Analysis of incentive contracts for sustainable ecosystem management in the buffer zone of Podocarpus National Park, Ecuador

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Abstract (English)

Payments for environmental services (PES) schemes have been proposed in order to internalize the costs and benefits of nature conservation and ecosystem service provision, and as such create more positive incentives to landholders to safeguard certain ecosystems and the benefits they provide to people. In addition, PES are often considered as a tool to improve rural household income. However, there has been little empirical verification to what extent the two objectives of environmental conservation/restoration and rural development can be achieved simultaneously through PES.

The aim of this PhD research is to analyse the (potential) impacts and trade-offs of PES contracts for the maintenance and improvement of ecosystem services and the provision of rural income to households in the buffer zone of Podocarpus National Park in Ecuador. As a second objective the use of these contracts is compared with other incentive schemes and with land uses implemented in the research area to increase understanding of the impacts.

PES schemes, as analysed in this PhD, cover a range of arrangements, from small markets to (almost) complete hierarchical organisational arrangements. Our first analysis suggests that over time an increasing number of schemes have incorporated characteristics of a hierarchy to organise the users' side. However, contractual agreements based on bilateral negotiation of payments or payments set by intermediaries, with mainly individual and communal landholders as providers, remain the core focus of most of the schemes analysed. Intermediaries are important actors in almost all schemes, mainly organising and/or representing ecosystem service users, and are most often national or local governments.

The results of our second analysis, on the cost-effectiveness of implementing several programmes with the dual goal of improving conservation and rural livelihoods, show that purchasing land as done by the water fund of the research area has the highest additionality in ecosystem service provision. The PES scheme studied (Socio Bosque) is the most cost-effective both for current as for increased ecosystem service provision and for extra rural job creation. Finally, we found that organic coffee certification has the highest positive impact on rural income creation.

In our third analysis we applied a portfolio analysis to study the impact of combining milk production with Socio Bosque contracts for forest conservation and the restoration of pastures to its natural vegetation, and an incentive programme for tree plantation. The results of this analysis suggest that most farmers would increase the area under conservation and/or restoration in a risk reduction strategy (minimum variance portfolio) and when maximizing profit per unit increase of risk (optimal portfolio), beyond what would be expected from decisions considering expected return only. However, none of the portfolios would increase all households' income to above the poverty line due to small farm sizes and low payments. Moreover, a negative impact on household income was observed from adopting the land-use portfolios for milk producers without forest. For most producers without forest, there seemed to be a trade-off between

maximizing household income on the one hand, and risk reduction through a land-use combination of restoration payments and tree plantations, on the other.

In our final analytical chapter we conclude that farmers in the area are interested in contracts for silvopastoral systems, but differ in their preferences regarding the additional requirements. The results suggest that farmland area, agricultural income, the share of this income in total income, and landowners' perception of environmental problems provide a partial explanation for the heterogeneity observed in the preferences for specific contracts. Offering flexible contracts with varying additional requirements within the same scheme may contribute to enhance participation, and thus improve the provision of ecosystem services. Building of trust and combining PES with integrated conservation and development projects may further increase participation and thus the adoption silvopastoral systems within the research area.

Overall we see a trade-off between maximizing conservation and improvement of ecosystem services and increasing household income in the PES schemes analysed. Within the research area rural poverty seems best to be addressed by the implementation of projects that aim at increasing productivity. These could be linked with environmentally friendly practices to ensure a certain level of ecosystem service provision. PES could only have that impact if payments would be increased substantially for poorer households. However, PES as a positive incentive (or reward) for forest conservation can provide an additional and stable income source of income for forest holders. For non-forest holders PES for productive actions could also provide an additional source of income, and could strengthen the assurance that the new land use practices continue to be implemented for the duration of the contract.

To improve their impact the broader institutional, economic and social environment should be considered. PES is only one tool available to governments, conservation practitioners, and other actors interested in improving and sustaining ecosystem service provision. PES is influenced by a wide range of factors such as market prices for agricultural products and the national and local political situation. Within the research area, PES could be implemented as part of a policy mix and of different programmes to achieve the best outcome both in terms of ecosystem provision and rural development.

Abstract (Nederlands)

Schema’s voor het organiseren van betalingen voor ecosysteemdiensten (PES, naar zijn Engelse afkorting) worden voorgesteld als een systeem om de kosten en baten van natuurbehoud en het leveren van ecosysteemdiensten te internaliseren. Deze schema`s zouden een sterkere economische drijfveer kunnen vormen die grondbezitters aanzet om bepaalde ecosystemen, en de baten die deze bieden aan mensen, te beschermen. Daarnaast worden PES vaak beschouwd als een instrument om het inkomen van rurale gezinnen te verbeteren. Er zijn echter relatief weinig empirische studies over de vraag in hoeverre twee doelstellingen van milieubehoud, namelijk restauratie en plattelandsontwikkeling, tegelijkertijd kunnen worden verwezenlijkt door middel van PES.

Het eerste doel van dit doctoraatsonderzoek is een analyse van de (potentiële) effecten en de trade-offs van PES-contracten in de bufferzone van Podocarpus Nationaal Park in Ecuador uit te voeren. De effecten en trade-offs worden onderzocht in functie van het behoud en de verbetering van ecosysteemdiensten, en van het inkomen voor gezinnen. Ook wordt het gebruik van deze contracten vergeleken met enerzijds andere vormen van betaling of premies die natuurbehoud en het leveren van ecosysteemdiensten moeten stimuleren, en met anderzijds typisch landgebruik in het onderzoeksgebied. Het vergelijken van PES met ander landgebruik en programma’s heeft tot doel inzicht te verkrijgen in de effecten van PES ten opzichte van andere maatregelen.

PES schema´s, zoals deze werden geanalyseerd in dit doctoraat, bestrijken een scala aan organisatievormen, van kleine markten tot (bijna) volledige hiërarchische bestuursvormen. Uit onze eerste analyse blijkt dat de schema´s die zijn bestudeerd met de tijd steeds meer gekenmerkt worden door een hiërarchie om gebruikers van ecosysteemdiensten te organiseren. Maar wederzijdse overeenkomsten op basis van bilaterale onderhandelingen over de betalingen, of betalingen bepaald door tussenpersonen, met voornamelijk individuele en gemeenschappelijke grondbezitters als aanbieders van ecosysteemdiensten, blijven de kern van de meeste van de geanalyseerde schema`s. Tussenpersonen zijn belangrijke actoren in bijna alle schema's, vooral voor het organiseren en/of vertegenwoordigen van gebruikers van ecosysteemdiensten,. Het zijn meestal nationale of lokale overheden.

Het tweede object van het doctoraatsonderzoek is een analyse van de effectiviteit in termen van kosten van een aantal programma’s die als tweevoudig doel hebben het verbeteren van natuurbehoud en het verhogen van rurale inkomens. De resultaten tonen aan dat de aankoop van grond, zoals dit werd gedaan door het water-fonds in het onderzoeksgebied, de grootste additionaliteit heeft als het gaat over het leveren van ecosysteemdiensten. Het PES schema dat werd onderzocht (Socio Bosque) is het meest kosteneffectief, voor de huidige zowel als voor de verhoogde levering van ecosysteemdiensten, alsook voor extra tewerkstelling in het onderzoeksgebied. Tot slot wordt aangetoond dat biologische koffiecertificering de hoogste positieve impact heeft op het creëren van het ruraal inkomen.

Het derde object van het doctoraatsonderzoek is een portfolio-analyse. Hierin werd de impact gemeten van het combineren van melkproductie met Socio Bosque contracten voor bosbehoud, het herstel van weiden naar hun natuurlijke vegetatie, en een programma dat premies geeft voor houtplantages. Uit de resultaten van deze analyse blijkt dat de meeste boeren in een risicobeperkingsstrategie (minimum variantie portefeuille) en bij het maximaliseren van de winst per eenheid van risico (optimale portefeuille), het gebied onder behoud en/of restauratie sterker zouden verhogen, dan wat zou worden verwacht van beslissingen die enkel kijken naar de hoogste verwachte opbrengst. Geen van de portefeuilles van landgebruiken slaagt er echter in om de inkomens van alle huishoudens tot boven de armoedegrens te verhogen. Dit is het gevolg van de vaak kleine boerderijen en van te lage betalingen. Bovendien werd er een een negatief effect op inkomen waargenomen voor huishoudens die geen bos bezitten, als deze groep de bekomen portefeuilles van landgebruiken zou toepassen. Voor de meeste melkproducenten zonder bos blijkt er een trade-off te zijn tussen het maximaliseren van het gezinsinkomen aan de ene kant, en het verminderen van inkomensrisico's door middel van combinaties van landgebruiken met ecosysteemrestauratie en boomplantages, aan de andere kant.

In het laatste analytische hoofdstuk wordt vastgesteld dat de boeren in het onderzoeksgebied interesse tonen voor contracten voor silvopastorale systemen. De ondervraagde boeren verschillen echter in hun voorkeuren met betrekking tot aanvullende milieueisen. De resultaten suggereren dat het areaal landbouwgrond, het landbouwinkomen, het aandeel van deze inkomsten in de totale gezinsinkomsten, en de perceptie van milieuproblemen een gedeeltelijke verklaring bieden voor de waargenomen heterogeniteit in de voorkeuren voor specifieke contracten. Het aanbieden van flexibele contracten met verschillende aanvullende eisen binnen eenzelfde schema zou dus een deelname in een PES schema kunnen verhogen en bijgevolg de levering van ecosysteemdiensten kunnen verbeteren. Het versterken van het vertrouwen en het combineren van PES met geïntegreerde projecten voor ontwikkeling en milieubehoud zouden deelname nog kunnen verhogen en daarmee dus de introductie van silvopastorale systemen binnen het onderzoeksgebied.

In het algemeen zien we in dit onderzoek een trade-off tussen het maximaliseren van het behoud en verbetering van ecosysteemdiensten en het verhogen van gezinsinkomens in de geanalyseerde PES schema's. Binnen het onderzoeksgebied lijkt rurale armoede het best te worden aangepakt door de uitvoering van projecten die gericht zijn op het verhogen van de landbouwproductiviteit. Deze kunnen worden gekoppeld aan milieuvriendelijke landbouwpraktijken om een bepaald niveau in de voorziening van ecosysteemdiensten te verzekeren. PES kan alleen een sterke invloed hebben op armoedevermindering indien de betalingen voor armere huishoudens substantieel zouden worden verhoogd. PES als een positieve stimulans (of vergoeding) voor bosbehoud, kan echter zorgen voor een extra en stabiele bron van inkomsten voor boseigenaars. Voor landeigenaars zonder bos, kan PES voor milieuvriendelijke productiesystemen ook zorgen voor een extra bron van inkomsten, en kan het meer zekerheid bieden dat de nieuwe landgebruiken uitgevoerd worden, op zijn minst voor de duur van het PES contract.

Ter verbetering van de impact moet er rekening worden gehouden met de bredere institutionele, economische en sociale omgeving. PES is slechts één van de instrumenten die beschikbaar zijn voor overheden, natuurbeschermingsorganisaties, en andere actoren die geïnteresseerd zijn in het verbeteren en onderhouden van de levering ecosysteemdiensten. PES wordt beïnvloed door een breed scala van factoren, zoals de marktprijzen voor landbouwproducten en de nationale en lokale politieke situatie. Binnen het onderzoeksgebied kan PES worden toegepast als onderdeel van een beleidsmix en van verschillende programma's om het beste resultaat te behalen, zowel in termen van ecosysteemdiensten als van rurale ontwikkeling.

Abstrakt (Deutsch)

Zur Internalisierung der Kosten und Nutzen von Maßnahmen des Naturschutzes und der Bereitstellung von Ökosystemleistungen wurde das Instrument der sog. Zahlungen für Ökosystemleistungen (engl. Payments for Environmental Services, PES) entwickelt. PES sollen Landnutzern positive Anreize für die Bereitstellung von Ökosystemleistungen setzen, dies dadurch befördern und somit einen Beitrag für das menschliche Wohlbefinden leisten. Darüber hinaus werden PES oft auch als Instrumente zur Verbesserung der Haushaltseinkommen ländlicher Bevölkerungsgruppen betrachtet. Bisher gibt es jedoch kaum empirische Erkenntnisse darüber, inwieweit diese beiden Ziele – Naturschutz und ländliche Entwicklung – gleichzeitig durch PES erreicht werden können.

Das Ziel dieser Doktorarbeit ist es daher, die (potenziellen) Auswirkungen und Zielkonflikte der Umsetzung von PES für die Bereitstellung von Ökosystemleistungen und die Verbesserung der Einkommen privater Haushalte, in der Pufferzone des Podocarpus-Nationalpark in Ecuador, zu analysieren. Darüber hinaus wird der Einsatz von PES mit anderen Anreizsystemen und Formen der Landnutzung im Untersuchungsgebiet verglichen, um ein besseres Verständnis ihrer Auswirkungen zu ermöglichen.

Die PES, die in dieser Doktorarbeit untersucht werden, weisen eine große institutionelle Spannbreite auf. Sie reicht von kleinen Märkten bis hin zu (fast) vollständig hierarchisch ausgestalteten Institutionen. Die Ergebnisse des ersten Teils der Analyse legt nahe, dass im Laufe der Zeit eine zunehmende Anzahl der untersuchten PES auf Nutzerseite Merkmale hierarchisch organisierter Institutionen aufweisen. Den Fokus der hier analysierten PES bilden jedoch bilaterale vertragliche Vereinbarungen sowie von Intermediären festgelegte Zahlungsvereinbarungen mit individuellen oder kollektiven Grundbesitzern auf der Anbieterseite. Die Ergebnisse zeigen, dass Intermediäre wichtige Akteure in fast allen hier untersuchten Systemen sind. Sie vertreten die Nutzer der bereitgestellten Ökosystemleistungen und übernehmen die Organisation. Meist sind die Intermediäre lokale oder nationale staatliche Institutionen.

