Monitoring and evaluation of Payment for Watershed Service Schemes in developing countries

Ina Porras[(a)], Jeff Dengel[(b)] and Bruce Aylward[(b)]

Final draft August 2012

To be presented at the 14th Annual BioEcon Conference on “Resource Economics, Biodiversity Conservation and Development”. 18-20 September 2012. Kings College, Cambridge. United Kingdom

[(a)] International Institute for Environment and Development, UK. [(b)] Ecosystem Economics LLC, Oregon USA

Useful definitions in PWS monitoring and evaluation

Additionality	Environmental additionality is the net positive impact in the provision of ecosystem services, in comparison with the baseline scenario or hypothetical situation where the scheme is not in place (Pascual et al. 2009). In other words, it is a change in land use achieved by the Payment compared to what would have happened without the program ('business-as-usual') (Wunder, Engel, and Pagiola 2008).
Effectiveness	Environmental effectiveness refers to degree on which a policy (i.e. PWS) achieves specific environmental goals better than alternative policies (i.e. national park). Its measurement depends on the definition of the outcomes, for example, whether it is a specific land use (i.e. forest cover), or the level of environmental service.
Efficiency	Efficiency effect is the difference between the gross welfare effects induced by the scheme on the target population and the total cost incurred (Pascual et al. 2009). In PES it is determined by the extent to which incremental ecosystem services are provided, but also by the cost (opportunity, implementation, transaction) at which this was achieved (Wunder, Engel, and Pagiola 2008). Efficiency can be measured in different ways. For example the dollar amount paid for each hectare at risk of deforestation; and the number of hectares of land at risk for a given budget, in a study in Mexico (Alix-Garcia et al. 2005). Low transaction costs may be obtained using flat payments and low monitoring, but this may affect the effectiveness of the scheme.
Equity	Equity effect is the net impact of the implemented payments on inequality, among a reference set of participants, compared to the baseline scenario. Its calculation depends on depends on two factors. The first factor relates to the fairness criterion chosen in the design of the scheme for the distribution of the net benefits accruing from the implementation of the payments. The second factor is related to the evaluative space chosen to assess the final equity impacts caused by the PES. (Pascual et al. 2009)
Cost-effectiveness:	Compares the costs and outcomes of an intervention to assess the extent to which it can be regarded as providing value for money. This informs decision-makers who have to determine where to allocate limited resources. In conservation, a policy is considered more cost-effective than another if its conservation outcome is higher for given total costs. Alternatively, a policy is more cost-effective than another if an equal conservation outcome is attained at lower total cost (Wätzold et al. 2010).
Targeting	Targeting in PWS refers to a process where the Program administrator moves beyond the self-selecting nature of voluntary participation in PES programs away from areas that add relatively little to the provision of ES to areas previously determined as more important. Targeting can take place in three forms: (1) environmental benefits (i.e.

	gap analyses to identify high-benefit priority areas for biodiversity conservation); (2) program costs (negative correlation between costs and biodiversity can lead to low-cost/high-benefit solutions), or (iii) benefit-to-cost ratios (Wünscher, Engel, and Wunder 2008)
Compliance	Refers to the degree to which PES recipients comply with their contracts, which in turn requires appropriate monitoring)
Leakage (spillage)	Leakage results in situations where the environmentally-damaging land uses that the PES program is replacing are displaced elsewhere (Wunder, Engel, and Pagiola 2008)
Permanence	Refers to whether the desired change in either land use or level of ecosystem services are provided on a long-term basis ('permanence');

1. Introduction

Sustainable management of land and water can provide multiple environmental services, including watershed services. Sustainable management includes the conservation of existing natural ecosystems, management of agricultural and agroforestry landscapes, and restoration of degraded ecosystems. These efforts are often driven by instruments, ranging from direct regulation such as prohibitions and zoning, 'soft' approaches such as information and capacity building, market-based instruments such as taxes and fines, and hybrid approaches using regulatory authority and market mechanisms such as cap-and-trade approaches.

The use of payment systems to promote sound ecosystem and watershed management is an idea that moved from economic theory into the policy debate, and finally into practice over the last 30 or so years. In watersheds, economic incentives are used to promote upstream land management expected to result in the protection or improvement of downstream water quantity and quality. As with "putting a price" on carbon through carbon taxes or carbon cap and trade schemes, the intent of these payment for watershed service (PWS) schemes is to internalize the externalities of land use into the land managers production process, so as to achieve greater economic efficiency[1].

The objective of this paper is to take stock of our understanding regarding the monitoring and evaluation of PWS schemes. In this paper, we argue that the question of PWS environmental performance is best assessed with reference to the conditionality and additionality of such schemes.

PWS schemes are increasingly looked to and looked upon as a viable policy alternative to watershed management problems. In some places, local governments, donor agencies and NGOs are actively trying to upscale and replicate PWS schemes across the landscape. IIED's first review of PWS identified 41 proposed and ongoing schemes in developing countries (Landell-Mills and Porras 2002). By 2008, this figure had increased to 50 ongoing schemes and 45 proposed ones (Porras, Grieg-Gran, and Neves 2008). IIED's latest update has identified many new schemes and proposals including national schemes in Ecuador and Vietnam, and numerous local schemes particularly in Brazil, where in the Atlantic Forest and Cerrado alone there are eight schemes up and running, with 33 more in the pipeline (Veiga and Gavaldão 2011).

Yet, despite all this apparent success and progress with launching new PWS schemes there remains the suspicion that lessons have yet to be learned from the formative experiences thus far, and this is nowhere more true than with regard to monitoring and evaluation. While a handful of efforts have been made to review global progress with PWS, there has to date been no systematic, wide-spread and rigorous review of the achievements of PWS schemes. By focusing in on the monitoring and evaluation

[1] Following the Coasean paradigm, the justification for the design and implementation of PES schemes is that market failures result in the absence of markets for ecosystem services. Incentives can be used to encourage private agents to take into account the costs and benefits of the provision of ecosystem services into their own decisions about land use, and resolve potential environmental externality problems (Pascual et al., 2009). If transaction costs are how enough, and property rights are in place, individuals would trade their rights until a Pareto-efficient allocation is achieved

question it is hoped that this paper can stimulate activity in this direction, with a view towards promoting better program design and implementation in the future.

On-the-ground actions must represent a change in behaviour and practice. This may be a change from past behaviour or a change from expected future behaviour. Only if the payment secures an action that otherwise would not have happened does the payment produce a real "additional" benefit. At the heart of the monitoring problem, is then, the difficulty of knowing with any certainty what would have happened absent the PWS scheme – given that the PWS scheme is all that can be observed and measured, i.e. monitored (Ferraro and Pattanayak 2006). While some regard this as a major issue with regard to biodiversity conservation, the problem can be generalized to incentives of any kind that seek to avoid future behaviour. Inherently such actions are uncertain, but nonetheless necessary. Arguably good and consistent program monitoring and evaluation that takes account of local views and perceptions can serve to root out what actions are truly additional and which are merely self-serving enrichment at the public trough.

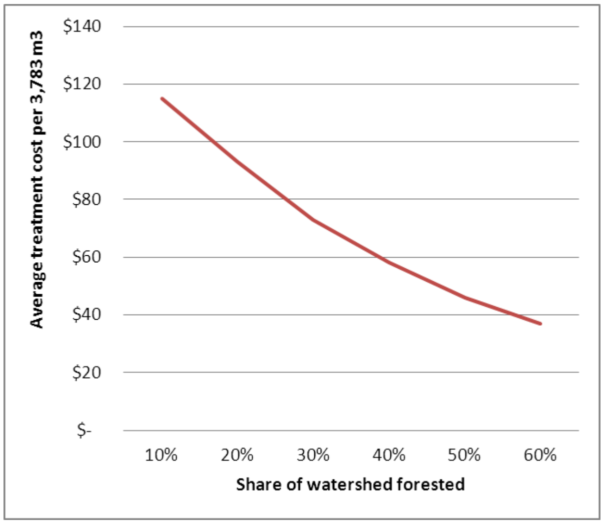
2. Payments for Watershed Services: A market-development overview

2.1 The economic benefits of watershed protection

Apart from spiritual and non-use values, the keen interest in PWS schemes is their direct link to economic impacts and how they affect water filtration and purification; seasonal flow regulation; erosion and sediment control; and habitat preservation.

The capacity for regulation of relatively intact soils is highlighted by scientists (Bruijnzeel 2004; Calder 2005). The proportion of natural forests is often seen as an indicator of how healthy the watershed is (Kaimowitz 2004). For example, an economic analysis of the average treatment costs in relation of forest cover in the USA show a clear inverse relation (Figure 1)., reducing the need to build or upgrade treatment facilities (Postel and Thompson 2005). Other ecosystems, like wetlands and páramos naturally absorb, clean and release water flows to provide major cities in South America with little need for purification apart from chlorine (Buytaert et al. 2006; Crespo et al. 2009). Highly degraded slopes increase siltation and turbidity. Compaction of soils and poor cover reduces the soil's ability to infiltrate water, increasing losses through evaporation and the risk of flash floods.

Figure 1. Average treatment costs for water as proportion of watershed forested, USA



Source: (Postel and Thompson 2005). Analysis of 27 US water suppliers in the USA, based on treatment of 22 million gallons (83,270 m3) per day, the average daily production of the water suppliers surveyed.

But forest cover on itself is not necessarily the answer. Exotic tree species with high water requirements decrease water flows. In South Africa, it is estimated that, if left unchecked, alien tree species would use more than 2,720 million m3 or 16% of total registered water use (Turpie, Marais, and Blignaut 2008). Management plans at farm level have been successful in controlling non-point source pollution from agriculture in New York (Appleton 2002). A combination of mixed cropping, terracing and agro forestry are suggested as practically feasible techniques for rain-fed small scale farmers to retain soil moisture in the Upper Tana in Kenya, suggesting benefits between US$ 12 and 95 million, compared to annual costs of US$ 2 to 20 million (IFAD 2012).

2.2 A working definition

Underpinning the Payments for Environmental Services idea is the notion that ecosystem degradation takes place because the ecosystem services are not properly valued. PES schemes try to create an economic mechanism (usually to complement regulation) that will internalize the positive environmental externalities associated with the production of particular ecosystem services, including watershed protection, biodiversity, and carbon sequestration (Porras, Grieg-Gran, and Neves 2008; Kosoy et al. 2007). The literature associated with this topic frequently cites the definition of PES constructed by Wunder (2005), who defines a 'true' PES schemes as:

1. A voluntary transaction where
2. A well-defined environmental service (or land use likely to secure that service)
3. Is being "bought" by a (minimum of one) ES buyer
4. From a (minimum of one) ES provider
5. If and only if, the ES provider secures ES provision (conditionality)

Porras et al. (2008) indicate that the restrictions imposed by solely relying on Wunder's five criteria to identify PES schemes can potentially exclude 'PES-like' programmes that include, or are part of, non-market institutional arrangements; in particular, schemes funded though government taxes and other non-voluntary funding mechanisms, e.g. user fees imposed by utilities. In response to these possible exclusions, they modify Wunder's definition by identifying a PES scheme as:

- a transaction in which a supplier or seller of the ecosystem service is responding to the offer of compensation from a single or multiple beneficiaries (NGO, private party, local or central government entity) and/or a beneficiary separate from the seller which is not a central government entity,
- compensation is conditional upon the land management practices specified by the program, and
- the voluntary component is only attached to the supply-side of the transaction in that the provider 'voluntarily' enters in to the contract.

In relation to the level of ecosystem service compensation, PES will ideally, in theory at least, provide suppliers with payments equivalent to the opportunity cost associated with previous or potential alternative land uses outside of those specified within the PES contract. In addition, the compensation should not to exceed the value attributed to the environmental externality (Kosoy et al. 2007; Asquith and Wunder 2008).

