 2006; Xu *et al*., 2010; Xu *et al*., 2004). Forest Ecological Compensation Sloping Land Conversion Programme (SLCP) Conversion of Cropland to Forest Programme (CCFP)	National input-based schemes	National government programme in which farmers must set aside erosion-prone farmland in critical areas of the watersheds of China's main rivers. Monitoring is conducted at the local level by local governments.	The reported figures on plantations are staggering – a total plantation area of 53 million hectares, with 28 million ha of plantations created in 6 years, 8.8 million ha of crops converted to plantations, and soil erosion reduced by 4.1 million ha – but some studies on the SLCP suggest low survival rates and insufficient monitoring, indicating that the programme may slip into a one-off transfer to participants, with few substantive environmental outcomes. There is no information on its impact on water flows.
Colombia, Cauca Valley (Echavarria *et al*., 2003; Kosoy *et al*., 2005)	Local input-based schemes	Land use is monitored by water associations.	Land use is continuously monitored, but no studies have been made of water flows. Anecdotal evidence suggests that Desbaratado River levels between 1988 and 1998 did not match previous extreme flooding incidents, and that there were improvements in dry-season flows for the Nima and Amaime watersheds. However, these data are limited, and the absence of concrete figures makes it difficult to assess the actual hydrological impacts of project interventions (Echavarria 2004). According to Kosoy, *et al.* (2005), long-term trends for annual rainfalls and irregular fluctuations are due to large-scale geographical causes rather than the status of local forests.
Costa Rica, La Esperanza (Rojas and Aylward, 2003b; Porras, 2010b)	Local input-based scheme	Baseline established using extrapolated data from other local watersheds. No monitoring processes have been implemented, although changes in water quantity are estimated according to the amount of hydro-electricity produced.	Land where payments are made is protected by a private reserve. There have been no change in land use, and no changes in water flows are expected.

Annex 1: continued

Case	Extent	Design and monitoring	Reported results
Costa Rica, PSA Programme (Arriagada *et al.*, 2010; Blackman and Woodward, 2010; Porras, 2010; Sánchez-Azofeifa *et al.*, 2007)	National input-based programme	Hydrological sensitivity priority criteria for contract allocation (1 of 7 criteria) and higher payments per hectare (US$400/ha/5 years compared with US$320/ha/5 years for regular protection). Monitoring includes observation of potential changes to land use through a combination of visual inspections and satellite pictures. Non-compliance is punished by the withdrawal of payments, and in extreme cases through civil legal action. Payments are demand-led, and mostly for conservation. Compliance on forest cover has been observed, and effectiveness is increasing as the programme moves from a first-come-first-served to a targeted approach. However, there are no studies of its impacts on watershed services.	By February 2010 the programme covered nearly 730,000 hectares of forest, and had planted nearly 3 million trees in agroforestry plantations. Lack of targeting has resulted in modest to minimum impact on deforestation. Studies of cloud forests show modest impact on water budgets.
Ecuador, Cuenca (Crespo *et al.*, 2009; Echavarria *et al.*, 2004; Stanton *et al.*, 2010)	Local input-based initiative	The Macua Project was started in 2002 by the University of Cuenca, Ecuador, to compile land and hydrological data on the four watersheds that feed the town of Cuenca. Project activities included installing network-linked meteorological monitoring equipment, assessing the demand (designated) and supply (availability) of water in the watersheds, and conducting water quality and soil studies. Data collected for this project provide the ETAPA (Cuenca's municipal water utility) with well-established baseline data and the ability to continuously monitor implementation of future PWS programmes in the Yanuncay watershed.	Hydrological studies of the wet Andean páramo ecosystem in nearby areas show links between land use and water: 1) pine plantations reduce annual water yield due to increased evapotranspiration; 2) livestock grazing does not seem to affect the hydrological response owing to low herd densities; 3) cultivation mainly affects regulation by increasing the magnitude of peak flows and reducing base flows (Crespo *et al.*, 2009).
Ecuador, Fonag (Cannon *et al.*, 2010; Echavarria *et al.*, 2004; Southgate and Wunder, 2007)	Local input-based initiative	This programme does not monitor or evaluate conditionality. Payments from users of watershed services are directed to conventional conservation projects (Southgate and Wunder, 2007). So far FONAG has managed fairly sophisticated weather stations and soil moisture monitoring stations that have been installed and are working properly (Cannon *et al.*, 2010).	The programme was managing over 65,000 ha by 2008; however, the plantations are still too young to estimate their impact. Understanding of the water budget in the watersheds concerned is very basic, and almost nothing is known about how they might be affected by changes in land-use activities. The effects are likely to be small because (i) threshold levels are very low - about 2.5% of all areas are reforested; and (ii) farmers living upstream may use the additional flows for their own purposes (Cannon *et al.*, 2010).