Der zweite Teil der Analyse untersucht die Kostenwirksamkeit der Durchführung verschiedener Programme, die sowohl das Ziel der Bereitstellung von Ökosystemleistungen als auch die Verbesserung der ländlichen Lebensgrundlagen verfolgen. Die Ergebnisse im Untersuchungsgebiet zeigen, dass der Erwerb von Grundeigentum durch einen Wasserfonds imVergleich zu den anderen untersuchten Programmen, das höchste Maß an „Zusätzlichkeit“ in

der Bereitstellung von Ökosystemleistungen aufweist. Das untersuchte PES (Socio Bosque) stellt sich hingegen als die kostenwirksamste Maßnahme sowohl im Hinblick auf die Bereitstellung von Ökosystemleistungen als auch die Schaffung zusätzlicher Arbeitsplätze im ländlichen Raum heraus. Die Zertifizierung von biologisch angebautem Kaffee wiederum erweist sich als die Maßnahme, die das höchste Maß an positiven Auswirkungen auf die Einkommensentwicklung in ländlichen Räumen erreichte.

Im dritten analytischen Teil der Arbeit wurde eine Portfolio-Analyse angewendet, um die Auswirkungen einer Kombination verschiedener Einkommensmöglichkeiten zu untersuchen. Hierzu wurde Milchproduktion mit Verträgen des Socio Bosque Programms zum Waldschutz und der Renaturierung von Weideflächen und einem Anreizprogramm zur Aufforstung kombiniert. Die Ergebnisse dieser Analyse legen den Schluss nahe, dass die meisten Landnutzer, die geschützte oder renaturierte Fläche im Rahmen einer Risikobegrenzungsstrategie (Minimum-Varianz-Portfolio) und einer Gewinnmaximierung pro Risikoerhöhungseinheit (optimales Portfolio) erhöhen würden. Dies würde sogar das Ausmaß übersteigen, das erwartet werden könnte wenn allein der erwartete Profit als Entscheidungsgrundlage zu Grunde gelegt würde. Aufgrund der geringen bewirtschafteten Fläche und der niedrigen Zahlungen ist es in keinem Portfolio möglich das Einkommen soweit zu erhöhen, dass dies zu einer Überschreitung der Armutsgrenze führt. Darüber hinaus konnten in den Landnutzungsportfolios für Milcherzeugung ohne Waldflächen negative Auswirkungen auf das Haushaltseinkommen beobachtet werden. Für die meisten (Milch)Produzenten ohne Waldfläche, schien es einen Zielkonflikt zu geben zwischen der Maximierung des Einkommens auf der einen Seite und einer Risikoreduzierung durch eine Kombination mit Zahlungen für Renaturierung und Aufforstung auf der anderen Seite.

Das letzte analytische Kapitel legt den Schluss nahe, dass die Landwirte im Untersuchungsgebiet zwar ein Interesse an Verträgen für silvopastorale Systeme haben, sich aber in ihren Präferenzen hinsichtlich zusätzlicher Vertragsanforderungen unterscheiden. Die Ergebnisse deuten darauf hin, dass die Größe der bewirtschafteten Fläche, das Einkommen aus der Landwirtschaft, der Anteil dieser Einnahmen am Gesamteinkommen und die Wahrnehmung von Umweltproblemen eine Erklärung für die beobachtete Heterogenität der Präferenzen für unterschiedliche Vertragsinhalte darstellt. Das Anbieten flexibler Verträge mit unterschiedlichen Bedingungen könnte daher dazu beitragen, die Beteiligung der Landwirte zu erhöhen und damit letztlich zu einer verbesserten Bereitstellung von Ökosystemleistungen führen. Darüber hinaus könnte das Schaffen von

Vertrauen und die Kombination von PES mit integrierten Erhaltungs- und Entwicklungsprojekten zudem die Teilnahme und damit die Übernahme / Einführung silvopastoraler Systeme weiter erhöhen.

Insgesamt lässt sich in den analysierten PES ein Zielkonflikt beobachten zwischen der Maximierung der Bereitstellung und qualitativen Verbesserung von Ökosystemleistungen auf der einen Seite und der Erhöhung des Haushaltseinkommens auf der anderen Seite. Im Untersuchungsgebiet lässt sich der ländlichen Armut am besten durch Projekte begegnen, deren Fokus auf der Steigerung der Produktivität liegt. Diese könnten mit umweltfreundlichen Praktiken verknüpft werden, um ein gewisses Maß an bereitgestellten Ökosystemleistungen zu gewährleisten. PES könnten dieses Ergebnis nur erreichen, wenn die Zahlungen für ärmere Haushalte wesentlich erhöht würden. PES-Zahlungen als positiver Anreiz (oder Honorierung) für Walderhalt können jedoch eine zusätzliche und stabile Einkommensquelle für Waldbesitzer darstellen. Für Landnutzer ohne Waldflächen könnte PES für Produktivsysteme eine zusätzliche Einnahmequelle darstellen, und eine Absicherung dafür bieten, dass neue Landnutzungspraktiken während der Vertragslaufzeit durchgeführt werden.

Zur Verbesserung der Auswirkungen von PES sollte der breitere institutionelle, wirtschaftliche und soziale Kontext berücksichtigt werden. Denn PES stellen lediglich ein Instrument unter vielen dar, das staatlichen Institutionen, Praktikern im Naturschutz und anderen an der Bereitstellung von Ökosystemleitungen Interessierten Akteuren zur Verfügung steht. PES werden durch eine Vielzahl von Faktoren, wie z.B. Marktpreise für Agrarerzeugnisse und die nationale und lokale politische Ausgangslage beeinflusst. Im Untersuchungsgebiet könnte PES als Teil eines umfassenderen Politik-Mixes und verschiedener Programme umgesetzt werden, um das beste Ergebnis sowohl im Hinblick auf die Bereitstellung von Ökosystemleistungen als auch die Entwicklung des ländlichen Raums zu erreichen.

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Acronyms

BAU: Business-As-Usual

CE: Choice Experiment

CLM: Conditional Logit Model

EMAAL-EP: Municipal Public Company for Water Supply and Sewerage of Loja

ERWI: Extra Rural Work Indicator

ES: Ecosystem Services

ESI: Ecosystem Service Indicator

FORAGUA: Regional Water Fund

HIPI: Household Income and Participation Indicator

IAD: Institutional Analysis and Development

ICDP: Integrated Conservation and Development Project

ICERI: Increased Conservation and Ecosystem Regeneration Indicator

LCM: Latent Class Model

MA: Millennium Ecosystem Assessment

NGO: Non-Governamental Organisation

NK-CAP: Noel Kempff Mercado Climate Action Project

PES: Payments for Environmental Services

SES: Socio-Ecological Systems

TC: Transaction Costs

TEEB: The Economics of Ecosystems and Biodiversity

Chapter 1: Introduction

1.1. Problem statement

1.1.1. Concept and definitions

Ecosystem services (ES) are “the benefits people obtain from ecosystems”, and include a wide range of services (MA, 2005a). These services are classified by the Millennium Ecosystem Assessment into four categories, whereby the first three categories directly impact human well-being, while the fourth category has an impact by supporting the other three categories. Table 1.1 summarizes the four categories.

Table 1.1: Ecosystem Service Categories

ES Category	Examples
Provisioning Services	Food, timber
Regulating Services	Water quality regulation, climate regulation
Cultural Services	Cultural heritage, recreation
Supporting Services (or Processes)	Nutrient cycling, soil formation

Source: Based on the Millennium Ecosystem Assessment (MA, 2005a)

Biodiversity, referring to “the number, abundance, and composition of the genotypes, populations, species, functional types, communities, and landscape units in a given system” (Dıaz et al., 2005, p. 300), can be understood both as an ecosystem service, and – similar to regulating services – as underpinning ecosystem service supply (Díaz et al., 2006; Harrison et al., 2014; MA, 2005a). The difference between biodiversity and regulating services is that humans can directly impact biodiversity, and thus the supply of (other) ecosystem services. In most cases the relationship between biodiversity and the provision of ecosystem services is positive, especially when multiple ecosystem services are considered (Balvanera et al., 2006; Harrison et al., 2014; Hooper et al., 2005; Lavorel and Grigulis, 2012). Such a positive relationship has been reported for provisioning services, such as for fisheries (Heithaus et al., 2008; Ibelings et al., 2007; Palumbi et al., 2008; Rosenberg et al., 2000); regulating services such as pollination (Balvanera et al., 2005; Blanche and Cunningham, 2005; Hoehn et al., 2008; Klein et al., 2007), carbon storage (Balvanera et al., 2005; Conti and Díaz, 2013; Hatanaka et al., 2011), and water quality regulation and water flow regulation (Brauman et al., 2007; Buytaert et al., 2007; Hefting et al., 2003; Makarieva et al., 2006); and for cultural services, such as landscape beauty (Harrison et al., 2014; van den Berg et al., 1998).

Healthy, biodiverse ecosystems and the ES these systems provide are presumed to be important for human well-being (Chavas, 2009; Raudsepp-Hearne et al., 2010). The ability of ecosystems to provide goods and services to humans is seriously affected through human-inflicted damage to the environment (MA, 2005a). Important underlying causes of the degradation of ecosystems and ES are first that not all benefits they provide to people are taken into account in land-use decisions, and second, the public good characteristics of many ES, specifically those that provide indirect benefits (Farley, 2008; Kroeger and Casey, 2007; Norgaard, 2010; Pearce, 2007; Swallow et al., 2009; Tacconi, 2000).

The continuous supply of many ES is in most cases guaranteed through public sector provision (Libecap, 2005; OECD, 2004). The public good nature of ES has been used as a justification for (inter)governmental collective action and direct government regulation of resource use, such as through the establishment of protected areas (OECD, 2004; Swallow and Meinzen-Dick, 2009).

Conversely, many governments - and other actors involved in conservation and sustainable ecosystem management - have implemented an increasing number of projects and programmes that provide positive incentives to private actors to stop ecosystem degradation. This is done by translating external, non-market values of the environment into financial and/or non-monetary incentives to adopt, maintain or reinforce land uses that provide ES (Engel et al., 2008; Gauvin et al., 2010; Morse et al., 2013; Turner and Daily, 2008; Wunder, 2005). The delivery of incentives should make conservation and ES provision more rewarding compared to alternative land uses, and thus more interesting for landholders (Grieg-Gran et al., 2005). These incentives are often presented as effective mechanisms for the conservation and improvement of ES supply when existing laws, or other regulations and directives, are misunderstood, defied or not enforced (Wunder, 2007). In contrast to explicit government regulation, such direct monetary (or in-kind) incentives can supplement households' income instead of solely restricting people. These incentives are also often considered an adequate alternative to coercive or prescriptive laws to achieve environmental outcomes (Jack et al., 2008; Pirard, 2012a).

Payments for Environmental Service[1] (PES) schemes provide such financial incentives to conserve or change specific land uses. They have been increasingly implemented over the last two decades (Engel et al., 2008; Le Coq et al., 2011). According to Matzdorf et al. (2013), Muradian and Rival (2012) and Vatn (2015) amongst others, PES can be seen as hybrid[2] governance structures for the management of ES (Chapter 3 discusses PES as hybrid governance structures in more detail). These authors add that "core elements [of PES] are payments for good stewardship of well-defined ES (including payments for activities thought to yield well-defined ES) and where payments are aimed to be made conditional on goal achievement or implementation of the activity" (Matzdorf et al., 2013, p. 59). PES schemes offer financial or in-kind payments to landholders who are potential ES providers. These landholders in turn can agree to participate by setting aside farmland, conserving or regenerating ecosystems, or adopting specific farming techniques or new technology (Gauvin et al., 2010).

[1] Throughout this PhD we use the terms ecosystem services and environmental services, and the use of these terms in PES, interchangeably. We exclude provisioning services such as agricultural products or timber, as these provisioning services are not the focus of PES. Shelley (2011) and Derissen and Latacz-Lohmann (2013) provide a detailed discussion on the use of these terms.

[2] Hybrid is here understood as the term for a 'combination' or 'mix' of modes of governance (Matzdorf et al., 2013; Muradian and Gómez-Baggethun, 2013; Vatn, 2015), whereas PES a mode of governance is not simply understood as 'a transfer of resources', but as how this transfer (payment) is put into practice (governed) – see also Chapter 3.

Different definitions and attempts to conceptualize PES exist in literature (Schomers and Matzdorf, 2013). Wunder (2005, p. 3) defines PES as "a voluntary transaction where a well-defined environmental service (or a land-use likely to secure that service) is being bought by a (minimum of one) ES buyer from a (minimum of one) ES provider if and only if the ES provider secures ES provision conditionally". This definition is rather market-oriented (Shapiro-Garza, 2013; Vatn, 2015), and a Coasean conceptualization of PES (Sattler et al., 2013). In the Coasean approach on PES, socially optimal levels of ES are obtained through private negotiations in markets, supported by the assignment of property rights (Gomez-Baggethun and Ruiz Perez, 2011; Muradian et al., 2010). However, this conceptualization of PES is seen as too narrow with only a few of the existing schemes complying with the criteria identified in this definition (e.g. Farley and Costanza, 2010; Kinzig et al., 2011; Tacconi, 2012; Vatn, 2010). Muradian et al. (2010, p. 1205) thus proposed that PES can be better defined as "a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land-use decisions with the social interest in the management of natural resources". This definition does not exclude government payments, and can be understood as a Pigouvian vision on PES (Sattler et al., 2013). In 'Pigouvian PES' the existence of externalities is corrected through public intervention with taxes and subsidies (Gomez-Baggethun and Ruiz Perez, 2011). More recently Wunder (2015, p. 8) proposed a revised definition, and defined PES as "voluntary transactions between service users and service providers that are conditional on agreed rules of natural resource management for generating offsite services".