Like PES, payments for watershed services (PWS) is an economic conservation mechanism encouraging land owners to adopt land management practices intended to influence biophysical attributes of an ecosystem that are expected to affect the provision of the provision of freshwater supplies[2]. The hydrological processes (i.e. interception, infiltration, storm runoff, and evapotranspiration) are heavily influenced by vegetation distribution and composition, and soil conditions, which in turn effect the provision of the watershed services. As such, existing schemes concentrate on land-use practices that include conserving and securing existing ecosystems, reforestation, afforestation, improved land management and ecosystem interaction practices, and restoration of degraded ecosystems (Porras, Grieg-Gran, and Neves 2008)

Stretching the complementarity of carbon and water?

In the Piura region in Peru a pilot group of UK-based Cafédirect's AdapCC project and Peru-based Cepicafe—one of Cafédirect's suppliers— is combining the carbon market, protection against flash floods, and sustainability of local farming. Farmers reforest degraded grasslands at higher elevations using pine trees (70%) and native quinal species (30%). Under CarbonFix standard, it is expected that 10% of carbon credit sales from saplings will go to Cepicafe. Until now, Cafédirect is pre-paying funds to purchase credits in order to help the project get up and running.

The total at the moment is 224 hectares but the aim is to cover 5,000 hectares ((Lee 2012) in The Ecologist) . The project has so far been widely applauded, yet some questions arise regarding the nature of the upscaling regarding the wider ecosystem, especially if pine remains the main specie used. Piura is located in the northern coastal desert of Peru, with very low average precipitation during the year and marked rainfalls

[2] Specific watershed (or hydrological) services addressed in current schemes include: sediment reduction, flood abatement, reduced peak flows, base flow maintenance, erosion control, fisheries, and improved water quality (through nutrient, chemical, and salinity control) – Porras et al. (2008).

during the rainy season, and exposure to weather extremes. Hydrologist have for a long time warned against the high water requirements of fast growing forest species in large plantations, which may result in significant reduction of water flows downstream (Calder and Aylward 2006)., a particular problem in areas with high water scarcity like the desert of Peru.

2.3 Tracking the interest

In developing countries, PES programmes have not focused much on measuring and rewarding the delivered changes in the ecosystem level, but on different types of land practices that are expected to provide a particular environmental service (see **Table 1**). In some cases, activities supporting the provision of bundled environmental services can be complementary. For example, conservation of forests can help preserve existing water quality and quantity, protects biodiversity, and increases landscape beauty, and are 'marketed' as a bundle. In other cases, activities aiming at incentivising provision of a given ecosystem service (i.e. carbon through large-scale reforestation) can potentially harm other important ecosystem services (in this case, reduced water flows and reduce biodiversity especially in water-strained environments, and when using fast-growing, exotic species, see Calder (2005).

Table 1. Examples of land use and ecosystem services targeted (Porras et al, 2011)

Land use promoted	Ecosystem Service	Example
Conservation and protection of existing ecosystems	Bundled	• Costa Rica: payments made per hectare per year for protection, and represent most of the demand for PES in the country (Porras 2010; Sánchez-Azofeifa et al. 2007). In 2012, a set of criteria for allocating contracts for protection assigns extra points and 25% extra payment in water protection areas. • A compensation programme in Finland provides incentives for the creation of new nature reserves providing habitats of threatened species or of great natural beauty (Tikka 2003) • Voluntary forest conservation contracts in Norway (Barton 2010; Skjeggedal et al. 2010) • Sweedish Nature conservation agreements (Naturvårdsavtal), normally signed for 50 years (EEA 2010; Mayer and Tikka 2006). • Austrian Natural Forest Reserve Programme (launched in 1995) which compensates for not harvesting for a period of 20 years
Agricultural practices: aim at providing environmental services and on-site economic returns to farmer	Usually biodiversity and water	• Silvopastorial projects in Colombia, Nicaragua, and Costa Rica (Ibrahim et al. 2010; Casasola et al. 2009; Montagnini and Finney 2010) • Organic agricultura 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 for agricultural ecosystems conservation funding, co-financed by the EU'S Common Agricultural Policy (CAP), for example the High Nature Value areas promoted by the in Europe[3] (EEA 2010).
Reforestation for commercial purpose (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 in the Working for Water in South Africa • PCJ in Brazil to restore riparian forests (Porras, Grieg-Gran, and Neves 2008)

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); Wätzold et al.(2010).

Interest in PWS-type of projects escalated fast between 2002 and 2008. In 2002 IIED conducted a review of emerging markets for environmental services, and found 41 proposed and ongoing schemes in developing countries (Landell-Mills and Porras 2002). A new review in 2008 showed that, despite major setbacks in some cases, there were at least 50 ongoing schemes, and nearly as many proposals at different stages in developing countries (Porras, Grieg-Gran, and Neves 2008). Implementation of schemes has not been lineal, and many proposals are abandoned or absorbed into other projects or type of instrument (see also a recent review by Forest Trends (2010) looking at PWS and water quality trading). In Africa, for example, approximately half of proposals never make it to pilot or implementation stages (2011).

[3] In Europe, a (relatively small) part of the agricultural subsidies under the Common Agricultural Policy (CAP) can be regarded as PES for biodiversity enhancement. For example, in France the agri-environment schemes amounted to less than 4% of total CAP expenditure in 2005, and in the Netherlands to less than 2%. In particular, the so-called 'second pillar' of the CAP provides for EU co-financing of projects and measures under the Member States' 'rural development programmes'. One of the 'thematic axes' of the second pillar is 'improving the environment and the countryside'. Financial support can be given to farming practices that benefit biodiversity in 'High Nature Value' areas.

2.4 Sources of funding for PWS

Watershed payments can be provided by any number of entities, including end-users, utilities, government agencies, NGOs, user groups, or a combination of them. Generally, three sub-types of PWS schemes can be identified when assessing current PES schemes: 1) voluntary user-financed schemes in which private-sector buyers and sellers execute transactions; 2) non-voluntary utility-financed schemes in which a utility compensates watershed service providers with funds provided by user fees; 3) non-voluntary government-financed schemes in which a government entity compensates services providers with tax/tariff revenues.

As noted below these may have their funding source based either on voluntary or non-voluntary payment mechanisms. Thus, even though these payment systems are often conflated with "markets" for environmental services, it is important to realize that such payments may not always represent true consumer demand for the service (Hartwell and Aylward 2008).

- *User-financed Programs*

  Direct user-financed schemes are typically conducted within a smaller scale than those associated with government run schemes, and as a result often operate within one or a few targeted service areas or watersheds. As a result, these schemes tend to be adapted to local influences and external factors, have better monitoring and enforcement, and be less-oriented toward political objectives in comparison to government funded programs (Asquith and Wunder 2008). These programs tend to be less prevalent given the lack of excludability associated with environmental services, but it is the closest scheme type to a 'real' market definition (Kroeger and Casey 2007). These schemes are often referred to as "private deals" and as they reflect true consumer demand for the watershed service are examples – in the economist's lexicon – of Coasian Bargains (Johnson, White, and Perrot-Maitre 2001; Aylward 2007).

- *Utility-financed programs*

  Utility-financed programs acquire funding to compensate watershed service providers through allocating revenues derived from user fees or tariffs from a public utility or a regulated private utility. These utilities tend to be water supply utilities with plants and facilities located downstream of land-use operations influencing watershed function. These may also include large irrigation service districts or hydropower facilities. Though the utility and the service provider have entered into a voluntary transaction, compensation derived from the end-use beneficiaries (utility users) may be provided on a voluntary or involuntary basis. In the latter case, the users have no choice but to pay the fee, whereas in the former the user can choose to "check-off" a box and add the fee to their utility bill. Given the local government orientation of most such utilities, these programs tend to be site-specific and oriented towards the particular conditions in the source watershed.

- *Government-financed programs*

  Government-financed programs acquire funding to compensate watershed service providers through allocating revenues derived from earmarked tax revenues or general budget. While the action is taken in principle on behalf of the public and the service users, participation from the end-user standpoint is not necessarily voluntary and in many cases induced by regulations (Porras et al. 2008). Pro-poor and development-related side objectives are often found to be additional components influencing the operation and effectiveness of these PWS schemes. In addition, the focus tends to implemented on a national or regional scale, and include other environmental services, like carbon and biodiversity (Asquith & Wunder 2008). Government-led programmes also tend to include multiple-environmental objectives, for example, biodiversity and carbon sequestration. They can be, however, the most effective way to ensure significant funding and by adopting a national-level approach they can overcome transboundary and leakage issues common to local-scale schemes.

3. Conditionality and additionality

Conditionality refers to the link between the incentive and the action that is contracted under a PWS scheme. Additionality refers to the net benefits created by the PWS. The latter has two primary pathways, first how the incentive changes behaviour, and second, how and to what extent the on-the-ground actions lead to changes in watershed services. Both these criteria respond to the nature and thoughtfulness exercised at the design state, taking into account reliable scientific information, and the care with which they are implemented.

Each criterion also comes with its own form of monitoring and evaluation: compliance monitoring for conditionality and effectiveness monitoring for additionality. Compliance, or transactional, monitoring is necessary to ensure that the contracts – whether for input or output based – are lived up to by the land manager and the “buyer.” Input-based systems monitor transactional compliance, making assumptions on the impacts of predetermined land-use activities, and monitoring these changes accordingly. In simple terms, compliance monitoring seeks to ensure that the conditionality inherent in a PWS scheme is carried out, that the project achieves a high degree of efficacy in its implementation. Only then is the conditionality in place to drive additionality.

Effectiveness monitoring attempts to ascertain how well a PWS scheme has achieved its larger objectives, irrespective of the degree of compliance. Effectiveness monitoring tracks how the reward offered by a PWS scheme leads to improved watershed services and ecological outcomes. If the same ecological outcomes would have come to pass without the PWS scheme, then the scheme is not effective, there is no additionality.

Generally, it can be expected that in order to be effective a PWS scheme will demonstrate a high level of compliance. The conditionality implicit in PWS contracts is typically a necessary step in generating benefits that are additional to those that would have come to pass without the PWS scheme (or in the counterfactual case). However, a high degree of compliance does not necessarily produce a high degree of effectiveness. A poorly designed scheme, for example, may target the wrong land managers and land at least risk and, thus, the payments may not generate the desired hydro-ecological or conservation benefits (for example, the little additionality shown by the early part of the Costa Rican PES scheme (2011))

4. The evidence

Using theory and evidence available from existing schemes, this paper examines these criteria and the emerging evidence and lessons from monitoring and evaluation efforts to date. We expand on the framework presented by Wunder (2008). Although the 'market' is relatively new, some schemes have been going for several years but evidence of impact is still short on the ground (Porras, Grieg-Gran, and Neves 2008; Wunder 2005). For our analysis, we review information reported in papers and publications on existing PWS schemes (presented in the Annex).

4.1 Reported impacts from ongoing schemes

Regardless of the size of the payment, existing input-based schemes in developing countries seem to be affecting land management behaviour. It appears that contracts, if well-designed and monitored, can show substantial compliance and hence impact on land use (see Section 4.1), and for some on water quantity and quality. What affects this delivery, and how to monitor is discussed in the following sections.