Annex 1: continued

Case	Extent	Design and monitoring	Reported results
France, Vittel (Perrot-Maitre, 2006; Wunder, 2008)	Private output- and input-based local scheme	Water quality is monitored extensively (300 tests per day) due to the strict regulations covering products marketed as natural spring water. In addition to in-house monitoring, the Institut National de la Recherche Agronomique (INRA) conducts compliance monitoring at 17 participating farm sites, and INRA and Agrivair (the intermediary developed by Vittel to negotiate and implement the PWS scheme) monitor compliance of farming practices on participating land.	The monitoring practices that have been incorporated into the Vittel scheme have generated clear and measurable improvements in water quality.
Honduras, Jesus de Otoro (Kosoy *et al.*, 2005)	Local input-based scheme supported by PASOLAC	The water authority has been monitoring water quality since 1999. Base studies included basic hydrological information, topography, precipitation, soil types, etc., but no direct measurement of sediments.	Monitoring shows some 'improvement' since the introduction of PWS. However, the small scale of this pilot PWS scheme and lack of knowledge about underground water movements mean that it is unlikely to be the main reason for this improvement, which can also be linked to the chlorine used to treat water. Fieldwork for an independent study (Jesus de Otoro, Puerto Barrios and San Pedro Norte) showed that none of the case studies can confidently demonstrate that the environmental service is provided, or identify the threshold areas required to reach specific quality targets.
Honduras, Campamento (Ardón and Barrantes, 2003; Kosoy *et al.*, 2005)	input-based local initiative (no cash payments made yet)	No baseline studies have been conducted. Changes in land use are monitored, but changes in water quality and quantity are not.	A perceived reduction in pollution from coffee-processing wastes was reported in 2003. However, it was suggested that this had more to do with falling coffee prices and levels of activity, and that pollution would increase again as coffee prices recovered.
Honduras, El Copan (Alpizar *et al.*, 2007; Villanueva *et al.*, 2008)	Local input-based scheme (proposed)	Highly detailed farm-level management plans at have been designed to improve productivity, with payments based on existing land use and required activities. Work is targeted in areas with important users downstream (high-benefit) and where drainage has maximum impact (high-risk). A baseline has been developed for current land uses, which are given points according to their perceived benefits (primary forest with surveillance ranks the highest).	It is too early to assess the results of this new initiative, which builds on lessons from previous studies (especially Silvopastoril).