By using financial or in-kind incentives, practices that are socially desirable but privately unprofitable become rewarding to land users. PES offers thus the possibility to change land use decisions, and can as such be used to regulate the provision of ES (Engel et al., 2008; Sommerville et al., 2009). The ES that are most commonly considered in PES schemes are biodiversity, carbon sequestration and storage, watershed protection, and landscape beauty (Muñoz Escobar et al., 2013).

Besides decreasing ecosystem degradation through the provision of positive incentives, the possibility for governments and other actors to tap into potential new sources of financing for conservation and sustainable ecosystem management is one of the promises of PES (Milne and Niesten, 2009). Conservation activities are underfunded almost everywhere, but the gap between current expenditure and what is needed, is particularly extreme in the tropics. Tropical countries contain the highest concentration of animal and plant biodiversity (Lewandrowski et al., 1999; Parker et al., 2012). Securing sufficient and continuous financing for conservation activities is a major problem. Many opportunities for conservation are lost due to inadequate funding, whereas new financing sources need to be continuously identified

(Balmford and Whitten, 2003; Turner et al., 2003). The identification of ES users[3] could potentially be a first step towards new strategies to secure financing for conservation projects. ES users can finance PES to secure ES provision, whereas ES providers receive the payments (Milne and Niesten, 2009).

PES can be understood as a coordination mechanism, in which the key element is a contractual agreement between users of ES or an entity representing them, and individual and/or communal landholders who are expected to provide ES (Kurttila et al., 2006; Milne and Niesten, 2009; van Noordwijk et al., 2007). A coordination mechanism is defined by Muradian and Gómez-Baggethun (2013, p. 1116) as "the means used by social agents to align their activities, in such a way that the potential benefits (for all the parties involved) of concerted actions can be realized" (additionally a coordination mechanism is an institutional arrangement, see Section 1.2.2.).

According to Engel et al. (2008), PES were originally designed and implemented as a tool to increase the efficiency of natural resource management, and not to achieve poverty reduction. However, PES are often considered to have the potential to improve the income of rural households (Barham et al., 2011; Beuchelt and Zeller, 2011; Gauvin et al., 2010; Grieg-Gran et al., 2005; Pagiola et al., 2005a; Porras et al., 2008). This is especially valid for the tropics where areas with high biodiversity, threatened by high rates of ecosystem degradation, often suffer a high incidence of rural poverty (Milder et al., 2010). Hence, it is often aspired that PES can create a win–win situation through the combination of conservation objectives and livelihood improvements (Miles and Kapos, 2008). However, it is not established if these schemes benefit poorer rural households, and/or if a trade-off exists between achieving the highest levels of conservation and ES provision, and improving poverty alleviation (Beuchelt and Zeller, 2011; Grieg-Gran et al., 2005; Milder et al., 2010; Molnar et al., 2004; Pagiola et al., 2005b; Scherr et al., 2004). It has been suggested that PES are rather ineffective in involving poor (individual) land users, because the poor lack access to sufficient resources, such as land, to be devoted to ES provision, and because of higher transaction costs of participation in comparison to large landowners[4] (Albán and Argüello, 2004; Grieg-Gran et al., 2005; Porras, 2010; Swallow and Meinzen-Dick, 2009). However, even when PES schemes are not meant as a tool to reduce poverty, it is acknowledged that PES should not make participants in the scheme worse off than they were before (Corbera et al., 2009; Scheufele and Bennett, 2013).

Against the above background, it remains unclear to what extent the two objectives of environmental conservation and poverty alleviation through income generation can be achieved

[3] Often the term 'ES beneficiaries' is used, instead of ES users. However, the term beneficiary is also used to refer to those who 'benefit' from PES payments, i.e. the provider. In this PhD, similar to Wunder (2015), we thus use the term 'ES user' or simply 'users'.

[4] In the case of communities transaction costs for PES implementers can be lower, but the participating communities still have to organise themselves internally, creating new transaction costs for them. Krause et al. (2013) and Milne and Adams (2012) provide examples of problems that can occur with the organisation of communities to participation in PES schemes.

simultaneously through PES. There has been little empirical verification on how PES affect household income and ES provision at the same time (Engel et al., 2008; Gauvin et al., 2010; Landell-Mills and Porras, 2002; Smith and Scherr, 2002). In order to improve understanding of the impact of PES it is imperative to analyse if existing or potential schemes can improve ES provision, if these schemes impact positively on household income, and if conflicts arise when addressing these two goals concurrently.

Ecuador is an ideal case study area, because there is urgent need to stop ecosystem degradation, while rural poverty is pervasive. At the time of writing this PhD, over 19% of the country is declared as officially protected through a network of conservation areas. However, ecosystem degradation continues both inside and outside protected areas. One way to halt ecosystem degradation is through the implementation of conservation actions in the buffer zones of protected areas. In Ecuador several PES and PES-like schemes have been implemented in these spots. Through national and local governments, and with the support of national and international NGOs, a series of programmes have been realised that aim to conserve and improve the provision of ES, while at the same time contributing to income of rural households (Camacho, 2008; de Koning et al., 2011; Goldman-Benner et al., 2012). Examples are the Socio Bosque Programme for conservation and restoration of forest and *páramos* (Andean grasslands) on private and communal lands (see details in Annex 1), and so-called water funds for watershed conservation (see example in Annex 2). In addition, other programmes such as incentives for timber plantations and projects to implement organic production systems and certification for coffee have been implemented. Hence, Ecuador serves as an excellent case to study the impact of these schemes on conservation and ES provision, as well as to analyse how they may impact household incomes.

More in particular the zone of Podocarpus National Park is selected as the case study area. It hosts several PES and PES-like schemes, and is an important area for ES provision. Households living in the buffer zone are involved in land uses that are not all equally beneficial to ES provision and ecosystem protection, while poverty rates are significant.

In this PhD, the PES (PES-like) schemes we study focus on biodiversity and watershed services. The Convention on Biological Diversity defines biodiversity as: "Biological diversity means the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems” (UN, 1992). Watershed services are ES provided by watershed ecosystems, and include improved water quality and quantity, and regulation of (seasonal) water flows (Brauman et al., 2007; Fisher et al., 2010; Quintero et al., 2009).

1.1.2. Research aim

Against the background presented above, the following research aims are presented, namely to:

(1) analyse the (potential) impacts and trade-offs of PES contracts for the conservation and improvement of ES and the provision of rural income in the buffer zone of Podocarpus National Park in Ecuador; and

(2) compare the PES contracts of the Socio Bosque programme with other programmes that use positive incentives to landholders, and with the current land uses implemented in the research area in order to increase understanding of the impact of PES relative to other systems and land uses.

1.1.3. Study rationale

According to Carpenter et al. (2009) and Ferraro and Pattanayak (2006) it is difficult to find causal evidence of the effectiveness of conservation instruments commonly used in developing countries. Research projects have analysed the impact of PES or other methods such as market incentives (e.g. organic labelling) or command-based conservation actions (e.g. the creation of protected areas). However, only a few studies compare these different approaches (Clements et al., 2013, 2010; Ferraro and Simpson, 2002; Le Coq et al., 2011), although this would improve the impact analyses, as it additionally provides evidence of the impact of PES compared to alternative approaches. In addition also the debate on possible trade-offs or synergies between conservation and rural development of PES is ongoing (Davies et al., 2014; Gauvin et al., 2010; Porras et al., 2008; Rodríguez et al., 2011). It is essential to evaluate both environmental and social outcomes of PES in order to guarantee long-term PES support and improve its effectiveness (Jack et al., 2008).

The trade-offs between conservation and income generation have not yet been established for Ecuador nor were different programmes compared. At the start of this PhD research only a few analyses of the Ecuadorian programmes were available. Noteworthy exceptions are the studies by de Koning et al. (2011) and Farley et al. (2011) on Socio Bosque, while Redondo-Brenes (2009) analysed a water fund implemented in Southern Ecuador. In addition, some studies are available on the potential of implementing carbon payments to farmers in an area adjacent to the research area (Knoke et al., 2009a, 2009b). However, since the start of the PhD research, an increasing number of papers were published focusing on the programmes evaluated. For Socio Bosque, Krause and colleagues analysed legitimacy, safeguards, benefit distribution and equity, and the measurement of biodiversity with a main emphasis on the programme’s community contracts (Krause et al., 2013; Krause and Loft, 2013; Krause and Nielsen, 2014; Krause and Zambonino, 2013). Bremer and colleagues analysed Socio Páramo, a component of Socio Bosque that focuses on the conservation of *páramos* instead of forests. They studied which factors influenced participation in the programme, and the conservation and livelihood outcomes of the programme (Bremer et al., 2014a, 2014b). Some general studies are available on the regional water fund (FORAGUA) that has been operating in the research area (Goldman-Benner et al., 2012; Kauffman, 2013). Yet, these analyses dealt with more than one water fund and did not measure impacts of the conservation/restoration actions implemented.

Within Ecuador, studies have been carried out for some similar PES programmes (Albán and Argüello, 2004; Camacho, 2008; Echavarria et al., 2004; Rodríguez de Francisco et al., 2013; Wunder and Alban, 2008). These studies are, however, mostly descriptive and do not focus on environment-rural income trade-offs. In addition analyses have been carried out to assess payments to incentivize the adoption of coffee agroforestry systems, but not based on existing PES schemes in Ecuador (Benitez et al., 2006; Castro et al., 2013). It are these gaps in research that are addressed in this PhD thesis.

1.2. Framework

1.2.1. SES framework

Many ES have characteristics of public goods and common-pool resources. The Institutional Analysis and Development (IAD) framework, developed by Elinor Ostrom and colleagues, has been used for the analysis of public goods and common-pool resources (not to be confused with common-property regimes) at multiple levels (Ostrom and Cox, 2010; Rudd, 2004), as resource allocation problems arise in the context of common-pool resources and public goods (Lant et al., 2008). This framework is defined by Ostrom (2010a) as "a general language for analysing and testing hypotheses about behaviour in diverse situations at multiple levels of analysis, and studies how rules, physical and material conditions, and attributes of community affect the structure of action arenas, the incentives that individuals face, and the resulting outcomes". A number of PES studies have used the IAD framework (e.g. Fisher et al., 2010; Muñoz Escobar et al., 2013), or an adaptation of this framework (e.g. Corbera et al., 2009; Hejnowicz et al., 2014; Prokofieva and Gorriz, 2013; van Noordwijk et al., 2007).

Based on the IAD framework, Ostrom and colleagues presented a Socio-Ecological Systems (SES) framework (Ostrom, 2009, 2007) (Figure 1.1), which enlarges the IAD framework by incorporating variables that better characterize the ecological dimensions of the system under analysis (Ostrom and Cox, 2010; Schlüter and Madrigal, 2012). The SES framework was originally designed for the analysis of common-pool resources, and has increasingly been used to analyse resource management (Epstein et al., 2013). Subsequently, as socio-ecological systems not only generate common-pool resources, but also public goods, McGinnis and Ostrom (2014) modified the original SES framework to generalize its applications. Yin et al. (2013) used the SES framework to study a large-scale PES programme in China.

The SES framework focuses on the interactions and outcomes that are likely to result from using a specific set of rules for the governance and use of a resource system (Figure 1.1). It shows the relationships between the four subsystems of a SES which interact with each other, as well as the interlinked social, economic, and political settings. These four subsystems are: the attributes of (a) resource system(s); the resource unit(s); the governance system(s); and the actors involved. Each subsystem is composed of multiple, second-level variables (McGinnis and Ostrom, 2014; Ostrom, 2009).

The ***resource system***, such as a designated protected area, is characterized by multiple attributes, such as specific land use systems shaped by farm household behaviour and bio-physical factors (Le et al., 2008; Ostrom, 2009). The resource system itself also has an influence on the land uses adopted. For example, geography (slope, altitude, etc.) has an impact on the crops that farmers can choose to plant. There can be multiple resource systems within a SES (Ostrom and Cox, 2010).

The ***resource units,*** such as trees, wildlife types and water flow, are generated by the resource system (Ostrom, 2009, 2007). These units are used, extracted, from the resource system by resource users. They are thus the users of the SES system, and are for example individuals who extract timber from a forest (McGinnis and Ostrom, 2014; Ostrom, 2007).

Although in its origin the SES framework only included resource users (e.g. Ostrom, 2007), also the behaviour of third parties who do not directly use the resource must be considered (McGinnis and Ostrom, 2014). As a result, the concept was broadened to include diverse ***actors***. They are the 'players of the game' (North, 1990), and have the ability to act upon their interests (Prokofieva and Gorriz, 2013). Groups of actors can be differentiated by the types of activities in which they are involved (McGinnis and Ostrom, 2014). The actors also manage the resource system according to rules and procedures established through a governance system (McGinnis and Ostrom, 2014; Ostrom, 2009, 2007).

The ***governance system*** consists of the organisations and rules that govern the resource, and include government and non-government organisations, as well as the systems of property right and different choice rules' levels (McGinnis and Ostrom, 2014; Ostrom, 2009).

Linked to broader political, economic and social settings and to related ecosystems, these four subsystems interact within action situations and produce specific outcomes, which through feedback affect the subsystems, as well as other SESs (Ostrom, 2009; Ostrom and Cox, 2010).