Table 2. Reported impacts from ongoing schemes

Impacts on land use
• In Sumberjaya, Indonesia, the Rivercare project has engaged now about 70% of the forest margin building on the Indonesian law for community-based forest management (7% only in 2004), and there is conditional land tenure in place for 6,400 farmers implementing practices like multi-story coffee gardens (Suyanto 2010). • The Costa Rica national program had covered nearly 730,000 hectares of forest, and planted nearly 3 million trees in agroforestry plantations by February 2010. • Between 2000 and 2007 the PSAH programme in Mexico had reduced the rate of deforestation from 1.6% to 0.6% (Muñoz-Piña et al. 2011) • The total plantation area in China amounts to 53 million hectares, including 28 million hectares of plantations in 6 years from the Sloping Lands Conversion Program, and 8.8 million hectares of crop have been converted to plantations from the Cropland to Forest Program (Sun and Chen 2006; Xu et al. 2010). • Small scale projects like San Pedro Norte in Nicaragua have now expanded to other areas, under the *Fondo Nacional de Desarrollo Forestal* (FONADEFO). For example, 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). • FONAG, Ecuador, had more than 65,000 thousand hectares under land management by 2008, but plantations established are still too young to have their impact evaluated and thresholds are regarded as very small (about 2.5% of all area are reforested). (Cannon, Hill, and McCarthy 2010)
Impacts on water quantity
• In Colombia, where there are statistic for local water availability upstream, long term trends for annual rainfalls and irregular fluctuations are due to large geographical scale causes rather than to the local forests status (Kosoy et al. 2005). • Economic impacts of protection of cloud forest on water are small. In Monteverde hydrological studies show that cloud forests have a modest impact on water budgets in the mountains, and even effects on dry-season flows are near neutral mostly because of the relatively regular rainfall in the area. Most economic value of marginal flows are 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, Carrasco, and Asquith 2009). Studies near the area covered by Fidecoagua in Mexico confirm this trend, although they also show that old secondary forest have roughly the same hydrological behavior than established natural cloud forest (Muñoz-Villers et al. 2010). • Studies of páramo nearby Cuenca, Ecuador, show that pine plantations decrease annual water yield, livestock grazing is not affecting soil compation because of low animal density, and cultivation mainly affects regulation through increase in the magnitude of peak flows and a reduction of base flows (Crespo et al. 2009). • Salas (2004) reported a severe reduction in water volume was reported following reforestation activities with fast growing exotic species (mainly mahogany and gmelina). • Not actually measured but estimated through models, the impacts on water flows of the Working for Water programme in South Africa have been estimated at 48–56 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, suggests that water quality in forested areas receiving payments is good, but it declines in lower areas of the watershed, suggesting potential problems with fish farms, and domestic or industrial wastewaters (Manson 2008). Fidecoagua implemented communal payments alongside the PSAH and reported less sediments but it is hard to judge the validity of this claim (Blanco and Rojo 2005)
- Protection of forest in El Triunfo has contributed to reduced sediment accumulation and lower heat levels, with positive effects on shrimp catches and the local fishing industry (Stem 2005), although it is difficult to attribute these impacts to a reduction in risk of deforestation without a baseline.
- San Pedro del Norte (Chinandega, Nicaragua) reported water sources becoming permanent after the introduction of the scheme (Obando 2007), although no in-depth hydrological study has been conducted and the relatively small scheme size makes it difficult unequivocally attribution. A study by Kosoy et al (2005) suggest that this scheme (alongside Jesus de Otoro and Puerto Barrios) are able to demonstrate with a degree of confidence that the environmental service is provided, or specify what threshold areas required to reach specific quality targets.
- A perceived reduction in pollution from coffee-processing wastes was reported in 2003. However, it was suggested that this had more to do with coffee prices and levels of activity, and that pollution would increase again as coffee prices recovered (Ardón and Barrantes 2003).
- A successful lesson from the Hunter River Salinity Trading in NSW shows that, covering a total area of 2.2 million hectares, salinity targets were met at all times in the Upper Sector of the river when discharges were allowed under the scheme (NSW 2009).

4.2 Transactional compliance monitoring

Monitoring efficacy implies providing assurances that the agreed upon land management or production practices have in fact occurred. Compliance and the resulting enforcement of PWS implementation is pivotal to program effectiveness and must be considered as a necessary condition for success. Inclusion of compliance and enforcement mechanisms into PWS schemes occurs more frequently than the inclusion of hydrological monitoring.

4.2.1 Indicators and activities promoted under PWS

Kroeger and Casey (2007) draw attention to the potential loss of ecosystem services resulting from the difficulty in identifying and defining a particular service unit and resulting ability (or inability) to measure service changes. Thus, the determination of how a particular service is to be measured indicates a reasonable constraint in monitoring and evaluation. Clear biophysical interconnections between land use activities prescribed by a PWS scheme and resulting hydrological effects need to be identified, understood, and communicated to all program participants, particularly providers and beneficiaries (Landell-Mills and Porras 2002).

As previously mentioned, PWS schemes are based on the implementation of particular land management practices intended to positively influence the provision of watershed services. When monitoring and evaluating PWS schemes these practices often serve as a proxy for watershed services, given the assumption that particular land management techniques will ‘increase’ the probability of desired service provision to downstream recipients (Porras, Grieg-Gran, and Neves 2008; Asquith and Wunder 2008; Ortiz, Sage, and Borge 2003). The word “increase” is used given that external factors – i.e. counfounders - will inherently influence hydrological conditions within watersheds (Asquith and Wunder 2008).

The utilization of a proxy action as representative of the production of the desired watershed service in programme assessment is repeatedly based on local assumptions and perceptions of biophysical interconnections, which tend to ‘oversimplify’ these complex ecosystem functions (Kaimowitz 2004; Echavarria et al. 2004; Gutrich et al. 2005). For example, the prevailing assumption is that a positive correlation exists between forest cover and the watersheds’ provision of water quality and quantity (Saberwal 1998; Rojas and Aylward 2003; Echavarria et al. 2004). Along this line of thought, changes and conditions in vegetation cover are used to indicate changes in hydrological functions (Asquith and Wunder 2008). Some exploit this occurrence by basing programme assessments on surveys of stakeholder perceptions pertaining to situational changes and perceived correlations between hydrological changes and reforestation resulting from PWS scheme (see Box 1) . However, given the complexity of watershed ecosystem functions and the clashes with common myths, the use of extrapolations from traditional actions (i.e. reforestation) to services (expected increased in water flows) can at least be questioned (Aylward 2005; Bruijnzeel 2004; Calder 2005; Bruijnzeel et al. 2010).

Box 1. Perceived impacts of some schemes

Asking users whether they perceive changes in water service since a PWS scheme was introduced is becoming a practice in lieu of scientific information. According to Kosoy *et al.*, (2005) the majority of water users in Jesus de Otoro, Honduras and Heredia, Costa Rica, perceive that water provision is the most important benefit from forests, and that a larger forest cover will lead to both better water quality and greater water availability. They also report that 64% of 100 users interviewed in Jesus de Otoro, Honduras, thought that water availability had improved during the two previous years, and 39% of 100 interviewed in Heredia, Costa Rica. However, in both cases, it is hard to see that any change could be attributed to the PWS schemes, given the small geographical area covered and timeframe. This difficulty of attribution to PWS applies to other cases as well. In Campamento, Honduras, there was a perceived reduction in pollution from coffee-processing wastes. However, it was suggested that this had more to do with coffee prices and levels of activity, and that pollution would increase again as coffee prices recovered (Ardón and Barrantes 2003).

The case of Meijiang (China) illustrates the difficulties in drawing conclusions on the impact of land-management practices introduced by PWS when farmer perceptions vary and other factors such as extraction of river sand may be contributing to reduced sedimentation. Leshan et al. (2005) interviewed several farmers involved in the scheme. Overall, in terms of water quality farmers perceive an initial increase in soil erosion while the new activities are in place (for example, building terraced strip, level ditch, bamboo ditch, and planting), while environmental pollution is not perceived as a problem despite fertilisers and pesticides being commonly used. There is no consensus on the impact of orchard development on water flow, with half of the interviewees saying that it had increased and half that it had decreased.

4.2.2 Monitoring adequacy of payment levels

PWS schemes involve some type of compensation, cash or in-king, and at different time scales (one-off, continuous). The type of payment will have implications on programme compliance. In-kind, one-off compensation may provide immediate benefits, but will be very difficult to retract in case of non-compliance. Smaller, continuous payments may encourage long-term compliance but if too small they may not be sufficient to induce expensive initial farm investments.

Opportunity cost calculations can play a significant role in assessing the adequacy of compensation allocated to service providers (Hoffman 2009). In theory at least, from the supplier (landowner) perspective (or compensation effectiveness/adequacy standpoint) payments must exceed, or at the very least meet, the opportunity cost of intended land-uses outside of those specified by a PWS scheme (Engel, Pagiola, and Wunder 2008; White and Minang 2011).

In addition to assessing adequacy, valuation of opportunity cost has the capability of enhancing environmental efficiency through the differentiation of payments, especially in instances when schemes operate under budgetary constraints or excess of demand. For example, within Bolivia's Fundacion

Natura scheme, overcompensation was prevalent among approximately 75 percent of enrolled lands, while others were considerably undercompensated (Hoffman 2009).

However, opportunity cost analyses tend to be unduly simplistic for a number of reasons:

- Such calculations of opportunity cost are often based on fairly basic cash flow analyses that fail to account for risk and uncertainties, as well as different time-scales that the rural land manager faces.

- Changing behaviour involves giving prior practices up, or "losing" what was previously realized through traditional land management. In such cases, behavioural economics has long noted the existence of an endowment effect (also known as divestiture aversion), which reflects the tendency of people to focus on the loss from a transaction (what they give up) and not on the gains from the transaction.

- Simple cash flow analyses do not take into account inner motivations that the seller may experience as part of the process of developing and implementing a PWS transaction.

These factors imply that a simple opportunity cost analysis (which can only provide a very static picture) will understate the incentive necessary to motivate the land manager to enter into a transaction and change their behaviour. Even where such opportunity cost pricing succeeds, the decision to participate could be influenced by non-monetary issues, like seller remorse, with subsequent impacts on program participation as word spreads through the community of potential sellers. Keeping an eye on opportunity costs is a necessary but not sufficient condition for successful PWS.

Beyond monitoring the sufficiency of payments 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). Methods for the valuation of hydrological services exist, as for any non-marketed good or service (Freeman 1993). However, in practice these can be difficult and costly to apply. To date much of this type of valuation work has taken place either in the context of academic studies – and thus not linked to PWS – or as preliminary work to justify PWS schemes. Unfortunately, much of the latter work attempts to use the opportunity costs of foregone production as the measure of value, an approach that is largely meaningless in terms of the environmental service (Rojas and Aylward 2003). The impact on the farmer's net profits from farming tells nothing about what the value of watershed services that would be derived from the land under alternative scenarios. The use of replacement costs approach suffers from similar limitations (Freeman 1991), as most analysis are theoretical and not actual behaviours.

4.2.3 Monitoring strategies

Monitoring as discussed here is field monitoring and should be distinguished from overall project monitoring, which can involve multi-stakeholder boards with independent audits (i.e FONAG trust fund in Ecuador). Thus, compliance monitoring so far focuses on enforcement of contracts, either directly with individuals, associations or local facilitators. In some cases, stakeholders and participants can conduct this process to a certain degree and thus reduce transaction costs when capacity is adequate on both sides (Reis, Sydness, and Barton 2007), although more often this type of "monitoring" refers merely to the collection of data (e.g. readings from staff gages or water quality tests) and not to the additional step of determining whether environmental additionality has been achieved.

This delegation of monitoring and enforcement to program participants is common in institutions responsible for managing sustainable common pool resources (Ostrom 1990). Intermediaries such as non-governmental organizations (NGOs) also offer cost-reducing assistance, such as that found in FUNDECOR's and CODEFORSA's provision of technical assistance to the Costa Rican national PSA, in exchange of a proportion of the payment (Porras et al. 2012). What this means in effect is that many transaction costs are shifted from the buyer to the seller.

The most common types of compliance monitoring that we see in existing schemes include:

- Self monitoring,
- Participatory monitoring: a) group monitoring, or groups of providers, b) interested parties monitoring.
- Expert monitoring through measurements and remote sensing monitoring (usually complemented by independent audit);

Insights from *Principal Agent Models* suggest that the most cost-effective schemes are likely to have a structure where provider has an incentive to truthfully monitor and report their own actions (Laffont and Martimort 2002). For example, *Bio-Rights* provides micro-credit to community members for sustainable projects. If the project delivers the expected outputs, then the loan becomes an actual payment. In this, and for group monitoring (below), clearly specified measurement protocols must exist for forest owners and households to follow when collecting data for monitoring. The Myrada project in India reports from 32 microwatershed reports that loans (rather than contributions) improves motivation to better monitoring and manage investments (Prakash Fernandez 2003).