Annex 1: continued

Case	Extent	Design and monitoring	Reported results
Indonesia, Sumberjaya (Harto Widodo *et al.*, 2006; Suyanto, 2010)	Local input-based scheme supported by RUPES; also testing output-based measures (payments for reduced sediments). Builds on new law.	The scheme, which is supported by RUPES and builds on the new law, is testing output-based measures (payments for reduced sediments). The socio-economic baseline is established through rapid rural appraisals. Monitoring covers infrastructure, institutional aspects and actual sediment reduction in the river. Community groups learn how to capture and use sediments. A financial reward scheme provides some funds upfront and then pays additional specified amounts based on the effects achieved.	About 70% of the protection forest is covered by agreements (work on the remaining forest is in progress; only 7% was covered in 2004). Practices like multi-storey coffee gardens are being implemented, and conditional land tenure is in place for 6,400 farmers.
Mexico, Chiapas (including the El Triunfo and La Encrucijada Biosphere Reserves) (Stem, 2005)	Local input-based initiative linked to the national PSAH Programme	Monitoring is input-oriented. It has been difficult to establish a link between output indicators and changes in higher-level watershed conservation targets.	Evidence is mostly anecdotal. It supports the positive perception of restoration, suggesting reduced sediment accumulation and lower heat levels, and positive effects on shrimp catches and the local fishing industry.
Mexico, Fidecoagua, Coatepec (Blanco and Rojo, 2005; Holwerda *et al.*, 2010; Muñoz-Villers *et al.*, 2010)	Local input-based initiative linked to the national PSAH Programme	An initial baseline study to establish forest cover in properties was complemented by a literature review, but water quantity and quality have not been measured. PES is in place, along with direct purchase of plots for conservation. GIS images are used to monitor forest cover, also field visits and field supervision. The impact of cloud forests on water flows have been measured by Independent studies.	An increasing amount of land is covered by community payments (over 2,300 ha between 2003 and 2010). There is a general perception of positive impacts on water quality due to less sediment, but no hard information on this. Hydrological studies conducted between 2002 and 2007 show that the impact on flows is less than expected; also that the impact on secondary forest is similar to primary forests in terms of water at cloud levels.
Mexico, PSAH Programme (Alix-Garcia *et al.*, 2010; Muñoz-Piña *et al.*, 2008; Muñoz-Piña *et al.*, 2011)	National input-based programme	Higher payments for cloud forests and forest at high risk of deforestation. Monitoring is based on changes in forest cover (GIS and satellite images).	Only 20% of the 26 priority criteria are concerned with significant hydrological factors and risk of deforestation. Secondary criteria (administrative, social, etc.) divert funds from areas where they could be used more effectively. The programme has resulted in a small but significant reduction in deforestation, but slippage effects can only be accounted for at the national level.

Annex 1: continued

Case	Extent	Design and monitoring	Reported results
Mexico, Zapaliname (Canales, 2006; Lechuga, 2009)	Local input-based initiative linked to the national PSAH Programme (protecting existing reserves)	No baseline study was conducted. Monitoring is done as part of the reserve's regular activities.	Until 2009 only 14%of water users paid the voluntary fee. There is some monitoring of spring water quality, but the results are not available yet.
Mexico, Procuenca, (Valle del Bravo) (Manson, 2008; Porras *et al.*, 2008)	Local input-based initiative linked to the national PSAH Programme	Baseline studies have been conducted on the quantity and quality of water in rivers and dams. The scheme provides equipment and training for local stakeholder-led hydrological assessments and monitoring.	Results indicate that water quality in forested areas receiving payments is good, but that it declines in lower areas of the watershed, suggesting potential problems with fish farms and domestic or industrial wastewater.
Nicaragua, Esteli (Espinoza, undated; FAO, 2010)	Local input-based scheme supported by PASOLAC, which uses in-kind downstream payments in the form of labour.	Patchy hydrological baseline study provides average monthly precipitation and a rough description of the catchment area. No attempt has been made to differentiate seasonality from land use, and there are no plans for monitoring.	The project has not been able to start collecting fees due to the lack of a legal framework. The use of municipal ordenanza has not been enough to ensure that users make the requisite payments. The funds collected by 2010 were barely enough to cover the cost of their collection (FAO, 2010).
Nicaragua, San Pedro del Norte (Obando, 2007)	Local input-based scheme supported by PASOLAC (includes building dykes and water retention units).	Basic feasibility studies identify the main hydrological variables (precipitation, soils, etc.), but no in-depth hydrological study has been made of the area. The local municipality has some records of water before the initiative took place.	Reports indicate that water sources improved after implementation of best management plans began in 2004, with seasonal water sources becoming permanent and significant increases in total water flows. Some of this extra water is used on farms, and greater availability downstream has helped increase water supply from 14% to 32% of total demand. The impact of the scheme may increase if there is sufficient funding to include more farmers, as it currently pays only 5 of the 43 farmers in the catchment area.