An ***action arena*** consists of an 'action situation' and the actors. An action situation is the social space where individuals or groups of individuals interact and exchange goods and services, and where outcomes are produced (Ostrom, 2010; Whaley and Weatherhead, 2014). The actors in the action situation refer to theories of the behaviour of actors (Whaley and Weatherhead, 2014). An action arena can be characterized by a group of actors, the positions they take, potential outcomes of different actions, sets of allowable actions for actors in each position, control actors have over actions, actors' information, and costs and benefits of actions and outcomes (Ostrom, 2010; Ostrom and Cox, 2010). Actors make choices based on their preferences, objectives, the costs and benefits assigned to alternative actions and outcomes, and strategic considerations (Rudd, 2004). The IAD and SES frameworks distinguish different choice levels, namely: the operational choice level; the collective choice level; and the constitutional choice level (McGinnis and Ostrom, 2010).

Interactions occur when, within a given set of external ecological, social and institutional constraints, actors consider the costs and benefits of various actions, and act according to the incentives they perceive (McGinnis and Ostrom, 2010; Rudd, 2004). Within the action area actions (e.g. a farmer's production decisions) lead to observable ***outcomes***, such as land uses, that can be evaluated (e.g. hectares of forests cut down) (McGinnis and Ostrom, 2010; Rudd, 2004).

The SES framework thus facilitates starting an analysis of how attributes of the four subsystems "jointly affect and are indirectly affected by interactions and resulting outcomes achieved at a particular time and place" (Ostrom, 2007, p. 15182).

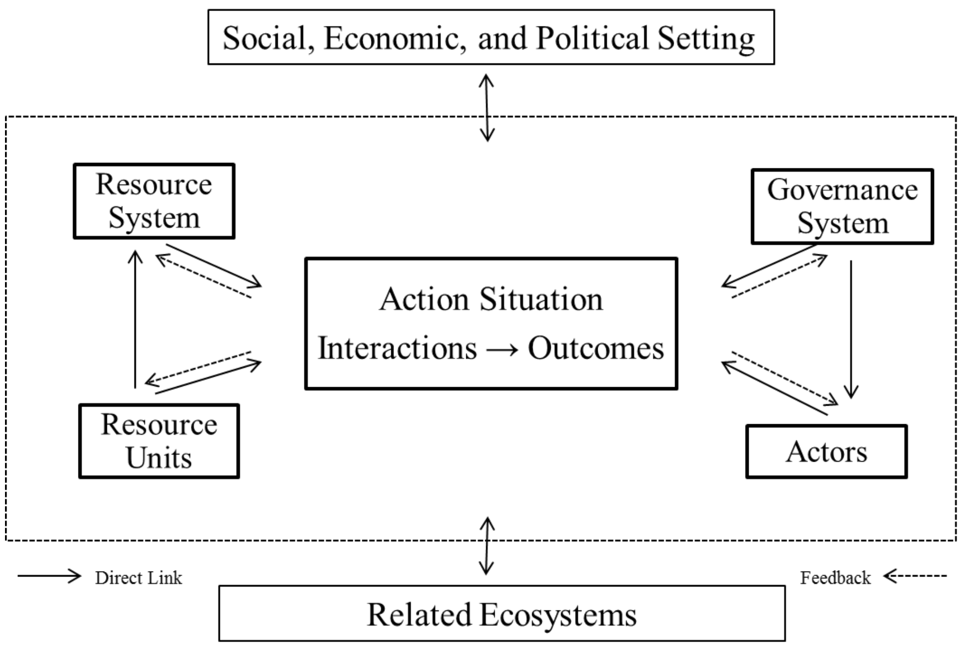

Figure 1.1: SES Framework (Source: adapted from McGinnis and Ostrom, 2014; Ostrom, 2009)

1.2.2. Research framework

The SES framework provides an initial map to understand which variables and their links have to be considered when analysing a socio-ecological system. Yet, in principle, this framework was developed by Ostrom and colleagues to study resource extraction and to show how common-property regimes emerge over time. Because of important differences with respect to the main aim of our PhD research, which focuses on analysing the impact of a specific mode of governance (contracts) embedded within a broader governance system (e.g. environmental legislation, or the existing property rights' regime), we modified and adapted the SES framework. Important to note is that our focus is not on resource extraction (e.g.

quantity of fish caught or litres of water consumed), but on the production and conservation activities within the buffer zone of a protected area.

As alternatives to the extraction-oriented ‘resource units’, McGinnis and Ostrom (2014, 2012) proposed the terms ‘resource units and goods’, or ‘goods and services’, the last term with a “more explicit consideration of production, exchange, and other core concerns of the discipline of economics” (McGinnis and Ostrom, 2012, p. 15). In this PhD, the main focus of the analysis is a ‘specific land use system’. Although land is a resource, simply using the SES variable ‘resource units’ would not be accurate for our analysis, because the latter’s focus is not on the extraction of units of a resource or merely of the land itself, but on the production of agricultural and forestry goods and the provision of ES as a result of specific ways of using the land. Thus, guided by the proposals of McGinnis and Ostrom (2014, 2012) ‘resource units’ are translated into ‘land uses’ (Figure 1.2). This not only changes the content covered by this variable, but also influences the linkages between the different variables as will be discussed further.

The main actors within the research area are households using land, in addition to governmental and non-governmental organisations. Following Hinkel et al. (2014), we place governmental and non-governmental organisations among the actors, instead of considering them as being part of the governance system as suggested by Ostrom (2009), because organisations can be understood as a special kind of actors. According to McGinnis and Ostrom (2014), the focus should be on actors when an analysis looks at actions taken by agents of an organisation, such as a government or NGO.

A last modification to the SES framework adopted in this PhD research is related to the SES variable ‘governance systems’. The SES framework aims to understand how governance systems emerge and are designed. The focus in this PhD research is more explicitly on analysing the impact (outcomes) of one specific mode of governance, i.e. PES contracts, embedded within an existing broader governance system. The research such as that carried out here is termed by Hagedorn (2008) as taking an ‘ex-post institutional change perspective’, as the analysis starts at the point where the coordination mechanism is already agreed upon. Within the analysis of PES as a specific coordination mechanism, we can distinguish institutions, the institutional environment and the mechanism itself, i.e. the institutional arrangement.

Institutions are the ‘rules of the game’ (North, 1990). Bromley (1989) defines institutions as the “rules and conventions of society that facilitate coordination among people regarding their behaviour”. They consist of formal rules and/or informal prescriptions, such as norms (Crawford and Ostrom, 1995). The SES framework assigns norms to actors and rules to the governance system (Hinkel et al., 2014; Ostrom, 2009).

The institutional environment refers to the collection of political, social and legal foundations that create the basis for production and distribution of goods and services (North, 1991). Laws and property rights are two attributes of the institutional environment (Fauzi and Anna, n.d.), which are part of the broader social, economic and political setting within which the institutional arrangements - that are the focus of this research - operate (Figure 1.2).

Institutional arrangements are "the arrangements between economic units that govern the ways in which these units can cooperate and/or compete" (Davis and North, 1971, p. 7). They are the contracts or arrangements created for a transaction or a set of related transactions, and are organisational solutions to make institutions effective (Van Huylenbroeck et al., 2009). Contracts thus provide ways of coordinating relationships among transactors (Chaddad, 2009; Ménard, 2004). They are also referred to as 'modes of governance' (Ménard, 2005), 'mechanisms of governance' (Ménard, 2012) or 'governance structures' (Williamson, 2000).

The PES contracts that are the focus of this research (e.g. Socio Bosque contracts) are similar to agri-environmental contracts which are termed 'contractual arrangements' by Van Huylenbroeck et al. (2009). Hence, we consider the PES contracts in this PhD research as contractual arrangements because a state body (the Socio Bosque Secretariat) is the coordination centre, which makes individual contracts with private actors who provide ES to society.

Finally, the other variables of the SES framework have remained unchanged in this PhD research. In our analysis, the focus of the action situation is on operational choice levels, i.e. the choices of individual households and, in a first analysis (Chapter 4) also the choices of organisations. The interpretation of the observed outcomes within the action arena is the final step of this analysis. Outcomes can be measured with outcome metrics (Hinkel et al., 2014). According to Corbera et al. (2009), institutional performance assesses how PES achieve their stated objectives (outcomes). To carry out an institutional performance analysis of PES, the focus should be on outcome evaluation criteria (Prokofieva and Gorriz, 2013). Whether a specific outcome can be regarded as sustainable is evaluated based on ecological, social and economic indicators, such as economic performance (Hinkel et al., 2014).

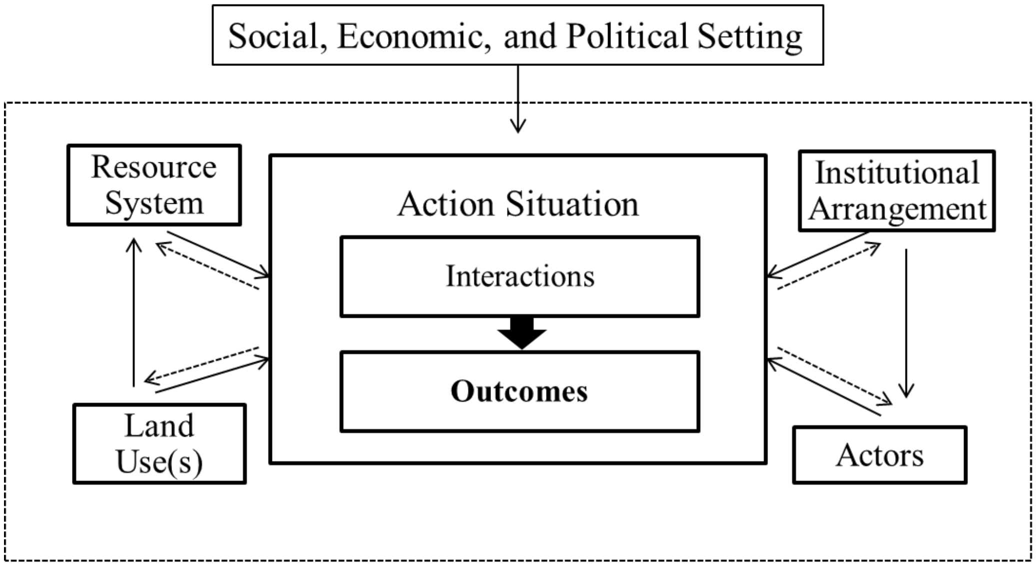

Figure 1.2: Research framework (Source: Raes adapted from McGinnis and Ostrom, 2014; Ostrom, 2009)

1.2.3. Operationalizing the framework

The framework (Figure 1.2) shows how PES as an institutional arrangement (coordination mechanism) operates within a broader environment, whereby it both influences and is influenced by the other variables of the framework. In this section, we operationalize the framework by defining the different variables used in this PhD research.

PES as an institutional arrangement can be characterized by the actors who participate, the way these actors participate, the roles these actors perform, and the land uses (or ES) that are considered within this institutional arrangement (first analysis, Chapter 3). Organisations can offer specific institutional arrangements that - depending on their uptake - will impact on the costs and benefits for the different actors involved and the resulting land use choices they make. These impacts can be measured (second analysis, Chapter 4). The adoption of specific institutional arrangements (such as PES) will have an impact on households (actors) and on their land use choices. The decision-making process is influenced by characteristics of the households (actors) and the land uses they implement or potentially can implement (third analysis, Chapter 5). Finally, households (actors) can have preferences for specific PES contracts (institutional arrangements), based on their own situation and the characteristics of the different contracts. The differences in preferences for diverse contracts may thus be explained by specific households' characteristics (fourth analysis, Chapter 6).

The variables used in our analysis are now presented:

- ***social, economic, legal and political setting in the country of Ecuador*** (see Chapter 2 on the research context)

- Ecuador's constitution, which determines the institutional environment, Ecuadorian legislation and municipal ordinances, and the laws on property rights.
- The general level of economic development.
- The economic setting includes Ecuador's productive sectors, among them the agricultural sector as well as markets. While the price setting of agricultural products is not studied (because we assume that prices are not influenced by production changes occurring within the research area) they are an important determinant of benefits and costs and hence of the trade-offs that actors face.

- ***resource system***
  - Watersheds and land are important for biodiversity conservation in the buffer zone of Podocarpus National Park (see research context). The nature of the resource system has an influence on the choice for specific modes of governance to coordinate actions between people who own and/or use land in the buffer zone, inhabitants of the canton of Loja who use water, and the national and global users of biodiversity.
- ***land uses***
  - On an individual farm, several land cover types can be found or are implemented: agricultural land (crops, pasture), agricultural land under agroforestry (organic coffee and silvopastoral systems), natural forests (primary, secondary, recent growth), and tree plantations.
  - The different land uses considered in the analysis include:
    - Conservation, which refers to the protection of an ecosystem. In PES, this normally does not refer to more detailed measures of conservation such as those targeting specific species, but remains limited to the protection of a specific area.
    - Restoration, which constitutes the process of assisting recovery (Ormerod, 2003).
    - Plantations, which are understood as reforestation for timber production. These plantations can be of species with a rather negative impact on ES, e.g. pine (*Pinus patula*) (Farley et al., 2004; Hofstede et al., 2002), or less negative such as Andean alder (*Alnus acuminata*) (Castaño-Villa et al., 2014; Knoke et al., 2014; Murcia, 1997).
    - Agroforestry, which can be defined as "a form of sustainable land use that combines trees and shrubs with crops and/or livestock in ways that increase and diversify farm and forest production while also conserving natural resources" (Molua, 2005, p. 199). In this PhD thesis, we consider silvopastoral systems and coffee agroforestry.
    - Milk production and coffee production.