Participatory monitoring can take place in two ways: monitoring within providers, and monitoring through multi-stakeholder groups (that may include service buyers). In the first case, service providers monitor each other and the buyer (i.e. intermediary, government, direct user) monitors the group and holds the group accountable for the input and outcome. Monitoring by peers is useful in systems where service delivery is contingent on multiple resource-management units. This has been successful in community forest management contracts in India, and social disapproval an important control tool in some cases, and it is suggested as a useful tool to ensure transparency and participation in REDD+ projects (Skutsch et al. 2009). Group pressure is useful in group contracts, where entire group of service providers has to bear the cost of a single individual's noncompliance. Monitoring by multi-stakeholder groups takes place especially in small schemes. For example, a voluntary commission is formed by representatives from the local communities, municipalities and/or NGOs involved, and make field visits and recommendations. This method has been used in Los Negros in Bolivia (Asquith, Vargas, and Wunder 2008), San Pedro Norte, Nicaragua (Porras, Grieg-Gran, and Neves 2008; Marín et al. 2006) and Fidecoagua, Coatepec in Mexico. Group monitoring can also prove ineffective in weakly formed groups with limited options for enforcing cooperative behaviour and dealing with free-riders, like the short-lived Group Contracts in the PSA in Costa Rica (Porras 2010).

Box 2. **Multi-stakeholder monitoring in Los Negros, Bolivia (Asquith, Vargas, and Wunder 2008)**

Baseline water flow and bird species diversity were not established before the scheme began, as it concentrated on vegetation cover and an assumed relationship to environmental services. Basic hydrological relationships remained unknown and assumed. Measuring and demarcating forest conservation plots is done with a hand-held GPS receiver, plotted onto a 2001 satellite image-based land-use map. The various forest types in the parcel are then mapped and their areas calculated. Farmers receive a copy of "their" map along with their contract. Field demarcation is by natural boundaries or trails, signs and wire fencing.

The Project Control Team (one member of upstream and downstream environmental committee, a nature field technician and the landowner) visits each enrolled property once a year. Monitoring is limited to land use changes, and uses GPS and maps (US$20 per diem) and submitting a report to the Enforcement Directorate (President of Natura and presidents of upstream and downstream environmental committees). The directorate makes recommendations on how to respond to infractions, if necessary.

More recently water flows began to be measured in several tributaries, and an avifaunal survey was done in 2005. Recent results show that the scheme is not having an impact on additionality, as land put under conservation is the least threatened by agricultural clearing (Robertson and Wunder 2005), and a rapid hydrological assessment showed no initial impact on water flows (Le Tellier, Carrasco, and Asquith 2009). A larger study in Rio Grande, along several universities including Vu University has began (2010-2014) http://www.naturabolivia.org/eva2.html .

Remote sensing monitoring is useful when checking for rough changes in forest cover. Some national schemes, like Mexico PSAH, Costa Rica PSA and Ecuador BioBosque use these systems, complemented by independent audits by forest regents. Several models exist 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.

Nevertheless, even compliance monitoring is typically less than optimal. Inadequate funding, lack of institutional capacity and capabilities (e.g. scarcity of enforcement personnel), poor communication between stakeholders, intermediaries, and regulatory officials all contribute to a reduction in monitoring compliance and enforcement (Echavarria et al. 2004; Wunder and Albán 2008). Compliance and enforcement mechanisms should be implemented during the beginning stages of a scheme, though some have been set up as 'learning by doing' exercises, as is the case in Pimampiro (Echavarria et al. 2004).

Sanctions resulting from enforcement typically take the form of time-based exclusion, cancellation or suspension of payments (Wunder and Albán 2008; Echavarria et al. 2004; Claassen, Cattaneo, and Johansson 2008), or the threat of civil legal action in case contracts are breached, like in the PSA Program in Costa Rica. Observed penalties and possibility of enforcement do have the capability of reducing the costs associated with monitoring through the 'perceived threat' of sanction imposition, particularly in smaller communities where social pressures and learning has the potential to encourage cooperation (Wunder, Engel, and Pagiola 2008; Wunder and Albán 2008). Information asymmetries in the form of hidden action (or moral hazard) can affect conditionality if the conservation agent may find monitoring contract compliance expensive and sanctioning noncompliance politically costly (especially in contracts are with poor groups), and thus fail to enforce the contract. While high fines are often used to deter non-compliance, the voluntary nature of PWS limits the range of sanctions. Under such conditions, the landowner has an incentive to breach contractual responsibilities (Pattanayak, Wunder, and Ferraro 2010; Wunder 2005).

On the other hand, the payments themselves can act as a compliance inducing mechanism to some degree. When participation is properly monitored and payments are truly conditional, providers 'learn' to comply, particularly when the level of payment truly compensates the provider for the services rendered (Kosoy et al. 2007; Claassen, Cattaneo, and Johansson 2008).

In sum, compliance monitoring and enforcement are necessary to demonstrate efficacy with regard to delivery of contracted-for watershed actions. Clarity as to the action required is one thing, clarity that taking this action leads to change in land management from what would have happened otherwise is, as mentioned above, another thing. The question remains of whether discussion of efficacy with regard to contracted-for actions is sufficient to demonstrate service delivery, or whether hydrological monitoring is also necessary.

4.3 Effectiveness compliance: additionality of the scheme

The land management or production practices that are incentivized through PWS schemes are intended as the cause that leads to an effect; the effect being a change in watershed or hydrological function. In the PES context, additionality can be defined as the scheme's ability to induce the intended outcomes, in an aggregate manner, as opposed to outcomes that would have happened had the scheme not been implemented (Shrestha and Timilsnia 2002; Muñoz et al. 2005; Claassen, Cattaneo, and Johansson 2008). This change in function is itself presumed to be of some consequence to human welfare – either in terms of economic costs and benefits or off welfare, particularly for the poor. Here we focus on the first step, which involves monitoring whether or not the PWS scheme can be said to be environmentally effective.

4.3.1 Indicators and baselines

With regard to watershed function the measurement of one or another component(s) of water quality and quantity is necessary in order to show an effect. When assessing water quality, measurement of sediment, nutrient, and chemical load reduction, and temperature are often employed. From a quantity standpoint, measurement of some feature of the hydrograph, such as base, peak or low flows are indicated. Demonstrating environmental additionality may well require a substantial baseline in both cases. Even when available, data sources are difficult to compare. Where hydrological records for water flow exist, they are from gages that have a long record of operation, while technologies for remote and continuous water quality monitoring are much more recent.

Asquith et al. (2008) identify the importance of composing a baseline for monitoring in mentioning that sound measurement and evaluation of PES schemes are reliant upon a preliminary understanding of biophysical interconnections in targeted areas. Most baselines existing until now focus on forest cover in properties, usually with GIS-based technology with ground-truth, rather than measuring the quality or quantity of the watershed service. The Working for Water programme in South Africa is an exception in the hydrological baseline developed prior to the beginning of the programme (Turpie, Marais, and Blignaut 2008). This is not to say that projects do not monitor changes in watershed services, but 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, Hill, and McCarthy 2010). Los Negros, Bolivia, currently has hydrological monitoring looking at changes in water quantity and also for changes in biota (see box above). The Working for Water programme in South Africa monitors the amount of alien species removed from river edges, but changes on water quantity are only estimated through modelling (Ferraro 2009). But many of these examples face the problem of attribution discussed before, as causal links remain uncertain, for example because of patchy or short-termed collection of information, stochastic events (such as weather patterns) masking local impacts, and uncertainty about groundwater movements.

Schemes from developed countries have been more successful at implementing monitoring based on outputs (i.e. level of ecosystem service provided) using baselines. In New Wales, Australia, the Hunter River Salinity Trading scheme has been successful in managing salinity levels along the river. The scheme monitors and caps the discharges of twenty-three coal mining and power generation facilities, and allows trading of credits 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). The River case project in Sumberyaja is an early example of payments made upon delivery of reduced sediments, with the added value of creating a market use for sediments trapped (Harto Widodo et al. 2006).

Ultimately, there is significant scarcity of schemes that have developed a measuring program based on a well-established baseline (Agarwal et al. 2007; Porras, Grieg-Gran, and Neves 2008). Obviously, the lack of even a baseline as to prior land management or practices is a fundamental obstacle to monitoring transactional compliance or service provision.

Box 3. Collecting information

How much research is then needed prior and during PWS implementation? Information is priceless, but it has a cost. How much research is needed for implementing PWS schemes varies depending on the nature and scale of the problem, and how much funds exist for this type of studies. The Bellagio Conversations (Asquith and Wunder 2008) provide a useful rule-of-thumb to begin a PWS scheme. The simplest case is conservation of existing ecosystems under threat, for maintenance of water quality or quantity. A mechanism 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 2007). Restoration of habitat is more complicated. Research will be required to demonstrate biogeochemical linkages, threshold levels and cost-effectiveness for water quality. Research for water quantity is even more complicated if no site-specific information is already available. The wisest course of action recommended is to undertake a series of inexpensive "no-regret" actions until full research is done to determine whether or not to implement a full-scale PWS scheme.

Van Noordwijk (2005) presents a useful description of the type of watershed service, key indicators, ways to measure them (or proxies if needed), and how these indicators can be monitored locally. n their book about Forests, water and people in the tropics, Bonell and Bruijnzeel (2005) present several papers on collection of information, including remote sensing technologies, statistical methods, and selection of models. These methods provide, within their own limits, the best existing information about the impacts of land use changes on water variables.

4.3.2 Attribution

Demonstrating environmental additionality involves much more than having a prior record of measurement of the desired hydrological variable. It is also necessary to understand the relationship between the action and that variable. This requirement may pose a further obstacle, even where a baseline record of water quantity or quality data do exist. While various methods exist the basic requirement here is about attribution, basically understanding whether and how factors other than the target actions affect water quality and quantity over time. This implies the ability to consult data on potentially confounding drivers, not just that of the principle action being addressed. With regard to PWS this step can be quite difficult and time-consuming. If this crucial step is not undertaken then faulty conclusions about cause and effect can be made and a PWS program declared a success, when in fact it has had little to no impact on watershed services, or the other way around: lack of evident outcomes may be linked to external factors other than programme design.

The problems of attribution can be easily attested in the Maasin watershed program in the Philippines. Three years after the program began the area was affected by a severe reduction in water flows, and significant siltation. Some blamed the use of fast growing species used (mainly mahogany and gmelina). However, other factors affecting included water utilities using more water than previous quotas, and increased siltation from tillage of new lands after farmers were removed from some parts of the watershed. None of the hypothesis were explored, however, and arguments stopped when it began to rain again (Salas 2004).

4.3.3 Targeting strategies

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

Targeting not only requires the selection of the appropriate location but also choosing the resources that will deliver services of the required quality. Typically in PWS there is one buyer (usually the government) providing monopsonic powers, including the ability to select and change prices which is an important tool for targeting by using price-differentiation.

Targeting critical areas can be done by directing payments towards watersheds important for human use of watershed services (i.e. for domestic water use or hydroelectricity), areas prone to degradation (high slopes, or along river edges), and those areas more at risk from changes in land use (for example, forest clearing for agriculture).

It is easier for local schemes to target precisely areas surrounding headwaters of main water supply (i.e. Pimampiro in Ecuador, Tacuba in El Salvador, Jesus de Otoro in Honduras and San Pedro Norte in Nicaragua). According to Wunder and Albán (2008), targeting of the small scale scheme in Pimampiro has contributed to stopping deforestation and to the marked recovery of native vegetation, contrary to trends happening in most neighbouring villages. Before the introduction of payments in 2000, approximately 30% of the total area had been converted to cropping and pastures, compared to a reduction of only 14% in 2005. It is hard to check what the effect on water availability is, with no baseline studies to draw on or appropriate counterfactuals. Downstream perceptions of increased water flows can be strongly affected by improved infrastructure created parallel to the beginning of the PES scheme (Echavarria et al. 2003). Lack of targeting in Los Negros, Bolivia, matched with very low compensation (which covers only 2–10% of the opportunity costs for setting aside agricultural land) means that the areas chosen by farmers to put into payments are those with less potential for agriculture and therefore in less risk of conversion (Asquith, Vargas, and Wunder 2008; Robertson and

Wunder 2005). Better targeting of critical areas may result in fewer participants but higher levels of compensation for each one, therefore increasing the attractiveness of the scheme (Asquith, Vargas, and Wunder 2008; Wunder 2008).