Annex 1: continued

Case	Extent	Design and monitoring	Reported results
South Africa, Working for Water programme (DWAF, 2006; Le Maitre *et al.*, 2000; Swallow *et al.*, 2009; Swallow *et al.*, 2007; Turpie *et al.*, 2008)	National input-based scheme focusing on the removal of invasive alien plants that consume large quantities of water (number of trees).	The monitoring programme is based on extensive studies of water consumption by non-native or invasive plant species. The baseline developed for the Working for Water programme relies on reduced streamflows associated with vegetation type and distribution. GIS-linked monitoring of the removal of invasive species (10,000 km²) is used to examine additionality.	Intensive modelling used to estimate the impacts of this programme on water flow puts it at 48–56 million cubic metres of additional water per annum.
The Philippines, Maasin (Salas, 2004)	Input-based programme	This long-term, government-led watershed management programme focuses on social forestry as a means of improving watershed management. It involves training and management for large-scale tree planting. Farmers were required to move from some parts of the watershed.	Severe reductions in water volume and significant siltation were reported following reforestation activities with fast-growing exotic species (mainly mahogany and gmelina). There were claims that reduced flows were caused by higher water outtakes by a local utility, and increased silt due to tilling by relocated farmers, but there are no studies supporting either side, and 'the arguments ended when the rains began' (Salas, 2004).
United States, Conservation Reserve Program (CRP)	National input-based scheme (Claassen *et al.*, 2008)	This programme has developed and implemented a highly effective targeting process. Eligibility is dependent on land and soil conditions and the resulting contribution to ecosystem improvement. Assessment of these conditions is dependent on the CRP's Environmental Benefits Index (EBI), which assesses various environmental concerns relative to the cost of enrolling land in the programme. The CRP's target-based effectiveness is increased through a reverse auction process in which service providers bid against others to participate in the programme. Additionality is determined by comparing programme-specific changes in land management with changes that it is assumed would occur without the programme.	

Annex 1: continued

Case	Extent	Design and monitoring	Reported results
United States, Castskills, New York	Local output-based municipal scheme (Appleton, 2002)	Combines land easements and acquisitions with tailored pollution control measures for each farm, to maximise effectiveness and minimise costs and ensure benefits for farmers.	Benefits often take the form of time and ease of labour (manure disposal, etc.) rather than cash. Some 93% of all farms in the NY City watershed were participating in the programme within 5 years of its creation, and it is hailed as the most successful non-point pollution control in the United States.

Annex 2: Definition of terms used in the monitoring and evaluation of payments for watershed services