- The different land uses can be characterized by different levels of productivity and production of:
    - agricultural and forestry products; and
    - ES including biodiversity, with land uses as a proxy for ES.

- ***actors***
  - Local and (inter)national NGOs.
  - National and local governments implementing conservation actions:
    - Both are characterized by the conservation and/or rural development actions they undertake in the research area, and the costs they incur when implementing these actions. The costs of a PES scheme include direct implementation costs, transaction costs and opportunity costs (Jack et al., 2008) (see Chapter 4).
  - Local households that own and/or produce on land within the buffer zone. UNSTATS (2014) defines a household as one person living alone or a group of two or more persons living together and who have common arrangements for the provision of food and other essentials for living. When it is more than one person, "the group may pool their incomes and have a common budget to a greater or lesser extent; the persons in the household may be related or unrelated or consist of a combination of related and unrelated persons" (UNSTATS, 2014).
  - Households are characterized by:
    - socio-economic variables (educational status, size, labour, land endowment, income);
    - risk aversion (Chapter 5);
    - assessment of environmental problems related to watershed services (Chapter 6); and
    - preferences for specific PES contracts (Chapter 6).

- ***institutional arrangements***
  - The focus is on PES, understood as contractual arrangements between the government and individual landholders (see actors). These contracts define parties and stakeholders in the agreement, their roles and responsibilities, contract duration, payment levels, and systems of performance monitoring, payments and sanctions (see Annex 1 and Chapter 3).
  - Other institutional arrangements present in the research area and which will be compared to the PES schemes are:
    - incentives for timber plantations;
    - markets for agricultural and forestry products;
    - markets with price premium for organic coffee; and
    - government creation of protected areas within the buffer zone (hierarchies) through land purchases and land sales.

- ***action arena***
  - In this study, the focus is on operational choice levels (McGinnis and Ostrom, 2010), i.e. the choices of either individual households or organisations (Chapter 4, 5 and 6). The decision-making mechanism is represented by a decision-making procedure which is influenced by the characteristics of organisations and households. The decisions focus on how to allocate resources based on cost criteria (Chapter 4), how to allocate land based on farm household's risk aversion (Antle, 1987; Gómez-Limón et al., 2003) (Chapter 5), and landholders' preferences (Chapter 6).
- ***interactions***
  - Implementation of conservation and rural development actions.
  - Production of agricultural and forestry products.
  - Enhancement of non-market ES.
- ***outcomes***
  - The effects of the interactions based on the implementation of specific institutional arrangements generate outcomes in terms of costs made (Chapter 4), land use systems applied and income generated (Chapter 5), or in terms of PES preferences (Chapter 6). For all outcomes, specific time frames are considered.
  - Outcomes are measured through:
    - Concrete land use systems, following Engel et al. (2008) and Ferraro and Kiss (2002), these are used as a proxy for:
      - ES provided by specific land use systems; and
      - Conservation, such as hectares of forest conserved or deforested.
    - Rural income generation, which is understood here as the income generated by a household from the land it uses.
    - Poverty reduction, defined as a process by which people move above a notional poverty line (Angelsen and Wunder, 2003). Absolute poverty means measuring poverty in relation to the amount of money necessary to meet basic needs, whereas relative poverty is defined in relation to the prevailing standards of living of other members of the society (Scheidel, 2013).
    - Cost-effectiveness.
    - Specific attributes of the contract, the institutional arrangement focus of this analysis.

1.2.4. Analytical blocks, research questions and linkage to framework

Each chapter addresses different aspects and linkages among the subsystems of the framework (land uses, institutional arrangements and actors). This thesis builds on one chapter that introduces the research area and social and political setting and three groups of analytical chapters, which are introduced here. Each of the analytical chapters stands alone as an individual study. Overlaps between the different chapters have been kept to a minimum, but may occur due to the area description and explanation of the programmes analysed.

A. Research area

Chapter 2 introduces the research area. In addition this chapter describes the social, economic, legal and political setting (Figure 1.2) in which the research took place.

B. First analytical block – Institutional arrangement of PES – Chapter 3

The first analytical block of this PhD (Chapter 3) relates to characterizing the institutional arrangements (Figure 1.2). This chapter gives the broader picture of PES in Latin America, specifically the Andean and Mesoamerican region. It aims to analyse and discuss the design characteristics of PES and PES-like schemes found in several Andean and Meso-American countries. According to Prokofieva and Gorriz (2013), an institutional design specifically addresses the design aspects of PES, i.e. characteristics of the ES it targets, evaluating the users, providers, intermediaries, etc. This chapter allows to understand how schemes have changed over time, it helps introducing the literature on PES design, introduce Ecuadorian schemes, as well as schemes of several other Latin American countries for comparison. This chapter departs from the following question:

- Do PES schemes, as hybrid forms of governance ("i.e. intermediary governance structures positioned between markets and hierarchies" (Muradian and Rival, 2012, p. 96)), evolve over time? If so, do they incorporate more market characteristics or more characteristics of hierarchies over time?

The following aspects are analysed:

- characteristics of ES involved (degree of commoditization);
- participation mechanisms of the actors in the schemes;
- the coordination and interaction mechanisms used by the actors in the schemes;
- conditionality related to the payments; and
- the existence and role of intermediaries.

In addition the chapter analyses whether the following factors aid in explaining the results of the analysis?

- size of the different groups of actors;

- type of ES; and/or
- resistance against the PES concept.

The analysis uses qualitative data on initial and current PES design characteristics. For each design characteristic, different classes were defined based on whether the characteristic could be part of a market, a hierarchy or a hybrid mode of governance. Data on the design of the schemes were obtained through established scientific and grey literature, and through a survey with 17 PES experts carried out from April to August 2014. The analysis is based on the observation of the defined characteristics in the initial and current design.

C. Second analytical block – Performance of PES contracts – Chapters 4 and 5

In the second analytical block we analyse the impact of PES contracts (Socio Bosque programme), other programmes using incentives for landholders and the existing land uses on conservation and ES provision, and the creation of rural income. This chapter evaluates the performance (outcomes, Figure 1.2) of PES contracts (Socio Bosque programme) implemented in the research area. It analyses whether these contracts can contribute to the conservation or change in specific land use systems to secure ES flows. In addition, these chapters evaluate the impact of contractual payments on households. Performance is measured through an ES indicator, in units of land under specific management actions (e.g. area conserved) and in terms of household income generation of PES contracts. These performance parameters are compared with those obtained through other coordination mechanisms (markets and creation of a hierarchy).

The first analysis of this second block (Chapter 4) focuses on both land use levels and national, regional and local actors. This chapter focuses on the costs organisations and households (actors) make through implementing (interactions) different modes of governance (institutional arrangements), and measures the outcomes in terms of cost-effectiveness of the ES provided by different land uses and of the income generated for households (Figure 1.2). According to Mönkkönen et al. (2011), cost-effectiveness is an important aspect of any successful conservation strategy. Specifically, in the buffer zone of Podocarpus National Park in the canton of Loja three programmes are implemented that aim at improving the conservation status of the area and the generation of rural income. In addition to the Socio Bosque Programme (the contractual arrangement), the other two programmes consist of a market-oriented mode of governance (organic coffee certification), and a hierarchical mode of governance (establishing conservation areas through land purchases by the municipal water company of Loja through FORAGUA). The latter two programmes serve as comparisons to Socio Bosque’s approach because of their distinct design. Chapter 4 tries to answer the following question:

- What is the cost-effectiveness of a PES contract with respect to ES provision, conservation and rural income generation?

- How does this compare with using a market mechanism for an environmentally friendly product, and the establishment of municipal reserves through the purchases of land (hierarchy)? This is analysed by comparing Socio Bosque contracts, FORAGUA's land purchases and an organic coffee label.

Implementers of the different programmes provided the data necessary for the cost calculations. In addition, quantitative socio-economic and land-use data were collected by surveying participants (and non-participants) in the different programmes. Landowners were surveyed from August 2011 until January 2012 in five parishes of the municipality of Loja. In total, 37 organic coffee farmers, 27 non-organic coffee farmers and seven participants of Socio Bosque were interviewed. In addition, a survey to grade ES provision of different land uses was carried out with 25 experts.

The second analysis of this analytical block (Chapter 5) focuses on both land use and household level; and its objective is to analyse the effect of using per hectare payments for conservation and restoration, and incentives to promote timber plantation establishment. This chapter focuses on different institutional arrangements (contracts, incentives, markets, and land sales). It assumes that households (actors) are risk-averse when making decisions (choices). Diversification of land uses (creating a portfolio of land uses) and/or choosing low risk land uses are strategies that can be used by farmers to reduce risk. Outcomes are measured in terms of the portfolio land use allocation, and change in household' income and poverty levels (Figure 1.2). Specifically, through portfolio allocation this chapter deals with the potential impact on milk producers of participation in the Socio Bosque Programme for conservation and restoration, and in an incentive programme for timber plantations. It also looks at the effect on land use of farmers' participation in the different PES programmes under study, and what the differences are in land use allocations between farmers with and those without forests on their land. Portfolio theory is used to model the household's decisions-making process, which is assumed to be influenced by household's initial land use and its productivity. Chapter 5 concerns the following research questions:

- Would dairy farmers participate in the Socio Bosque Programme for conservation and restoration, and in an incentive programme for timber plantations?
- What is the effect on land use of farmers' adopting a portfolio (combination) of the different programmes under study?
- Is there a difference in land use allocation on existing pastures between milk producers that have forest and those without forests on their land?
- What is the effect on dairy farmers' income and poverty levels of participation in the programmes studied?

For the analysis, farmers' socio-economic household and farm data were collected in the research area in November-December 2011 and from March to July 2013. During the first stage, 19 detailed surveys were carried out. In the second stage, a survey was carried out with 95 farmers (also part of the choice experiment of Chapter 6). Additional data for production calculations were obtained through scientific literature and official statistics.

D. Third analytical block – Household preferences for contractual arrangements – Chapter 6

The third analytical block of this PhD (Chapter 6) also aims at understanding the (potential) impacts of PES contracts, yet it studies the possible uptake of PES contracts by looking at the preferences (choice outcome) of milk producers (actors) for these type of contacts (institutional arrangements) (Figure 1.2). The analysis in this part focuses on the household level and aims to study whether dairy farmers choose contracts for the adoption of silvopastoral production systems with and without additional management requirements, what the characteristics of the chosen contracts, and the main determinants of farmers' preferences are. This chapter tries to answer the following questions:

- Do dairy farmers prefer contracts for the adoption of silvopastoral systems over the current situation?
- What are the characteristics of the preferred contracts?
- What are key determinants of farmers' choices?

For this analysis, a choice experiment was carried out to analyse landholders' preferences in terms of PES contracts for the adoption of silvopastoral systems. A test run was done with ten households in February 2013. The choice experiment survey with 120 milk producers was carried out from March to July 2013.

Finally, the research questions are reviewed in Chapter 7 to draw conclusions and to discuss their implications in relation to wider societal issues.

Chapter 2: Research Context

2.1. Research Area

The research was carried out in that part of the buffer zone of Podocarpus National Park that lies in the municipality of Loja (Figure 2.1.A and 2.1.B). Podocarpus National Park has a total area of 146,000 ha. The total research area is 40,717 ha. The area of the buffer zone, excluding the national park, accounts for 27,834 ha (Figure 2.1.C). More details of the research area are given in Chapter 2.

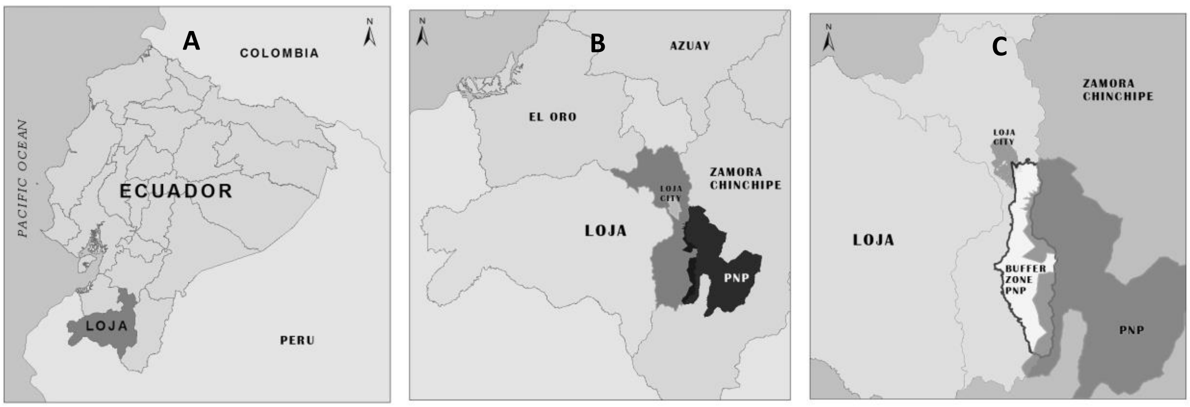

Figure 2.1: Research area and Podocarpus National Park (Source: Cevallos and Raes)

2.2. Social, economic and political setting

As indicated in our research framework, this chapter presents the social, economic and political setting in which the research took place. Figure 2.2 shows the different administrative levels ranging from national level (the country of Ecuador) to the level of individual households.

Figure 2.2: Different administrative levels of the overall social, economic and political setting

2.2.1. Political-administrative division

Ecuador gained its independency in 1824. It has a political-administrative division that consists of provinces (24), cantons (221) and parishes (1,500). These make up the different levels of territorial organisation of the country (INEC, 2012a).

The research took place in Southern Ecuador. Although not a legal entity, the three provinces of Southern Ecuador (i.e. El Oro, Loja and Zamora Chinchipe) are referred to as the Southern Region. This area is one of the nine Ecuadorian planning zones used by the National Planning and Development Secretariat (SENPLADES). The three provinces together make up 11% of Ecuador's area and account for 8% of the nation's population (INEC, 2011a).