Lack of targeting can dilute potential benefits from large, usually national, schemes. The strong top-down approach from the Sloping Land Conversion programme in China, for example, shows that that 38% of area converted from agriculture to forestry in Gansu Province (and 10% in Shanxi and 11% in Sichuan) was low slope area and hence at little risk of causing erosion. Although these new forest areas can have a benefit for biodiversity and landscape, it is doubtful that it would have much impact on sediments downstream (Xu et al. 2004; Bennett and Xu 2008). Using treatment analysis, Xu et al (2010) demonstrate that increasing local autonomy could lead to improvement in local targeting, program cost-effectiveness and outcomes.

Lack of targeting in some voluntary conservation contracts, where landowners determine the supply of potential targets (i.e. Norway and Finland), have been criticized because they may result in a conservation network which does not cover all focal ecological characteristics (Barton 2010), although early analysis of the Trading in Natural Values in Finland show that while the program meets its ecological goals but it is still early to assess the long-term ecological effects (Juutinen, Mönkkönen, and Ylisirniö 2009).

The first years using a first come-first served approach in the Costa Rica and Mexico national schemes were notorious for minimal impact on lands at risk. In the isolated Peninsula of Osa (Costa Rica), land under protection contracts corresponds mainly to forestland that may not be in direct danger of being converted because of its isolation and difficult access (Sierra and Russman 2006). Analysis by Sanchez-Azofeifa et al. (2007) found that there was no significant difference in the rate of deforestation in 1997–2000 between areas in the national PSA scheme and areas that were not, although this can be debated given the low rates of deforestation during the late 1990s and early 2000s. Barton et al. (2009) showed that Costa Rica's next PES phase achieved a higher efficiency level due to improved targeting with respect to its 1999-2001 period. Still, the program does not yet focus on areas with higher deforestation risk, and this is reflected in its inability to have an effect on deforestation (Robalino et al. 2008; Sánchez-Azofeifa et al. 2007).

Mexico has undergone several stages related to targeting criteria. Initial distribution of contracts was ad-hoc, and much of the land being put under payments was not really at 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% of the participating hectares in the scheme were classified as having high or very high deforestation risk but this increased to 28% in 2004. Priority criteria was introduced for eligibility, including protection of forest in aquifers; areas with water scarcity; and flood-risk areas and other disasters associated to weather extremes. At the moment, there is a combination environmental, administrative and social eligibility criteria (in 2006 there were 9 criteria for eligibility, and 26 by 2010). Secondary criteria receives more weight than primary, and only one third of enrolled areas were under risk of significant deforestation, and increasingly criteria other than environmental clutter the program's ability to be environmentally effective (Muñoz-Piña et al. 2008; Alix-Garcia et al. 2005; Muñoz-Piña et al. 2011). Lessons from Costa Rica and Mexico have been learned, and the Ecuador national Socio-Bosque PES program incorporates risk of deforestation as criteria for participation (alongside with poverty levels of population and representativity of forest in the protected areas system).

National programmes in Costa Rica, Mexico, and China for their most part have not been very successful at targeting land more at risk (Sánchez-Azofeifa et al. 2007; Alix-Garcia, Janvry, and Sadoulet 2005; Alix-Garcia, Shapiro, and Sims 2010; Bennett and Xu 2008). Additionality of the programme is significantly reduced by concentrating on conservation of forest not much at risk (i.e. all payments in Mexico and a very large proportion in Costa Rica are for conservation). Explanations are multiple, linked to political interests and pressure from groups, and governments' tendency to "learn on the job". Increasing demand from farmers, tighter supply of funds, and pressure from academic groups pointing at inefficiencies, is leading to a new generation of programmes where risk is more seriously taken into account. Mexico is moving from an ad-hoc approach in contracts to more inclusion of risk. It remains to see whether existing payments would be enough to attract farmers in targeted areas, or whether differentiated payments would be required (at present Costa Rica has introduced higher payments for *hydrologically sensitive* areas, usually near city centres and with higher opportunity costs).

4.4 Monitoring unintended impacts

4.4.1 Leakage

Leakage (or slippage) is defined as an unintentional effect of participation in PES or other conservation schemes in which participation efforts are offset by degradation in other areas under the control of a PES enrolee. Generally, this issue is associated with the extension of cropland areas outside of lands participating in the PES scheme while land within the scheme is being managed in compliance with program specifications (Hoffman 2009; Claassen, Cattaneo, and Johansson 2008). This is less of as issue in local PWS schemes that target specific catchments for participation, though potentially greater for larger (often government-financed) schemes (Ross, Depro, and Pattanayak 2006). Because of the largely unknown nature of inter-basin dynamics, one cannot assume that leakage-induced degradation outside the project area will not affect the hydrology within it.

Another unintended consequence of PES schemes is found in perverse incentives such as the purposeful degradation of lands in an effort to acquire a means to be compensated for restorations efforts specified by a PES program. Cessation of current restoration efforts to receive payments also falls into this category (Wunder and Albán 2008). For this reason FAO calls for the need to design monitoring mechanisms that identify perverse incentives given the potential impact of their effects and the likelihood of their occurrence (FAO/REDLACH 2004).

4.4.2 Monitoring equity impacts

Pro-poor strategies are often implicit or explicit objectives of PWS schemes, particularly in government financed programs. The justification of these inclusions is sourced in the observation that activities performed by providers of watershed services both directly and indirectly affect lower-income groups regardless of where they are located within the watershed. Increasingly, PES programmes (pushed by the REDD+ agenda) highlight the necessity to examine environmental effectiveness and the social and governance enabling conditions at the local level that affect legitimacy, participatory and distributional justice (Svarstad et al. 2011).

Within the PES context, land tenure tends to be the primary factor excluding the poor from participating (Porras, Grieg-Gran, and Neves 2008; Pagiola, Arcenas, and Platais 2005; Grieg-Gran, Porras, and Wunder 2005). For example, watershed development programs located in India have resulted in the exclusion of the poor from areas formerly utilized as common lands (Reis, Sydness, and Barton 2007). In response to parallel issues, the RUPES project in Sumberjaya, Indonesia has supported the acquisition of land tenure and user rights among participating farmer groups who perceive this compensation as highly valuable (Dasgupta 2007). Supporting enforcement of land tenure is a component in Los Negros, Bolivia, providing participants with property maps and barbed wire to reduce illegal seizure of lands enrolled (Asquith, Vargas, and Wunder 2008).

Property size is another key deterrent in two ways. First, small farmers usually require a mix of economic activities within their plot to distribute risk and ensure short-term food supply. Second, the need to reach specific threshold levels to provide a meaningful level of service implies the need to target large areas, for which small, scattered properties make less economic sense as transaction costs escalate.

PWS projects that focus on forest protection, or require high initial investments, will deter participation of smaller, farm-dependent landowners. Proposals like the Green Water Credits project in Kenya, which focus on feasible techniques for small rain-fed farmers (like mixed cropping, terracing and agro forestry) to increase the soil's ability to retain soil moisture after rain events are more likely to be have a better social impact that those focusing on protection-only.

It is relatively easier for small-scale projects to focus on specific areas, activities, participants (farmers and/or associations), and be flexible about property rights as other forms of tenure can be locally accepted. For example, in Pimampiro in Ecuador the local municipality has dealt directly with an association working out the best practices for farmers (Wunder and Albán 2008). In Central America PASOLAC has supported several small-scale schemes involving farmers and communities, making the process participatory and inclusive (Obando 2007).

This process is more difficult to reach for national-level schemes, like Mexico or Costa Rica. The selection process for contract allocation must be a transparent process with similar rules for all, and cannot be lifted easily. In Costa Rica, 'non-negotiable' requirements, like property titles, no-mortgages (only acceptable with certain banks), can leave out vulnerable farmers. The use of aggregate indicators of poverty (like the Social Development Index < 40 used in Costa Rica) as preference criteria to allocate contracts in poorer areas has limited success, as it targets areas and not farmers, and would allocate equal preference to large landowners living in the target area (Porras 2010). The balance between environmental and other (social, political, administrative, etc) criteria is a problem in national schemes that must also strive for transparency in the allocation of public funds. In the PWS program in Mexico, less than 20% of the points allocated to contract selection are of hydrological importance (Muñoz-Piña et al. 2011).

5. Conclusions

As the levering capacity for demanding 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 to raise funds (targeting the 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 how cost-effective this project might be as cost increase. By definition, the attractiveness of PWS schemes relies on its ability to create an incentive to change a behaviour, and whether this change in behaviour generates net positive impacts on the level of the watershed service.

In this article we discuss the monitoring and evaluation criteria behind compliance, or transactional, monitoring to ensure that the contracts are followed, and effectiveness conditionality, that looks at how the scheme has managed to achieve its environmental objectives regardless of the degree of compliance. Although both are usually linked, a high degree of compliance does not necessarily produce a high degree of effectiveness. A poorly designed scheme, for example, may target the wrong land managers and land at least risk and, thus, the payments may not generate the desired hydro-ecological or conservation benefits.

At the moment most schemes in developing countries focus on compliance monitoring. Even this is typically less than optimal. Inadequate funding, lack of institutional capacity and capabilities (e.g. scarcity of enforcement personnel), poor communication between stakeholders, intermediaries, and regulatory officials all contribute to a reduction in monitoring compliance and enforcement.

Demonstrating environmental additionality involves much more than having a prior record of measurement of the desired hydrological variable. It is also necessary to understand the relationship between the action and that variable. This requirement may pose a further obstacle, even where a baseline record of water quantity or quality data do exist.

While various methods exist the basic requirement here is understanding whether and how factors other than the target actions affect water quality and quantity over time. This implies the ability to consult data on potentially confounding drivers and a sound theory of change model behind the scheme, not just that of the principle action being addressed. With regard to PWS this step can be quite difficult and time-consuming. If this crucial step is not undertaken then faulty conclusions about cause and effect can be made and a PWS program declared a success, when in fact it has had little to no impact on watershed services. In particular, if no baseline information is available and no cause and effect relationship established, development of a PWS scheme will need to rely on significant scientific extrapolation or local knowledge. More to the point it will be effectively impossible to link cause and effect through monitoring. In such cases all that can be done is to monitor contract compliance and relevant hydrological variables. If the hydrological condition does not improve it can be intuited that the PWS scheme is not working, but why it is not working cannot be established. If conditions do improve, the only conclusion that can be drawn is that the PWS scheme has probably not hurt the situation, but there can be no proof that the improvement is due to the PWS scheme.

A better understanding of these relationships is vital for the long-term health of existing initiatives. So far, the growing interest shows that participants believe in the principle of land management. Evidence of impact will be required none the less to ensure that actions are truly additional, and root out those are merely self-serving enrichment at the public trough.

Annex 1. Monitoring and impacts in existing PWS schemes

Case	Extend[4]	Design and monitoring	Reported results
Australia, Hunter River Salinity Trading, New South Wales	Output based, local 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 twenty-three coal mining and power generation facilities. These facilities 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 over the scheme's online trading platform. During low flow conditions, no discharges are allowed, and during flood conditions discharges are unrestricted.	The service area covers a total of 2,200,000 hectares. The scheme achieved excellent results during 2008–09. Salinity goals were met in the Hunter River at all times in the Upper Sector of the river when discharges were allowed under the scheme
Bolivia, Los Negros (Asquith, Vargas, and Wunder 2008; Asquith and Vargas 2007; Le Tellier, Carrasco, and Asquith 2009)	Local initiative, input-based	Fundacion Natura has implemented an on-site monitoring program for participating farm lands that is conducted by a Project Control Team every 12 months. The team is composed of one upstream and downstream community member, a Natura Fundacion field technician, and the land owner. Reports derived from the monitoring process are provided to the enforcing agency that imposes final sanctions. In response to a formerly non-existent hydrological baseline and insufficient monitoring data, the Fundacion Natura executes biweekly quantity-based hydrological measurements within the watershed's tributaries.	Additionality of case questioned as land in scheme is not that with highest deforestation risk, but it has not been rigorously measured. A hydrological study found no relationship between forest cover and streamflow.
China (Sun and Chen 2006; Bennett and Xu 2008; Xu et al. 2010; Xu et al. 2004). Forest Ecological Compensation Sloping Land Conversion	Input-based national schemes	National government programme through which farmers must set aside erosion-prone farmland within critical areas of the watershed of the main rivers in China. Monitoring is done at the local level by local governments.	Although the numbers of reported plantations are staggering (28 million hectares of plantations in 6 years; converted 8.8 million hectares of crop to plantations, soil erosion has been reduced by 4.1 million ha, total plantation area 53 million hectares), some studies of the SLCP suggest low rate of survival and sufficient monitoring, suggesting that programme may slip into a one-off transfer to participants, with little substantive environmental outcomes, No info on impacts on water flows.