Additionality	Environmental additionality is the net positive impact on the provision of ecosystem services compared with the baseline scenario or a hypothetical situation where no scheme is in place (Pascual *et al.*, 2009). In other words, it is the change in land use generated by the payment, which can be compared with what would have happened without the programme ('business as usual', see Wunder *et al.*, 2008). Additionality can be measured in terms of contracts (input-based) and/or ecosystem services (output-based).
Compliance	This refers to the degree to which recipients of payments for environmental services (PES) comply with their contracts. It requires appropriate monitoring.
Cost-effectiveness	The costs and outcomes of interventions are compared to assess the extent to which they can be regarded as providing value for money, and to inform decision-makers who have to determine where to allocate limited resources. Conservation policies are considered more cost-effective if they produce higher conservation outcomes for the same total cost as other policies, or equal outcomes at less cost than other policies (Wätzold *et al.*, 2010).
Effectiveness	Environmental effectiveness refers to the degree to which a policy (in this case, PWS) outperforms alternative policies (such as national parks) in achieving specific environmental goals. The way that effectiveness is measured depends on how the outcomes are defined; whether it is in terms of a specific land use such as forest cover, or the level of environmental service.
Efficiency	Efficiency is the difference between the gross welfare effects of a scheme on the target population and the total costs incurred (Pascual *et al.*, 2009). In PES, it is determined by the extent to which incremental ecosystem services are provided and the opportunity, implementation and transaction costs of their provision (Wunder *et al.*, 2008). Transaction costs can be lowered by using flat payments and light monitoring, but this may affect the effectiveness of a scheme. Efficiency can be measured in different ways, such as looking at the amount paid for each hectare at risk of deforestation and the number of hectares of land at risk (see, for example, Alix-Garcia *et al.*, 2005a).
Input-based, output-based	Input-based schemes work on the assumption that a given land-based activity will deliver environmental services. Output-based schemes try to measure the actual environmental services provided.
Leakage (spillage)	Leakage refers to the displacement of the environmentally damaging land uses that the PES programme aims to replace (Wunder *et al.*, 2008).
Permanance	If the desired changes in land use or level of ecosystem services occur on a long-term basis they are deemed to be 'permanent'.
Targeting	In PWS targeting refers to a process where the programme administrator moves beyond the self-selecting nature of voluntary participation in PES programmes and areas that add relatively little to the provision of environmental services, to areas previously determined as more important. It mainly focuses on (i) environmental benefits (gap analyses to identify high-benefit priority areas for biodiversity conservation); (ii) programme costs (a negative correlation between costs and biodiversity can lead to low-cost/high-benefit solutions) or (iii) benefit-to-cost ratios (Wünscher *et al.*, 2008).

Monitoring payments for watershed services schemes in developing countries

Payments for watershed services (PWS) are schemes that use funds from water users (including governments) as an incentive for landholders to improve their land management practices. They are increasingly seen as a viable policy alternative to watershed management issues, and a means of addressing chronic problems such as declining water flows, deteriorating water quality and flooding. In some places, local governments, donor agencies and NGOs are actively trying to upscale and replicate PWS schemes across the area. While their apparent success and progress in launching new initiatives is encouraging, there is still much to be learned from formative experiences in this field, especially with regard to monitoring and evaluation.

In this paper we discuss the monitoring and evaluation criteria behind compliance or transactional monitoring, which ensures that contracts are followed, and effectiveness conditionality, which looks at how schemes manage to achieve their environmental objectives regardless of the degree of compliance. Although the two are usually linked, a high degree of compliance does not necessarily ensure that a scheme is effective. This is because a poorly designed scheme may target the wrong land managers and land that is at least risk, meaning that payments do not generate the desired hydro-ecological or conservation benefits. As the levering capacity to demand payments for better watershed management increases, so does the need to understand the dynamics of such activities and demonstrate their impacts. While the growing interest in such schemes shows that participants believe in the principle of land management, evidence of their impact is needed to determine which initiatives genuinely add value and are worth pursuing.

The International Institute for Environment and Development (IIED) is a policy and action research organisation working to promote sustainable development–development that improves livelihoods in ways that protect the environments on which these are built. Based in London and working on five continents, we specialise in linking local priorities to global challenges. In Africa, Asia, Latin America, the Middle East and the Pacific, we work with some of the world's most vulnerable people to ensure they have a say in the decision-making arenas that most directly affect them – from village councils to international conventions.

This paper was produced by IIED's Sustainable Markets Group. The Sustainable Markets Group drives IIED's efforts to ensure that markets contribute to positive social, environmental and economic outcomes. The group brings together IIED's work on market governance, business models, market failure, consumption, investment and the economics of climate change.

This research was funded by UK aid from the UK Government, however the views expressed do not necessarily reflect the views of the UK Government.

ISBN 978-1-84369-903-3

iied International Institute for Environment and Development

80-86 Gray's Inn Road, London WC1X 8NH, UK

Tel: +44 (0)20 3463 7399
Fax: +44 (0)20 3514 9055
email: info@iied.org
www.iied.org