Within this region, the research focuses on the province of Loja (Figure 2.3). This province consists of 16 cantons of which one is the canton of Loja, where the field research took place (INEC, 2012a). The canton of Loja consists of 4 urban parishes, including the city of Loja, and 13 rural parishes (Figure 2.3) (FORAGUA, 2014).

2.2.2. Legal and institutional environment

Contractual arrangements for the provision of ecosystem services (PES contracts) are in first instance regulated by the Ecuadorian constitution. Article 74 of the Ecuadorian constitution of 2008 specifies that: "Environmental services shall not be subject to appropriation; their production, delivery, use and development shall be regulated by the State" (Republic of Ecuador, 2008). In principle, PES seem to contradict Ecuador's constitution (Manzano Díaz, 2010), which creates a political and institutional grey area for PES.

This can be observed from the way the Socio Bosque Programme was designed (Annex 1), the scheme that is the key focus case of this research. This programme signs contracts with individual and communal landholders to conserve or restore ecosystems in return for payments for doing so. The Socio Bosque Programme itself avoids the use of the term '*pagos*' (payments), since that could be understood as acknowledging some form of private ownership over ecosystem services (Ministry of the Environment, Ecuador, 2012a). Instead, the term '*incentivos*' (incentives) is used, which is less controversial and in line with the constitution. Article 71 of the constitution states that: "The State shall give incentives to natural persons, legal entities and communities to protect nature and to promote respect for all the elements comprising an ecosystem" (Republic of Ecuador, 2008). In addition, according to the Ministry of the Environment, Ecuador (2012a), linking payments to conservation and restoration efforts (land uses) also avoids dealing with the issue of property rights over ecosystem services. To participate, the owner only needs to have the right of land ownership, and thus the right to implement specific land uses on her/his land, not the rights over ecosystem services.

On water governance the Ecuadorian constitution states in Article 411 that "the State shall ensure the conservation, restoration and integrated management of water resources [....] Any activity that may affect water quality and quantity, and ecosystem's balance, especially in springs and water recharge areas shall be regulated [....]". Additionally, Article 264 of the constitution and Article 55 of the Code of Zoning, Autonomy and Decentralization state that it is the authority of the Municipal Decentralized Autonomous Governments to exercise control over land use within their respective territories. Moreover, Article 137 states that "the competencies for the provision of public drinking water in all its phases shall be executed by the Municipal Decentralized Autonomous Governments" (Republic of Ecuador, 2008).

In addition to the general legislation, the municipality (canton) of Loja has a municipal ordinance that regulates land uses in its watersheds and establishes the declaration of municipal protected areas (Municipality of Loja, 2007). The agency responsible for implementing watershed conservation activities to assure watershed service provision is the '*Empresa Pública Municipal de Agua Potable y Alcantarillado'* (the Municipal Public Company for Water Supply and Sewerage, known by its Spanish acronym EMAAL-EP).

2.2.3. Environmental issues

Ecuador has been identified as one of the 17 most ecologically diverse countries in the world (Mosandl et al., 2008; WCMC, 2000). The country has a total surface of 283,560 km^2, of which between 113,076 to 122,620 km^2 is natural forest (FAO, 2003; Ministry of the Environment, Ecuador, 2012b). These forests include primary as well as regenerated secondary forests. About 68,000 km^2 of these forests are privately or collectively owned, the rest is State property. The deforestation rate is one of the highest in South America with an annual rate of 890 km^2 between 1990-2000 and 776 km^2 between 2000-2008 (Ministry of the Environment, Ecuador, 2012b; Mosandl et al., 2008). Several programmes have been implemented in Ecuador to halt this deforestation, among them Socio Bosque (see Annex 1).

Our research focuses on programmes that have been implemented in the area surrounding Podocarpus National Park in Southern Ecuador (Figure 2.3). Podocarpus National Park lies in the provinces of Loja and Zamora Chinchipe, in the southern part of the Ecuadorian Andes and has an area of around 1,463 km^2. According to Keese et al. (2007), the tropical Andes is "the global epicentre of biodiversity, leading all other hot spots in virtually every category of species diversity and endemism". The National Park has a very high diversity of tree species (Madsen and Øllgaard, 1994), vascular plants (Peters et al., 2010), epiphytes (Bøgh, 1992), and *páramo* communities (Keating, 2008); and is the habitat for numerous animals, including many birds species (Rahbek et al., 1995), bat species (Rex et al., 2008) and the endangered Andean or spectacled bear (*Tremarctos ornatus*) (DeMay et al., 2014). Threats to the different Andean ecosystems include road building, mining, expansion of agricultural land, lack of government coordination, and inadequate funding for conservation (Clark et al., 2009a, 2009b; Keese et al., 2007). The reason to focus on the area neighbouring a state-owned national park is that protected areas do not exist in isolation but interact constantly with their surroundings. The development and activity on adjacent lands influences the protected area (MA, 2005a). Management of buffer zones is a key part of a strong conservation strategy for protected areas (DeFries et al., 2010; Martino, 2001; Prins and Wind, 1993). According to TEEB (2010a), people in buffer and transition zones should have secure incomes from environmentally friendly resource use to support conservation in a protected area. Table 2.1 summarizes the different conservation areas within the research area (Figure 2.3).

Table 2.1: Area of all protected spaces in the research area

Protected area	Area (ha)
Protected Forest *Hoya de Loja Flanco Oriental*[a]	3,598
Pizarros Watershed	734
El Carmen Watershed	912
Protected Forest *El Bosque*	2,192
Protected Forest Rumi Wilco	39
Podocarpus National Park (within research area)[b]	12,883
Total Area Protected	20,358
Total Research Area	40,717
Total area not protected	20,359

[a] Total area: 7,326 hectares; [b] Total area: 146,300 hectares

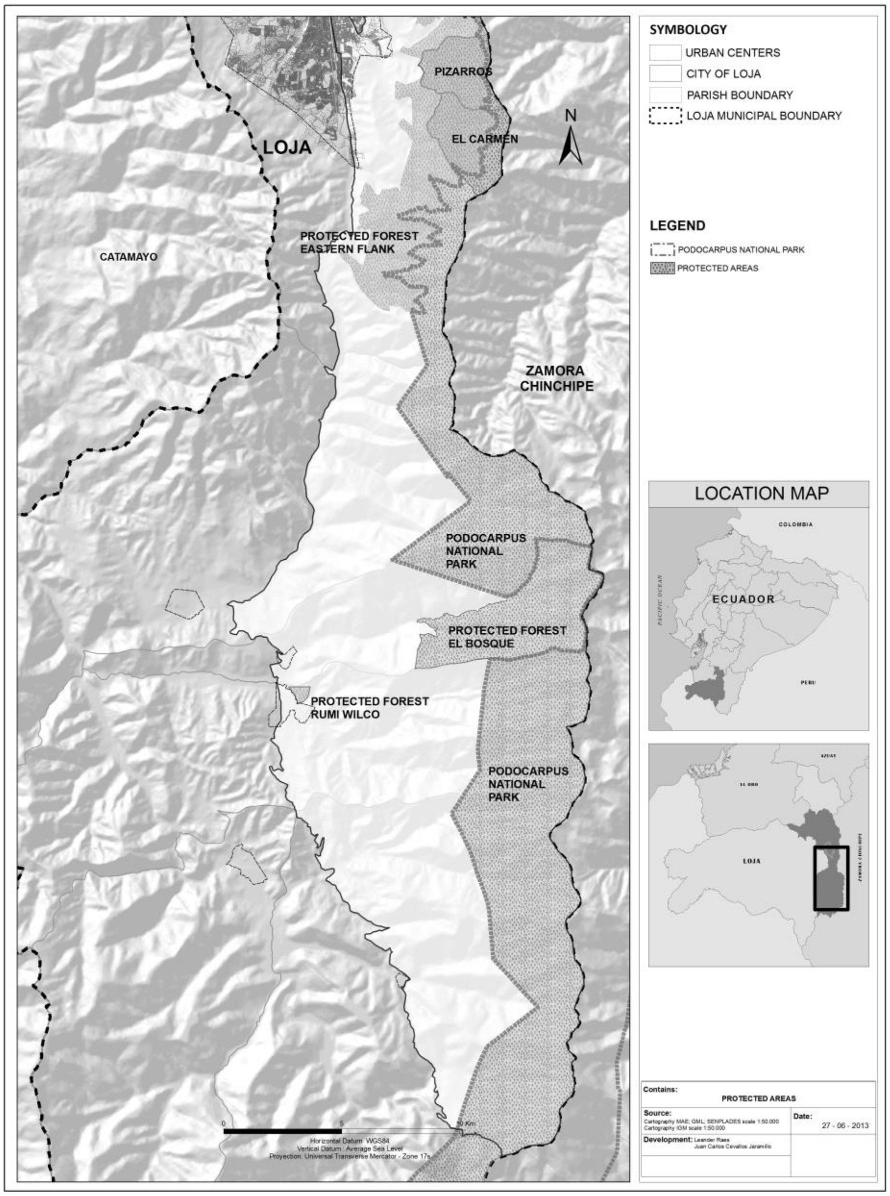

Figure 2.3: Protected areas in research area (Source: Cevallos and Raes with data from Ministry of the Environment of Ecuador, 2013; GLM, 2013; SENPLADES, 2013; IGM , 2013)

In the Andean Region of Ecuador, mountain forests and Andean grasslands (*páramos*) provide key watershed services. The most important services provided by these highland ecosystems are improved water quality through sediment retention (Brauman et al., 2007; Célleri and Feyen, 2009) and regulation of water flow (Bruijnzeel, 2004; Roa-García et al., 2011). In addition to providing drinking water for people, these ecosystems also provide a habitat for a multitude of plant and animal species (Josse, 2001; Mutke and Barthlott, 2005).

The ability of these natural ecosystems to provide watershed services to people in up- and downstream areas of the watershed was reduced by their conversion to agricultural land. Livestock grazing, periodic burning of pastures, and the use of pesticides have negative consequences for watershed services, such as reduced water retention capacity and contaminated water due to sediment, manure and pesticide effluence (FORAGUA, 2014; Webber, 2009; Zapata et al., 2012). Growing populations and subsequent increasing demand for water add to the problems of water provision. The latter problem is worsened by the periodic droughts of which the Southern Andean region suffers during a part of the year. These deficits partly result from naturally occurring drought periods, which can decrease water flow substantially. Another reason for these deficits is the loss of native ecosystems that function as water retention areas, thus increasing water run-off and subsequent loss of water retained within the watershed (Harden et al., 2013) (see also Annex 2).

The city of Loja receives water from eight watersheds: Shucos, Jipiro, Mendieta, El Carmen, San Simon, Namanda, Monica and Curitroje (of which the last five are situated within the research area, see Figure 2.4). The creation of the Regional Water Fund (FORAGUA) was inspired by a predicted threat to water quality, following the degradation of upstream highland ecosystems that are crucial for the provision of watershed services (Gordillo, 2013; Webber, 2009; Zapata et al., 2012). The water fund uses an environmental tariff on water use to finance conservation and restoration activities in the watersheds of the Southern Region (see Annex 2). Land purchases within the watersheds by the municipal water company of Loja, EMAAL-EP, are one of the strategies for ES conservation to which PES contracts are compared. Figure 2.4 maps the locations in the research area where currently water captions occur (numbers 1,2, 3, 7, 8, 9, 10, 11 and 12) and where more are planned (numbers 4, 5, and 6).

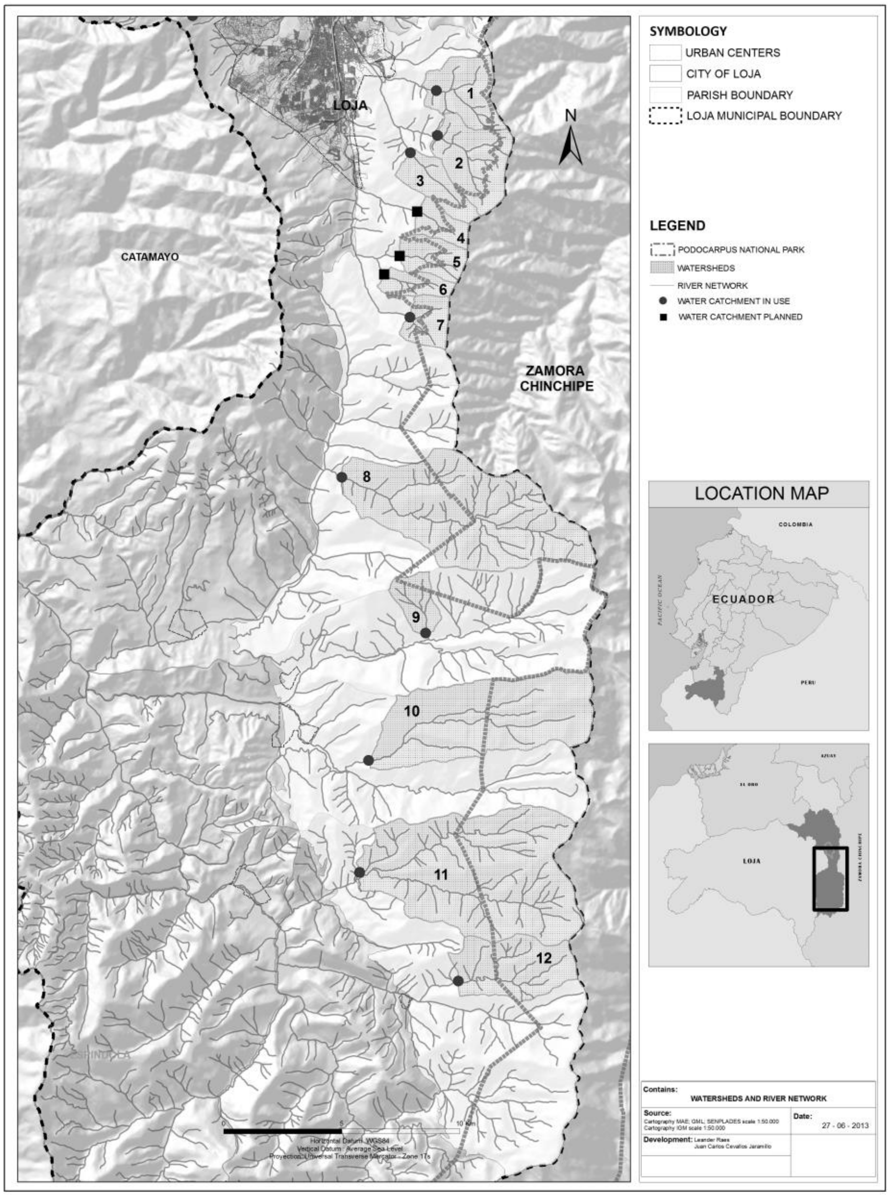

Figure 2.4: Water captions in research area (Source: Cevallos and Raes with data from from Ministry of the Environment of Ecuador, 2013; GLM, 2013; SENPLADES, 2013; IGM , 2013)

2.2.4. Socio-economical background

Ecuador has a population of around 14.5 million people of whom around 450,000 live in the province of Loja and 215,000 in the canton of Loja, mainly in the urban area (INEC, 2011a). Around 29% of the country's population live below the poverty line, which was set at US$ 73/month in 2011 (INEC, 2011b).