[4] Input-based schemes are based on the assumption of delivery of environmental services by a given land-based activity. Output-based schemes try to measure the actual environmental services provided. See Section **Error! Reference source not found.**

Programme (SLCP) Conversion of Cropland to Forest Program (CCFP)			
Colombia, Cauca Valley, (Kosoy et al. 2005; Echavarria et al. 2003)	Input-based, local schemes	Monitoring of land use is done by the water associations.	Despite continuous monitoring of land use, no studies on water flows have taken place. Anecdotal evidence suggest that between 1988 and 1998 levels of the Desbaratado River did not present the extreme flooding incidents that occurred previously, and there were improvements in dry-season flows for the Nima and Amaime watersheds. But this data is limited, and the absence of concrete figures makes it difficult to assess the actual hydrological impacts of the interventions of the project (Echavarria 2002). According to Kosoy (2005), long term trends for annual rainfalls and irregular fluctuations are due to large geographical scale causes rather than to the local forests status.
Costa Rica, La Esperanza (TEEBcase Porras 2010; Rojas and Aylward 2003)	Local independent scheme, input-based (payments based on energy production) but later on moved to output-based.	Baseline using extrapolated data from other local watersheds. No monitoring processes have been implemented, although changes in water quantity are estimated through the amount of hydroelectricity produced.	Land were payments are made are protected by a private reserve. No change in land use has taken place, and no changes in water flows are expected.
Costa Rica, PSA Program, (Porras 2010; Blackman and Woodward 2010; Sánchez-Azofeifa et al. 2007; Arriagada et al. 2010)	National programme, input-based	Hydrological priority criteria for contract allocation (1 of 7 criteria) and higher payment. Monitoring is done observing changes (or avoidance) to land use, visual inspections and satellite pictures. Non-compliance is punished by withdrawing payments, and in extreme cases through civil legal action. “Hydrological sensitivity” priority criteria for protection contract allocation (1 of 7) and higher payment per hectare (US$400/.ha/5 year as opposed to US$320/ha/5 years for regular protection) Payments are demand-led, and are mostly for conservation. Compliance on forest cover observed, and effectiveness increasing as the programme moved from first-come, first-served to targeting; but no studies on impacts on watershed services.	By Feb 2010 the program had covered nearly 730,000 hectares of forest, and planted nearly 3 million trees in agroforestry plantations. Lack of targeting results in modest to minimum impact on deforestation. Studies in cloud forests show modest impact on water budgets.
Ecuador, Cuenca,	Local initiative,	The Macua Project from the University of	Hydrological studies wet Andean páramo

(Echavarria et al. 2003; Crespo et al. 2009; Stanton et al. 2010)	input-based	Cuenca, Ecuador, was established in 2002 to compile land and hydrological data in the four watersheds providing water to the town of Cuenca. Activities composing this project included: installation of network-linked meteorological monitoring equipment, a demand (designated) and supply (availability) assessment of waters in the watersheds, water quality studies, and soil studies. Data collected for this project offers the ETAPA (Cuenca's municipal water utility) well-established baseline data and continued monitoring abilities for the implementation of future PWS programs in the Yanuncay watershed.	ecosystem in nearby areas show links of land use on water: 1) pine plantations decrease annual water yield as a consequence of increased evapotranspiration; 2) because of low animal density, livestock grazing does not seem to affect the hydrological response; 3) cultivation mainly affects regulation through increase in the magnitude of peak flows and a reduction of base flows (Crespo et al. 2009).
Ecuador, Fonag, (Cannon, Hill, and McCarthy 2010; Echavarria et al. 2003; Southgate and Wunder 2007)	Local initiative, input-based	Conditionality is not an issue pursued by the program. Instead, payments from users of watershed services are being directed to conventional conservation projects (Southgate and Wunder 2007). So far FONAG has managed fairly sophisticated weather stations and soil moisture monitoring stations installed and working correctly (Cannon, Hill, and McCarthy 2010).	By 2008 more than 65,000 ha were under management, but plantations are still too young to estimate their impact. A most basic understanding of the water budget in any of the watersheds, and almost nothing is known about how changes in land-use activities might affect any of their respective water budgets. But effects likely to be small, because 1) threshold levels very small - about 2.5% of all area are reforested; and 2) farmers living upstream may use the additional flows for their own (Cannon, Hill, and McCarthy 2010).
France, Vittel	Local private scheme, output and input based (Perrot-Maitre 2006; Wunder 2008)	Water quality monitored extensively (300 tests per day) given the strict regulations required to market their product as "natural spring water". In addition to in-house monitoring, the Institut National de la Recherche Agronomique (INRA) undertakes compliance monitoring at 17 participating farm sites, and the INRA, Agrivair (the intermediary developed by Vittel to 'negotiate and implement' the PWS scheme) monitors compliance of farming practices among participating lands.	These monitoring practices incorporated into Vittel scheme have generated clear and measurable improvements in water quality
Honduras, Jesus de Otoro, (Kosoy et al. 2005)	Input-based local scheme, PASOLAC supported	Water authority has been monitoring water quality since 1999. Base studies included basic hydrological information, topography, precipitation, types of soils, etc., but no direct measurements of sediments.	Monitoring shows some "improvement" since the introduction of PWS. Btu small scale of this pilot PWS scheme and the lack of knowledge of underground water movements make it unlikely that it is the main reason for this improvement, which can also be linked to using chlorine used to treat water. Fieldwork by independent study (Jesus de Otoro, Puerto Barrios and San Pedro Norte) showed that none of the case studies demonstrate with a degree of confidence that the environmental service is provided, or specify what threshold areas required to reach specific quality targets.
Honduras, Campamento, (Ardón and	input-based, local initiative (but no cash payments made yet).	No baseline studies prepared, and monitoring of changes of land use is done but not of changes in water quality or quantity.	A perceived reduction in pollution from coffee-processing wastes was reported in 2003. However, it was suggested that this had more to do with coffee prices and levels of

Barrantes 2003; Kosoy et al. 2005)			activity, and that pollution would increase again as coffee prices recovered.
Honduras, El Copan (Villanueva et al. 2008; Alpizar, Blackman, and Pffaf 2007)	input-based, local scheme (proposed)	Highly detailed management plans at farm level have been designed to improve productivity with payments based on existing land use and activities required. Targeting of work in areas with important users downstream (high-benefit), and where drainage has maximum impact (high-risk). Baseline developed for current land uses and given points according to their perceived benefits (i.e. primary forest with surveillance highest rank).	Too early for results, but this new initiative builds on lessons from previous studies (especially Silvopastoril).
Indonesia: RiverCare, Sumberjaya (Suyanto 2010; Harto Widodo et al. 2006)	input-based local scheme, supported by RUPES, and testing for output-based (payments upon delivery of reduced sediments). Builds up on new law.	The scheme, supported by RUPES, is testing for output-based (payments upon delivery of reduced sediments). Builds up on new law. Socioeconomic baseline through rapid rural appraisals. Monitoring is on three aspects: infrastructure, institutional aspects and actual sediment reduction in the river. Community groups who learn how to trap and use sediments. A financial reward scheme provides some funds upfront and then pays additional specified amounts based effects achieved.	About 70 percent of the forest margin is covered by agreements (remaining forest in progress – 7% only in 2004). Conditional land tenure in place for 6400 farmers. Implementation of practices like multi-story coffee gardens,
Mexico Chiapas (including the El Triunfo and La Encrucijada Biosphere Reserves) (Stem 2005)	Local initiative linked to national PSAH Program. input-based	Monitoring is input-oriented and struggles with how to establish a link between output indicators and changes in higher-level watershed conservation targets.	For the most part evidence is anecdotal and supports positive perception for restoration, suggesting lower sediment accumulation and lower heat levels, with positive effects on shrimp catches and the local fishing industry.
Mexico, Fidecoagua, Coatepec, (Holwerda et al. 2010; Muñoz-Villers et al. 2010; Blanco and Rojo 2005)	Local initiative linked to national PSAH Program. input-based	Initial baseline study for forest cover in properties and literature review, but no measurement of water quantity/quality. PES and direct purchase of plots for conservation. GIS images used to monitor for forest cover, field visits and field supervision. Independent studies have also been measuring impact of cloud forests on water flows.	Increasing number of land receiving community payments (over 2300 ha between 2003 and 2010). Perceived positive impacts in water quality because less sediments, but no hard info. Hydrological studies conducted between 2002 and 2007 show smaller than expected impact on flows, but also shows secondary forest impacts similar to primary forests in terms of water at cloud levels.
Mexico, PSAH Program (Muñoz-Piña et al. 2011; Muñoz-Piña et al. 2008; Alix-Garcia, Shapiro, and Sims 2010)	National programme, input-based	Higher payments for cloud forests and forest with high deforestation risk. Monitoring based on changes on forest cover (GIS and satellite images).	Hydrologic importance and risk of deforestation represents only 20% of total priority criteria (out of 26 criteria). Secondary criteria (administrative, social, etc) diverts funds from where they can be more effective. Program has significantly (but small) reduced deforestation. But slippage effects can only be accounted for at the national level..

Mexico, Zapaliname, (Canales 2006; Lechuga 2009)	Local initiative linked to national PSAH Program. input-based (protection of existing reserve)	No baseline study. Monitoring is done as part of the Reserve's regular activities.	To 2009 only 14% of water users were paying the voluntary fee. There is some monitoring of water springs for quality, but their results are not available yet.
Mexico. Procuenca, (Valle del Bravo). (Porras, Grieg-Gran, and Neves 2008; Manson 2008)	Local initiative linked to national PSAH Program. input-based	Baseline studies for water quantity and quality in the rivers and dams, provides equipment and training for local stakeholders-led hydrological assessments and monitoring.	Suggest that water quality in forested areas receiving payments is good, but it declines in lower areas of the watershed, suggesting potential problems with fish farms, and domestic or industrial wastewaters
Nicaragua, Esteli (FAO 2010; Espinoza no date)	Input-based local scheme using in-kind downstream payments in form of labour, PASOLAC supported.	Patchy hydrological baseline study listing average monthly precipitation and rough description of catchment area. No attempt at differentiating seasonality from land use. No plans for monitoring.	Lack of legal framework has not allowed project to begin collection of fees. The use of ordenanza municipal has not been enough to enforce. To 2010 the only funds that had been collected were barely enough to cover the cost of collecting them (FAO 2010).
Nicaragua, San Pedro del Norte, (Obando 2007)	Input-based local scheme (including building dikes and water retention units), PASOLAC supported	Basic feasibility studies highlighted the main hydrological variables (precipitation, soils, etc). Some records of water exist in the local municipality before the initiative took place, but no in-depth hydrological study for the area was conducted.	Reports improvements in water sources after implementation of best management plans began in 2004, reporting seasonal water sources becoming permanent with significant increases in total water flows. Some of this extra water is used in the farms, but the extra water available downstream has helped to increase supply of water from 14% to 32% of total demand. The impact of the scheme may increase if funds are sufficient to include more farmers, as at the moment it only pays 5 farmers out of 43 in the catchment area.
South Africa, Working for Water program (Le Maitre, Versveld, and Chapman 2000; Swallow et al. 2009; Swallow et al. 2007; Turpie, Marais, and Blignaut 2008; DWAF 2006)	Input based (number of trees) national scheme Focuses on the removal of alien invasive plants that consume large quantities of water.	Monitoring program based on extensive studies on water consumption of non-native/invasive plant species. The baseline developed for the WfW program is reliant upon levels of streamflow reduction associated with vegetation type and distribution. Additionality has been addressed with the use of GIS-linked monitoring of invasive species removal (10,000 kn2)	The impacts of this programme on water flow have been estimated, through intensive modelling, at 48–56 million cubic metres of additional water per annum.
The Philippines, Maasin, (Salas 2004)	input-based	This is a long-term government-led watershed management programme. The scheme focuses on social forestry to improve watershed management. The project involved training and	A severe reduction 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

		management for large-scale tree planting. Farmers were required to move from some parts of the watershed.	flows were the product of higher water outtakes from a local utility, and increased silt from relocated farmers tilling. There were no studies supporting any side, and “arguments ended when the rains began” (Salas, 2004).
USA, Conservation Reserve Program (CRP)	National scheme, input-based (Claassen, Cattaneo, and Johansson 2008)	Program has developed and implemented a highly effective targeting process. Eligibility is dependent on land and soil conditions and the resulting contribution to ecosystem improvement. The assessment of these conditions is dependent on the CRP’s Environmental Benefits Index (EBI) which assesses various environmental concerns relative to the costs of enrolling the lands) in the program. The CRP’s targeting-based effectiveness is increased through a reverse auction process in which the service provider bids against others to participate in the program Gains of additionality compare changes in land management specific to program against changes assumed to occur in the program’s absence.	
USA, Castskills, New York	Local municipal scheme, output based (Appleton 2002)	Combines land easements and acquisitions with custom-designed pollution control measures for each farm that maximizes effectiveness and minimizes costs that ensures benefits to the farmer.	Benefits are often not cash but time and ease of labour, for example on manure disposal. Within five years of programme creation 93% of all the farm in the NY City watershed were participating, and the programme is hailed as the most successful non-point pollution control in the USA.