Although agriculture accounts for only ten % of GDP it is an important source of foreign exchange income (INEC, 2011c). The country's most important export products are banana, cocoa, palm oil and coffee (FAOSTAT, 2014). Within the research area, the most important agricultural activities are the production of coffee and sugar cane, and dairy cattle. This research includes coffee, dairy and timber production. The province of Loja produces around 16 % of Ecuador's coffee (INEC, 2014), and around four percent of the country's milk (INEC, 2012b). Pastures and forests are the most important land uses in the research area (Figure 2.5). Although not an important source of export revenue, tree plantations for timber are important in some parts of the Ecuadorian landscape. There is around 167,000 hectares of forest plantations, mainly *Pinus* spp. and *Eucalyptus* spp. (around 75 % of total tree plantation area). Approximately 90 % of the tree plantations occur in the Andean Region of Ecuador, where also the research area is situated (FAO, 2003) (Figure 2.5). A more detailed description of the livelihoods of households in the research area is given in Chapters 4, 5 and 6.

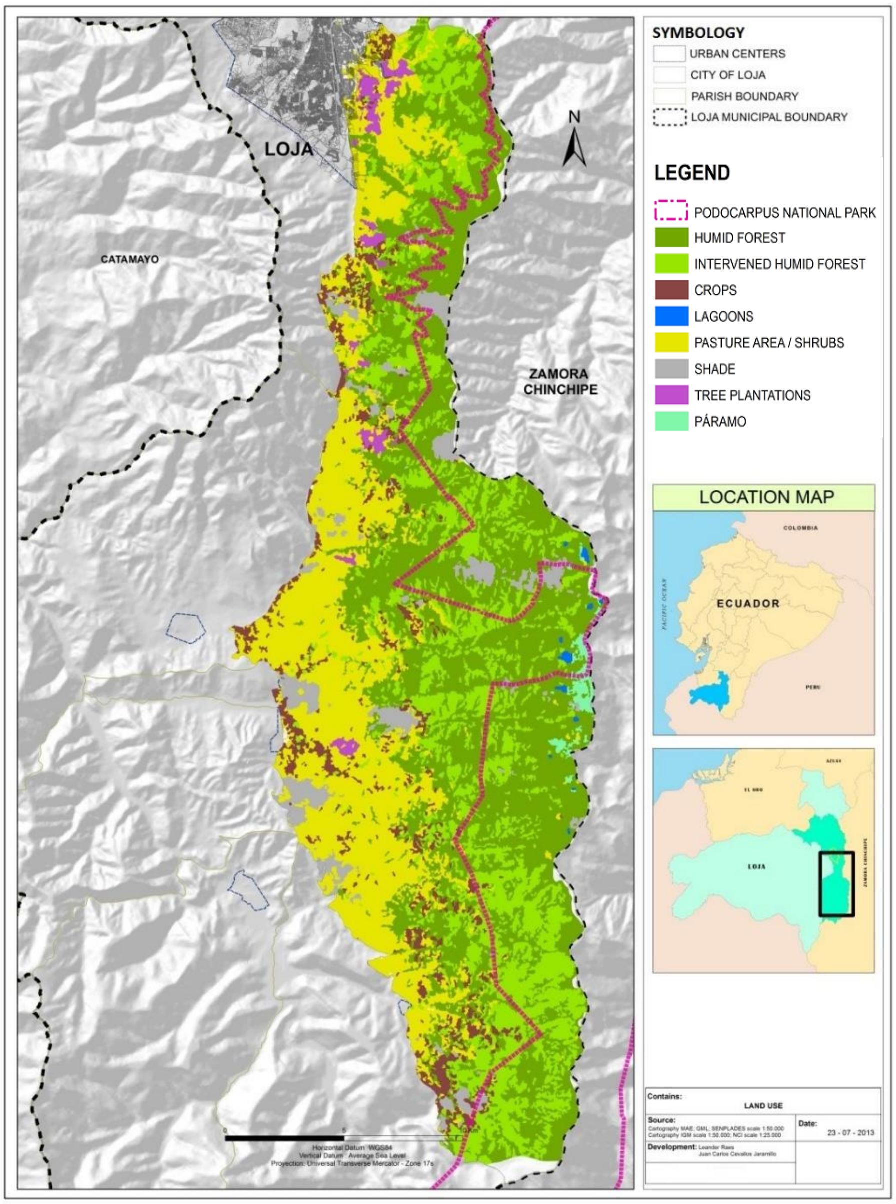

Figure 2.5: Main land uses in the research area (Source: Cevallos and Raes with data from from Ministry of the Environment of Ecuador, 2013; GLM, 2013; SENPLADES, 2013; IGM , 2013; NCI, 2013)

Chapter 3: Towards market-or command-based governance? Analysis of the evolution of PES design in Andean and Mesoamerican countries

Based on: Raes L., Loft, L, Le Coq, J.F., Van Huylenbroeck, G and Van Damme, P. Towards market-or command-based governance? Evidence of the evolution of PES design in Andean and Mesoamerican countries. (Under review).

Abstract

PES are a hybrid mode of governance, situated between markets and hierarchies. However, market structure has been used as a theoretical model to inform PES design. In this paper we analyze whether 16 Andean and Mesoamerican PES schemes have, since their initial design and implementation, gradually incorporated more market characteristics or conversely whether and to what extent these schemes have changed towards more reliance on command-based mechanisms. The schemes analyzed cover a range of governance mechanisms, from small markets to (almost) complete hierarchical organization. Results suggest that over time an increasing number of the schemes have incorporated characteristics of a hierarchy to organize ecosystem service users. Mainly through the use of taxes/tariffs and by governments acting directly on users' behalf. Contractual agreements, with payment levels bilaterally negotiated or set by intermediaries, with mainly individual and communal landholders as providers, remain at the core of most schemes studied. Intermediaries are important actors in almost all schemes analyzed. They mainly organize and/or represent users, and are usually national or local governments. Evolution of the schemes analyzed suggests that there is no convergence towards a market for ecosystem services, but an increasing complexity in the schemes' design and a (fuzzy) diversity of schemes.

3.1. Introduction

One way to provide incentives to improve or conserve ecosystem services (ES) provision, is through the establishment of so-called 'Payment for Environmental Services' (PES) schemes[5] (Adhikari and Agrawal, 2013). PES aim at influencing behaviour by providing (monetary) incentives instead of through direct regulation (Jack et al., 2008). PES are thus seen as different (even opposed) to more traditional policies such as government directives that aim to promote/enforce conservation (Adhikari and Agrawal, 2013).

While some scholars still refer to the mechanism of market governance when explaining how PES schemes work (Alvarado-Quesada et al., 2014; Lurie et al., 2013; To et al., 2012), literature increasingly emphasizes that PES are not 'real' markets, but rather hybrids that lie between markets and command-based coordination mechanisms (hierarchies) (Muradian and Rival, 2012; Pirard and Lapeyre, 2014; Vatn, 2015; Wunder, 2008). However, markets are still used as the theoretical governance model by many institutes, government agencies and multilateral organisations to inform on the functioning of PES (e.g. FAO, 2011; UNEP, 2008). As Shapiro-Garza (2013, p. 6) states, "the 'pure' market mechanism remains the ideal type amongst the primary promoters and funders of PES in the global south". Thus, for the latter, while initially PES should rely on state intervention, in order to be sustainable in the long run PES should evolve towards 'real' market working (see e.g. FAO, 2014).

To date, only limited information exists on how PES schemes have evolved over time (Pirard, 2012b). In order to fill this gap, in this chapter we assess the evolution of PES design over time. Although an increasing number of papers present a comparative analysis of PES (Hejnowicz et al., 2014; Martin-Ortega et al., 2013; Pirard and Lapeyre, 2014; Schomers and Matzdorf, 2013), often the distinction between initial and subsequent governance mechanism of schemes is not clear. Thus, in this chapter we focus on the evolution of PES schemes to find out whether PES, as hybrid modes of governance, are evolving towards more market or more command-based governance.

Following Vatn (2015), we consider PES using markets and hierarchies (command-based) as two opposing governance systems. . To assess the changes in PES design, we conduct a comparative qualitative study of 16 schemes, which were initiated between 1997 and 2009 in the Andean and Mesoamerican region of Latin America, known for its great variety of PES schemes (Balvanera et al., 2012; Schomers and Matzdorf, 2013). The schemes enable an evolution analysis from their initial design to their current status.

Following this introduction, the chapter introduces the different schemes analysed. In Section 3.3 we present the framework, and in 3.4 the methodology used. Section 3.5 describes the observed changes in the schemes. Section 3.6 provides a discussion. Finally, Section 3.7 concludes.

[5] We use the terms ecosystem services and environmental services, and the use of these terms in PES, interchangeably. We exclude provisioning services such as agricultural products or timber, which are not the focus of PES.

3.2. Case studies and data collection

The schemes studied deal with watershed services, carbon sequestration, preservation of biodiversity and landscape beauty, and are implemented in Bolivia, Colombia, Costa Rica, Ecuador, Mexico, Nicaragua, and Peru (Table 3.1). With the exception of two (Noel Kempff Mercado Climate Action Project (NK-CAP) and PROFAFOR[6]) all schemes are still operational today.

The cases were developed as follows. In a first stage, we identified the Andean and Mesoamerican countries where PES have been implemented, and specific schemes through an 'ISI Web of Knowledge' search (see Supplementary Table 3.1). Only schemes for which sufficient information on their initial design characteristics was available, were included in this study. Similar to Martin-Ortega et al. (2013) schemes that were explicitly designed or planned as PES schemes, following Wunder's (2005) definition[7], but that were relabelled or interpreted as PES in literature, were incorporated in the analysis[8].

In a second stage, 17[9] PES experts involved in the implementation and management of the schemes identified were surveyed by e-mail between April and August 2014. Their names and contact details were initially obtained through the websites of the respective schemes or the intermediary (NGO, government agency) funding and/or managing the scheme. The questionnaire included qualitative questions concerning the schemes' original and current governance mechanisms, based on a series of characteristics of markets and hierarchies (see Section 3.3). The questionnaires were prefilled with information collected through literature and a review of official documents (see Annex 3). The PES experts were asked to verify the prefilled information and to clearly point out initial and current governance of the schemes. The information obtained through the surveys was then used in the analysis (Section 3.5).

Several schemes studied provide watershed services[10], a subcategory of hydrological services[11]. These schemes are systems that pay for implementing conservation practices in upper ranges of watersheds that deliver drinking and/or irrigation water to lower-located areas (Landell-Mills and Porras, 2002). Such schemes either operate on a local level, such as in the municipality of Pimampiro, Ecuador and the ESPH-

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[6] We use the name of the company 'PROFAFOR' to refer to the carbon scheme that was analysed in the context of PES by Wunder and Alban (2008). It is this carbon scheme that no longer sells carbon credits, while the tree plantations continue to be managed, and the company 'PROFAFOR S.A' continues to operate.

[7] Although alternative definitions exist (see 1.1), this definition is the most widely used in both literature, and policy and project documents (Pirard, 2012b; Wunder, 2015).

[8] For example, Martin-Ortega et al. (2013) mention FONAG, which has been named a PES scheme (Goldman-Benner et al., 2012; Porras et al., 2008), PES-like (Southgate and Wunder, 2009), and not a PES (Wunder, 2012), while FONAG does not describe itself as a PES scheme (FONAG, 2014). On the other hand, the Municipality of San Pedro del Norte fits Wunder's definition in the municipal ordinance regulating the local PES scheme (Municipality of San Pedro del Norte, 2011).

[9] In total 29 experts of the 16 schemes were contacted. For the Colombian scheme 2 people replied simultaneously.

[10] Watershed services are ecosystem services provided by watersheds, such as the provision of water, and the regulation of water quality and seasonal flows (Wang et al., 2010).

[11] Brauman et al. (2007) define hydrological services as ecosystem services that "encompass the benefits to people produced by terrestrial ecosystem effects on freshwater".