6. Bibliography

Agarwal, Chetan;, Sunandan; Tiwari, Mamta; Borgoyary, Amitangshu; Acharya, and Elaine Morrison. 2007. Fair deals for watershed services in India. In *Natural Resource Issues*: International Institute for Environment and Development.

Alix-Garcia, Jennifer, Alain de Janvry, and Elisabeth Sadoulet. 2005. A Tale of two communities: explaining deforestation in Mexico. *World Development* 33 (2):219-235.

Alix-Garcia, Jennifer;, Alain; de Janvry, Elisabeth; Sadoulet, Juan Manuel; Torres, Josefina; Braña Varela, and Maria; Zorilla Ramos. 2005. An assessment of Mexico's payment for environmental services program. In *Roles of Agriculture Project Environmental Services*: Agricultural and Development Economics Division (ESA) of the FAO.

Alix-Garcia, Jennifer;, Elizabeth; Shapiro, and Katharine Sims. 2010. Forest conservation and slippage: evidence from Mexico's National Payments for Ecosytem Services Program. In *WORKING PAPER August 6, 2010*.

Alpizar, Franciso; , Allen; Blackman, and Alexander Pffaf. 2007. Payments for Ecosystem Services; Measurements with Impact. *Resources* 165.

Appleton, Albert 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. In *The Katoomba Conference* Tokyo, November 2002.

Ardón, Mario; , and Gerardo; Barrantes. 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. .

Arriagada, Rodrigo Antonio;, Paul; Ferraro, E; Sills, S.; Pattanayak, and S.; Cordero. 2010. Do payments for environmental services reduce deforestation? A farm level evaluation from Costa Rica.

Asquith, N., M. Vargas, and S. Wunder. 2008. Selling Two Environmental Services: In-Kind Payments for Bird Habitat and Watershed Protection in Los Negros, Bolivia. *Ecological Economics* 65:675-684.

Asquith, Nigel.;, and Maria Teresa; Vargas. 2007. Fair deals for watershed services in Bolivia. In *Natural Resource Issues* International Institute for Environment and Development.

Asquith, Nigel;, Maria Teresa; Vargas, and Sven; Wunder. 2008. Selling two environmental services: In-kind payments for bird habitat and watershed protection in Los Negros, Bolivia *Ecological Economics* 65:675-684.

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

Aylward, B. 2007. Agricultural Landscapes and Domestic Water Supply: the Scope for Payments for Watershed Services with special reference to sub-Saharan Africa. In *Payments for Environmental Services in Agricultural Landscapes Series*. Rome: FAO.

Aylward, Bruce. 2005. Towards watershed science that matters. *Hydrological Processes* 19 (13):2643-2647.

Barton, David;. 2010. Assessment of existing and proposed policy instruments for biodiversity conservation: Norway case study. In *POLICYMIX*

Barton, David; , D.P.; Faith, Graciela; Rusch, M.; Acevedo, and M. Castro. 2009. Environmental service payments: Evaluating biodiversity conservation trade-offs and cost-efficiency in the Osa Conservation Area, Costa Rica. *Journal of Environmental Management* 90:901-911.

Baylis, Kathy, Stephen Peplow, Gordon Rausser, and Leo Simon. 2008. Agri-environmental policies in the EU and United States: A comparison. *Ecological Economics* 65 (4):753-764.

Bennett, Michael T., and Jintao; Xu. 2008. China's sloping land conversion program: Institutional innovation or business as usual? *Ecological Economics* 65 (4):699-711.

Berttram, Davina. 2011. Positioning the Kagera TAMP project in the PES landscape of East Africa. edited by K. Agro-Ecosystems: FAO (NRL).

Blackman, Allen, and Richard T. Woodward. 2010. User financing in a national payments for environmental services program: Costa Rican hydropower. *Ecological Economics* 69 (8):1626-1638.

Blanco, R.;, and P.; Rojo. 2005. Fideicomiso para el pago de servicios

ambientales forestales Coatepec, Veracruz.

Bond, Ivan, Maryanne Grieg-Gran, Sheila Wertz-Kanounnikoff, Peter Hazlewood, Sven Wunder, and Arlid Angelsen. 2009. Incentives to sustain forest ecosystem services: a review and lessons for REDD. In *Natural Resource Issues*: International Institute for Environment and Development, London UK, with CIFOR, Bogor, Indonesa, and World Resources Institute, Washington D.C., USA.

Bruijnzeel, L.A. 2004. Hydrological Functions of Tropical Forests: Not Seeing the Soil for the Trees? *Agriculture, Ecosystems and Environment* 104:185-228.

Bruijnzeel, Leendert Adriaan;. 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? *Agriculture, Ecosystems & Environment* 104 (1):185-228.

Bruijnzeel, Leendert Adriaan;, Maarten; Kappelle, Mark; Mulligan, and Frederick N.; Scatena. 2010. Tropical montane cloud forests: state of knowledge and sustainability perspectives in a changing world. In *Tropical Montane Cloud Forests: Science for Conservation and Management*, edited by L. A. Bruijnzeel, F. N. Scatena and L. S. Hamilton: Cambridge University Press.

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

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

Calder, Ian;. 2005. *The Blue Revolution: land use and integrated water resources management*. London, UK: Earthscan Publications Ltd.

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

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

Casasola, Francisco; , Muhammad; Ibrahim, Claudia; Sepúlveda, Ney Ríos, and Diego Tobar. 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 *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*, edited by Sepúlveda C.and Ibrahim M. Turrialba.

Claassen, Roger, Andrea Cattaneo, and Robert Johansson. 2008. Cost-effective design of agri-environmental payment programs: U.S. experience in theory and practice. *Ecological Economics* 65 (4):737-752.

Crespo, P.;, R.; Celleri, W.; Buytaert, J.; Feyen, V.; Iñiguez, P.; Borja, and B.; de Bievre. 2009. Land use change impacts on the hydrology of wet Andean páramo ecosystems. Paper read at Status and Perspectives of Hydrology in Small Basins, 30 March-2 April 2009, at Goslar-Hahnenklee, Germany.

Dasgupta, Purnamita. 2007. Report on the Self-Evaluation of RUPES. Bogor: World Agroforestry Centre.

DWAF. 2006. Working for water strategic Plan 2004/5-2007/8, internal draft South African Department of Water Affairs and Forestry.

Echavarria, M., J. Vogel, M. Alban, and F. Meneses. 2004. The Impacts of Payments for Watershed Services in Ecuador: Emerging Lessons from Pimampiro and Cuenca. edited by I. I. f. E. a. Development. London: International Institute for Environment and Development.

Echavarria, Marta; , Joseph; Vogel, Montserrat; Albán, and Fernanda Meneses. 2003. The impacts of payments for watershed services in Ecuador. Emerging lessons from Pimampiro and Cuenca. International Institute for Environment and Development.

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

Engel, Stefanie;, Stefano; Pagiola, and Sven; Wunder. 2008. Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics* 65:11.

Espinoza, Abdón;. no date. Mecanismos de compensación relacionando bosques con agua en Centroamérica y El Caribe de habla hispana. INAFOR, FAO.

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

FAO/REDLACH, ed. 2004. *Electronic forum on Payment Schemes for Environmental Services in watersheds. Final report.* : FAO

Ferraro, Paul J., and Subhrendu K. Pattanayak. 2006. Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments. *PLoS Biology* 4 (4):482-488.

Ferraro, Paul;. 2009. Regional review of Payments for Watershed Services: Sub-Saharan Africa. *Journal of Sustainable Forestry* 28:525-550.

Freeman, A. Myrick. 1991. Valuing Environmental Resources Under Alternative Management Regimes. *Ecological Economics* 3:247-256.

Repeated Author. 1993. *The Measurement of Environmental and Resource Values*. Washington DC: Resources for the Future.

Grieg-Gran, Maryanne; , Ina; Porras, and Sven Wunder. 2005. How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. *World Development* 33 (9):1511-1527.

Gutrich, J.;, D.; Donovan, M.; Finucane, W.; Focht, F.; Hitzhusen, S.; Manopimoke, D.; McCauley, B.; Norton, P.; Sabatier, J.; Salzman, and V. Sasmitawidjaja. 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, Rudy;, S.; Suyanto, Bruno; Verbist, Betha; Lusiana, Susanto;, and Prantiknyo Purnomosidhi. 2006. Rewarding communities for keeping rivers clean? First steps in a RiverCare Program in West Lampung- Indonesia. In *International Symposium On Community Activities for the Conservation Of Water Environment*. Bangkok, Thailand: ICRAF.

Hartwell, Ray, and Bruce Aylward. 2008. The Appropriateness of Voluntary and Involuntary Actions in Supporting the Development of Regulatory and Non-Regulatory Sustainable Financing Mechanisms. Bend, OR: Ecosystem Economics.

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

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

Ibrahim, Muhammad; , Francisco; Casasola, Cristóbal; Villanueva, Enrique; Murgueitio, E.; Ramirez, J.; Sáenz, and Claudia; Sepúlveda. 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.

IFAD. 2012. Project Design Report. Upper Tana Catchment natural resource management project.: The International Fund for Agricultural Development.

Johnson, Nels, Andy White, and Daniéle Perrot-Maitre. 2001. Developing Markets for Water Services from Forests: Issues and Lessons for Innovators. Washington, DC: Forest Trends, World Resources Institute, The Katoomba Group.

Juutinen, Artti, Mikko Mönkkönen, and Anna-Liisa Ylisirniö. 2009. Does a voluntary conservation program result in a representative protected area network?: The case of Finnish privately owned forests. *Ecological Economics* 68 (12):2974-2984.

Kaimowitz, David;. 2004. Useful myths and intractable truths: the politics of the links between forests and water in Central America. In *Forests, water and people in the humid tropics: past, present, and future hydrological research for integrated land and water management.*, edited by M. Bonell, ; and L. A. Bruinjzeel. Cambridge, UK: Cambridge University Press.

Kalacska, M., G. A. Sanchez-Azofeifa, B. Rivard, J. C. Calvo-Alvarado, and M. Quesada. 2008. Baseline assessment for environmental services payments from satellite imagery: A case study from Costa Rica and Mexico. *Journal of Environmental Management* 88 (2):348-359.

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

Repeated Author. 2007. Payments for Environmental Services in Watersheds: Insights from a Comparative Study of Three Cases in Central America. *Ecological Economics* 61 (2-3):446-455.

Kroeger, T., and F. Casey. 2007. An assessment of market-based approaches to providing ecosystem services on agricultural lands. *Ecological Economics* 64:321-332.

Laffont, J.-J.;, and D.; Martimort. 2002. *The Theory of Incentives*: Princeton University Press.

Landell-Mills, Natasha;, and Ina; Porras. 2002. *Silver bullet or fools' gold: A global review of markets for forest environmental services and their impacts on the poor*. Edited by I. f. S. P. S. F. series. London: International Institute for Environment and Development (IIED).

Le Maitre, D.C.;, D.B.; Versveld, and R.A.; Chapman. 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.;, A.; Carrasco, and N.; Asquith. 2009. Attempts to determine the effects of forest cover on stream flow by direct hydrological measurements in Los Negros, Bolivia. *Forest Ecology and Management* 258:1881-1888.

Lechuga, Claudia. 2009. Zapalinamé: connecting cities and watersheds in Mexico. http://www.ecosystemmarketplace.com/pages/dynamic/article.page.php?page_id=6612&section=home.

Lee, Matilda;. 2012. Coffee farmers in Peru look to carbon market to fund climate adaptation *The Ecologist*.

Leshan, J.;, L.; Xiaoyun, and Z. Ting. 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. Beijing: China Agricultural University.

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, Xenia;, Marina; Ogier, Carlos; Pérez, and Manuel; Martínez. 2006. Elementos metodológicos para la implementación de pagos por servicios ambientales hídricos al nivel municipal en Centroamérica Tegucigalpa: Programa para la Agricultura Sostenible en Laderas de América Central PASOLAC.

Mayer, Audrey L. , and Päivi M.; Tikka. 2006. Biodiversity conservation incentive programs for privately owned forests. *Environmental Science & Policy* 9 (7-8):614-625.

Montagnini, Florencia;, and Christopher; Finney. 2010. Payments for environmental services in Latin America as a tool for restoration and rural development. *AMBIO: A Journal of the Human Environment*

39.

Muñoz-Piña, Carlos;, Alejandro; Guevara, Juan Manuel; Torres, and Josefina; Braña Varela. 2008. Paying for the hydrological services of Mexico's forests: analysis, negotiations and results. *Ecological Economics*:725-736.

Muñoz-Piña, Carlos;, Marisol; Rivera, Alfredo; Cisneros, and Helena García. 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.;, F.; Holwerda, M.; Gomez-Cardenas, M.; Equihua, H.; Asbjornsen, L.A.; Bruijnzeel, L.; Marin-Castro, and C.; Tobon. 2010. Water balances of old-growth and regenerating montane cloud forests in central Veracruz, Mexico. *Journal of Hydrology* Draft (Special Issue: Tropical Hydrology).

Muñoz, C.;, A.; Guevara, J.M.; Bulas, J.M.; Torres, and J.; Braña Varela. 2005. Paying for the hydrological services of Mexico's forests.

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

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. In *Programa para la Agricultura Sostenible en Laderas de América Central*. Tegucigalpa: PASOLAC, SDC, COSUDE.

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

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

Pagiola, Stefano; , Augustin; Arcenas, and Gunars Platais. 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.

Pascual, Unai;, Roldan; Muradian, Luis C.; Rodríguez, and Anantha Duraiappah. 2009. Revisiting the relationship between equity and efficiency in Payments for Environmental Services. In *Ecosystem Services Economics (ESE): Working Paper Series*: The United Nations Environment Programme: Division of Environmental Policy Implementation.

Pattanayak, Subhrendu K; , Sven; Wunder, and Paul; Ferraro. 2010. Show me the money: do payments supply environmental services in developing countries? *Review of Environmental Economics and Policy*:1-21.

Perrot-Maitre, D.;. 2006. The Vittel payments for ecosystem services: a “perfect” PES case? London, UK: International Institute for Environment and Development.

Pfaff, A.;, J.; Robalino, and G.A.; Sánchez-Azofeifa. 2008. Payments for Environmental Services: Empirical analysis for Costa Rica. In *Working Papers Series* Terry Stanford Institute of Public Policy Duke.

Porras, Ina. 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, Ina;. 2010. Fair and green? The social impacts of payments for environmental services in Costa Rica. London, UK: International Institute for Environment and Development.

Porras, Ina; , Maryanne; Grieg-Gran, and Nanete; Neves. 2008. *All that glitters: A review of payments for watershed services in developing countries*. Vol. 11, *Natural Resource Issues*: International Institute for Environment and Development (IIED).

Porras, Ina;, Miriam; Miranda, David N.; Barton, and Adriana; Chacón-Cascante. 2012. De Rio a Rio+: Lecciones de 20 años de experiencia en servicios ambientales en Costa Rica. In *Shaping Sustainable Markets*. Londres: International Institute for Environment and Development.

Postel, Sandra;, and Barton H Jr.; Thompson. 2005. Watershed protection: capturing the benefits of nature's water supply services. *Natural Resources Forum* 29:98-108.

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

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

Reis, N.; , G.; Sydness, and D. Barton. 2007. Feasibility of Payments for Watershed Services in India and Asia. Norwegian Institute for Water Research.

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

Robalino, Juan;, Alexander; Pfaff, and Laura; Villalobos. 2011. Assessing the impact of institutional design of payments for environmental services: the Costa Rican experience. In *Ecosystem services from Agriculture and Agroforestry: Measurement and Payments*, edited by B. Rapidel, F. DeClerk, J. LeCoq and J. Beer: Earthscan Press.

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

Rojas, Manrique, and Bruce Aylward. 2003. What are we Learning from Experiences with Markets for Environmental Services in Costa Rica? A Review and Critique of the Literature. London: International Institute for Environment and Development.

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

Ross, M., B. Depro, and S. Pattanayak. 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.

Ruhweza, A.;, and S. Waage. *The state of play: payments for ecosystem services in East and Southern Africa*. The Katoomba Group 2007. Available from http://www.ecosystemmarketplace.com/pages/dynamic/article.page.php?page_id=5108&section=home&eod=1.

Saberwal, Vasant K. . 1998. Science and the desiccationist discourse of the 20th Century. *Environment and History* 3:309-343.

Salas, J.;. 2004. Case study of the Maasin Watershed: analyzing the role of institutions in a watershed-use conflict. In *Developing Mechanisms for Rewarding the Upland Poor in Asia for Environmental Services They Provide*: RUPES, World Agroforestry Centre.

Sánchez-Azofeifa, Arturo;, Alexander; Pfaff, Juan; Robalino, and Judson; Boomhower. 2007. Costa Rica’s Payment for Environmental Services Program: intention, implementation, and impact. *Conservation Biology* 21 (5):1165-1173.

Sen, A.;. 1996. The political economy of targeting. In *Public spending and the poor: theory and evidence*, edited by K. N. D. van de Walle. Baltimore: The John Hopkins University Press.

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

Sierra, ; , and ; Russman. 2006. On the efficiency of environmental service payments: A forest conservation assessment in the Osa Peninsula, Costa Rica. *ECOLOGICAL ECONOMICS* 59.

Skjeggedal, Terje;, Vegard; Gundersen, Kjell A.; Harvold, and Odd Inge; Vistad. 2010. Frivillig vern av skog-evaluering av arbeidsformen (Voluntary conservation of forest-evaluation of working procedure). In *Samarbeidsrapport*: NIBR/NINA 2010.

Skutsch, Margaret M.;, Patrick E; van Laake, Eliakimu M.; Zahabu, Bhaskar S. Karky, and Pushkin; Phartiyal. 2009. Community monitoring in REDD+. In *Realising REDD+: national strategy and policy options*, edited by A. w. B. Angelsen, M., Kanninen, M., Sills, E., Sunderlin, W. D. and Wertz-Kanounnikoff, S. Bogor, Indonesia: CIFOR.

Southgate, D.;, and S.; Wunder. 2007. Paying for watershed services in Latin America: a review of current initiatives. In *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.

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

Stem, C.;. 2005. TNC and partner experiences with watershed valuation activities in the State of Chiapas, Mexico. In *Final Report*: The Nature Conservancy; USAID.

Sun, Changjin.;, and Liqiao; Chen. 2006. A study of policies and legislation affecting payments for watershed services in China. In *Developing markets for watershed protection services and improved livelihoods*. London: Research Center of Ecological and Environmental Economics Beijing, and International Institute for Environment and Development.

Suyanto. 2010. Site Profile: RUPES Sumberjaya. In *RUPES Program*. Bogor: ICRAF.

Svarstad, H.;, A.; Sletten, R.; Paloniemi, D; Barton, and M.; Grieg-Gran. 2011. Three types of environmental justice: from concepts to empirical studies of social impacts of policy instruments for conservation of biodiversity. In *POLICYMIX - Assessing the role of economic instruments in policy mixes for biodiversity conservation and ecosystem services provision*, edited by K. M. K. Tingstad.

Swallow, Brent.;, Mikkel; Kallesoe, Usman; Iftikhar, Meine; Van Noordwijk, Carina; Bracer, Sara; Scherr, K.V.; Raju, Susan; Poats, Anantha; Kumar Duraiappah, Benson; Ochieng, Hein; Mallee, and Rachael; Rumley. 2009. Compensation and rewards for environmental services in the developing world: framing pantropical analysis and comparison. *Ecology and Society* 14 (2).

Swallow, Brent.;, Beria; Leimona, Thomas.; Yatich, Sandra.; Velarde, and S. Puttaswamaiah. 2007. The conditions for effective mechanisms of compensation and rewards for environmental services. In *CES Scoping Study issue Paper no.3*. Nairobi, Kenya:: World Agroforestry Centre.

TEEB (The Economics of Ecosystems and Biodiversity). 2011. Rewarding benefits through payments and markets; chapter 5. In *TEEB in National and International Policy Making*, edited by P. ten Brink. London: Earthscan.

TEEBcase Porras, I. 2010. The Case of La Esperanza Hydroelectric Power Company, Costa Rica. In *PES as a strategy to minimize risk*, edited by TEEB.

Tikka, Päivi;. 2003. Conservation contracts in habitat protection in southern Finland. *Environmental Science & Policy* 6:271–278.

Turpie, J. K., C. Marais, and J. N. Blignaut. 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):788-798.

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

Villanueva, Cristóbal;, Muhammad; Ibrahim, Kelvin; Torres, and Marco Torres. 2008. Planificación agroecológica de fincas ganaderas: La experiencia de la subcuenca Copán, Honduras. In *Serie Técnica.* , edited by D. d. I. y. Desarrollo. Turrialba: Centro Agronómico Tropical de Investigación y Enseñanza (CATIE).

Wätzold, Frank, Melanie Mewes, Rob van Apeldoorn, Riku Varjopuro, Tadeusz Chmielewski, Frank Veeneklaas, and Marja-Leena Kosola. 2010. Cost-effectiveness of managing Natura 2000 sites: an exploratory study for Finland, Germany, the Netherlands and Poland. *Biodiversity and Conservation* 19 (7):2053-2069.

White, Douglas, and Peter Minang. 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. In *CIFOR Occasional Paper* CIFOR.

Wunder, Sven. 2005. Payments for Environmental Services: Some Nuts and Bolts. In *CIFOR Occasional Paper No. 42*. Jakarta, Indonesia: Center for International Forestry Research.

Repeated Author. 2005. Payments for environmental services: some nuts and bolts. Occasional Paper No. 42. Bogor, CIFOR.

Repeated Author. 2008. Payments for environmental services and the poor: concepts and preliminary evidence. *Environment and Development Economics* 13 (3):279-297.

Wunder, Sven, and Montserrat Albán. 2008. Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. *Ecological Economics* 65 (4):685-698.

Wunder, Sven, Stefanie; Engel, and Stefano; Pagiola. 2008. Taking stock: A comparative analysis of payments for environmental services programs in developed and developing countries. *Ecological Economics* 65:834-852.

Wünscher, Tobias, Stefanie Engel, and Sven Wunder. 2008. Spatial targeting of payments for environmental services: A tool for boosting conservation benefits. *Ecological Economics* 65 (4):822-833.

Xu, Jintao;, Ran; Tao, Zhigang; Xu, and Michael T.; Bennett. 2010. China's Sloping Land Conversion Program: does expansion equal success? *Land Economics* 86 (2):219-244.

Xu, Z.;, M.; Bennett, R.; Tao, and J.; Xu. 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 (3-4):317-326.