PROCUENCAS scheme in Heredia, Costa Rica (Porras et al., 2008; Redondo-Brenes and Welsh, 2010), or on a regional level, such as the Ecuadorian Regional Water Fund - FORAGUA (Goldman-Benner et al., 2012, see also Annex 2).

Also three 'pure' carbon schemes figure among the cases analysed. These schemes sell carbon credits internationally, but are implemented locally or nationally. A local-level scheme is the NK-CAP which focused on reduced emissions from deforestation in a protected area in Bolivia (Pereira, 2010). This scheme is now being transformed into a new one, the nationwide Bolivian scheme called COMSERBO (see Bryner et al., 2012, for more details). On a national level, the PROFAFOR scheme in Ecuador sold carbon credits for reforestation through a company, while the landowners and communities participating in the scheme benefit from timber sales (Wunder and Alban, 2008). Currently, only the Mexican *Scolel Té* scheme is still actively selling carbon credits.

Biodiversity as an ES is included in PES schemes bundled with the provision of watershed services and/or carbon sequestration, as in the national Mexican programme called PRONAFOR (previously Pro Árbol). At the start, there were two separate schemes, one focusing on watershed services (PSAH) and one focusing on carbon sequestration, biodiversity and agroforestry (PSA-CABSA) (Corbera and Brown, 2008). The current programme has four components: forest development, commercial plantations, conservation and ES. With PES falling under the fourth component (Rodricks, 2013). The PES component of the programme aims at providing watershed services and conserving biodiversity (CONAFOR, 2013). The national Ecuadorian scheme, Socio Bosque, includes watershed services, carbon sequestration and biodiversity conservation (de Koning et al., 2011, see also Annex 1), while the nationwide Costa Rican PES programme called '*Pagos por Servicios Ambientales*' (PSA) adds a fourth ES, landscape beauty, to these (Pagiola, 2008).

Table 3.1: Overview of the 16 study cases per country

Country	Case name	ES considered in the scheme	Starting year	Name given by implementers to mechanism used
Bolivia	Noel Kempff Climate Action Project (NK-CAP)[a,b]	Carbon sequestration	1997	Carbon credits
	RWA Los Negros	Watershed services	2003	Reciprocal Watershed Agreement (RWA)
Colombia	Water Fund for Life and Sustainability	Watershed services	2009	Water fund and Compensation for Ecosystem Services (CES)
Costa Rica	ESPH – PROCUENCAS	Watershed services	2002	PES
	PES La Esperanza		1997	PES
	PSA	Carbon sequestration; watershed services; biodiversity and provision of scenic beauty	1997	PES
Ecuador	FONAG	Watershed Services	2000	Water Fund
	FORAGUA		2009	Water Fund and CES
	PES Pimampiro		2001	PES
	PROFAFOR[b]	Carbon sequestration	1993	Carbon credits
	Socio Bosque Program	Biodiversity, carbon sequestration and watershed services	2008	Conservation incentive
Mexico	Plan Vivo Scolel Té	Carbon sequestration	1997	Carbon credits and PES
	PRONAFOR	Watershed services and biodiversity	2003	PES
Nicaragua	PES San Pedro del Norte	Watershed services	2000	PES
	PHES Gil Gonzalez		2008	PES
Peru	Alto Mayo Water Initiative	Watershed services	2009	Remuneration for Hydrological Ecosystem Services (RHES), CES and RWA

[a] Now in transformation stage to participate in the COMSERBO programme (see Bryner et al., 2012)
[b] No longer selling carbon credits

3.3. Analysis framework and criteria

We defined characteristics of markets and hierarchies that would allow us to portray the different governance structures and how these have evolved over time. Additionally, we categorized schemes according to group size, ecosystem service(s), and name used by the schemes' implementers, to analyse whether these categories provide explanations for the schemes' design and change over time.

3.3.1. Characteristics of markets and hierarchies

To characterize the difference in structure we follow Vatn (2015), and consider markets and hierarchies as the two opposed modes of governance. These modes can be defined through a series of attributes that explain how actors engaged in a transaction interact and organise their activities (Muradian, 2013; Williamson, 2010). To carry out our analysis, we first define key characteristics of markets and hierarchies, and how these would translate to PES governance. Between markets and hierarchies there are numerous hybrids (Ménard, 1996a). By analysing a series of different characteristics separately, instead of two fully differentiated modes of governance, we can take into account the hybrid governance structure of PES, as hybrid modes of governance refer to arrangements that are a combination of market and hierarchical elements (Muradian and Gómez-Baggethun, 2013). While our analysis still allows for a dichotomous analysis per characteristic (market vs. hierarchy), many variations based on the inclusion or exclusion of specific characteristics are possible.

Despite lacking a clear definition of the term 'market', the latter can be understood as a set of arrangements through which actors (buyers and sellers) exchange goods and services (Begg et al., 2003; Bromley, 1997). According to Vatn (2015), interaction through trade is the defining characteristic of markets. Markets coordinate decisions between actors using information provided through the price system (Begg et al., 2003; Parkin et al., 2011). Participants use the price system to signal preferences and as such can adjust mutually through decentralized coordination (Aspers, 2007; Lindblom, 2001; Ménard, 2005).

In addition to the pricing system, defining characteristics of markets include competition and voluntariness (Rosenbaum, 2000). The voluntariness criterion distinguishes markets from allocation mechanisms that use power and authority[12]. According to Vatn (2015), all trades are per definition voluntary. Competition is a necessary expression of the impersonal nature of market exchange (Rosenbaum, 2000). A competitive market consists of many buyers and sellers. According to neo-classical economics, in the absence of any distortions, competitive equilibrium will provide efficient quantities of goods and services in the market (Begg et al., 2003; Parkin et al., 2011). However, complete and perfectly competitive markets are a theoretical construct (Vatn, 2015). Non-competitive markets loose much of the theoretical efficiency advantage that competitive markets are supposed to have over alternative resource allocation strategies (Kroeger and Casey, 2007).

Hierarchies or vertical coordination (firms) are seen as opposed to market organisation (Williamson, 1991). A hierarchy consists of a system of command (Vatn, 2010). According to transaction cost economics theory markets are characterized by (1) high incentive intensity, as rewards for actors are directly related to their action, and (2) low administrative control to monitor and stimulate actors' activities, whereas hierarchies show low incentive intensity and high administrative control (Ménard, 1996b; Williamson, 2002).

For PES, when discussing hierarchies, the focus is more on governments, public bureaucracies rather than private ones. Coase (1960) states that governments in a sense can be considered 'super-firms' as they are able to influence the use of production factors by administrative decisions. The characteristic of command by one actor, normally a government, over the other (person or entity) is opposed to the idea that markets are based on self-regulation, characterized by voluntariness and competiveness, and a separated decision-making structure. However, also markets, especially efficient ones, require some form of government involvement e.g. to protect property rights (Rametsteiner, 2002), or in other words a broader

[12] Muradian and Gómez-Baggethun (2013, p. 1117) define authority as "a coordination mechanisms by which certain actors are able to influence the action of others using hierarchical relations".

institutional context that facilitates and constrains the behaviour of market participants (Bromley, 1997; Kroeger and Casey, 2007; Lindblom, 2001).

According to Lurie et al. (2013), PES are often theorized within the economic concept of markets. Market governance of PES would require: (i) the definition of ES units subject to trade; (ii) the establishment of a monitoring system; (iii) a transaction that is voluntary; and (iv) using price signals to allocate resources towards the provision of ES (Engel et al., 2008; Farley and Costanza, 2010; Kosoy and Corbera, 2010; Kroeger and Casey, 2007). Although PES only rarely function through competitive markets (Wunder, 2013), in theory competition should also be taken into account when analysing PES as markets for ES. However, according to Vatn (2010, p. 1247), “a trade with only one (few) agent(s) as provider(s) and one (few) as buyer(s) is not a competitive market, but still a market”. Competitive markets should not be used as the basis for defining markets in ES (Vatn, 2015).

A last issue to consider when analysing PES as a market mode of governance is the conditionality linked to payments in PES. As Goldman-Benner et al. (2012, p. 58) explain, “in a conventional market sellers receive payment only if they actually provide the contractually agreed goods or services”, i.e. an exchange must take place. In PES, this implies that payments are only made when the agreed-upon ES are delivered or when certain agreed-upon management actions are carried out (Tacconi, 2012; Wunder, 2005). According to Wunder (2013), conditionality has to be present to some extent in PES schemes.

Muradian and Gómez-Baggethun (2013, p. 1119) propose four key elements that should be considered when trying to classify PES, taking into account the complexity and diversity of governance structures. These elements are: “(1) the degree of voluntariness; (2) the degree of commoditization; (3) the social meaning of the monetary transfers involved; and (4) the role of the state (whether it is a regulator or also an actor engaged in transactions)”. These elements are taken into account in the characteristics analysed in this study, with the exception of the social meaning of payments (see Muradian, 2013). Muradian (2013) proposes additionality of the PES schemes among other characteristics to understand the social meaning of transfers. Yet, additionality is not a defining criterion of PES, but a measure of effectiveness (Wunder, 2015). We, however, add to this PES analysis how payment levels are set between the different actors.
To position these hybrids in the continuum between markets and hierarchies, we developed a set of parameters (characteristics):

- The actors: who participates?
- The degree of commoditization of ES, what is being paid for?
- Voluntariness of participation: how do actors participate?
- Coordination through the price system: how are payment levels determined?
- Conditionality related to payments: is there trade?
- What is the the role of the intermediaries?

3.3.2. The actors

According to Vatn (2015), when analysing markets as a governance structure, classifications should be focused on the type of actors involved and the format of their interactions. In markets, the actors that participate are suppliers, consumers, government(s) and possibly intermediaries. The main role of intermediaries in a market should be to facilitate exchange (Ahn et al., 2011; Gabre-Madhin, 2001). In the case of PES, the different actors can be classified as providers, users and intermediaries (Figure 3.1).

An ES provider is an entity whose management actions modify the quantity or quality of ES available to users (Corbera et al., 2007; Swallow et al., 2009). It is often mentioned that providers of ES in PES schemes should be private or communal landholders (Laurans et al., 2012; Pagiola, 2008; Wunder, 2005), however, governments also own land, and are thus potential ES providers. PES programmes could thus partially or fully target public land, as in the case of protected areas (Engel et al., 2008; FAO, 2009a; GEF, 2010).

ES users are individuals or groups of (un)organised people that make use of the targeted ES and are often those who will pay directly or indirectly for the provision of ES (Corbera et al., 2007; Swallow et al., 2009).

Intermediaries are entities that directly or indirectly shape interactions among providers, users, and the ecosystem itself (Corbera et al., 2007; Swallow et al., 2009). Providers are often the ones that get paid for ES provision or the implementation of a land-use activity, while users can be the ones that pay for the provision of ES, but it can also be an intermediary that gets paid or pays for the provision of the ES.

In addition, as mentioned before, we should consider the role of the state in PES classification (Muradian and Gómez-Baggethun, 2013). Following Vatn (2015), for the analysis we additionally distinguish between different types of actors: (i) local, national and international non-governmental organisations (type: NGO); (ii) households, households organised through associations, and communities[13] (type: Household); (iii) private businesses and producer associations (type: Business); (iv) local, regional and national governments, and official international cooperation (type: Government). The latter category includes municipal water companies managed through municipal governments, with the exception of the Costa Rican water company ESPH, which is regulated through corporate law.

3.3.3. Degree of commoditization

A market needs a good or service that is traded (Begg et al., 2003; Ménard, 2005). In the case of PES, this requires the definition of ES units subject to trade, and implies measurable and quantifiable ES (Engel et al., 2008; Farley and Costanza, 2010). According to Muradian (2013), defining a clear ES and 'exchanging'

[13] Vatn (2010) distinguishes between markets, communities and governments, as different forms of governance. Here, we put communities together with individual households. This does not imply that we are oblivious of community governance, but in this analysis communities can be clearly identified as being providers or users.

this ES are the two steps necessary in the process of ES commoditization. A pure market transaction in PES would require a clearly defined ES that buyers (and users) are able to evaluate and obtain in measurable quantities, while a low level of commoditization denotes an interaction between actors that is not based on transacting a clearly defined ES (Muradian et al., 2010).

Earlier research observed that instead of being based on clearly measured ES, most PES schemes focus on land uses that are expected to provide ES (Farley and Costanza, 2010). As such, the commoditization of ES is incomplete, with payments tied to proxies (land uses) (Wynne-Jones, 2013).

For our analysis we distinguish several categories: (1) PES schemes where users pay a certain amount for a clear quantity of delivered ES (ES units); (2) schemes where payments are directly linked to a specific (defined) ES, but not related to a precise amount of ES (direct ES); and (3) schemes where the connection with ES is more general and not clearly defined (indirect ES). Furthermore, we distinguish whether payments to providers are based on input or output performance. In an input-based PES approach, payments are normally made on a per-hectare basis for land or land management activities that are assumed to deliver the desired ES (Sattler et al., 2013). Whereas output-based PES focus on measured results of ES provision (Wynne-Jones, 2013). ES units and output-based PES show a high degree of ES commoditization.

3.3.4. Voluntariness of participation

In a market, participation should be voluntary (Vatn, 2015).

User participation

Voluntariness of user participation in theory implies that those who participate at the demand side are willing to buy a given quantity of a good or service at a certain price. According to neoclassical economics, the voluntary participation in markets will maximize social utility and welfare (Korff, 2008). In a command system on the other hand, participation is based on obligation. In order to differentiate between different levels of participation, we categorize them according to whether participation is (1) voluntary, (2) compulsory, or (3) if it is a government that acts directly in representation of ES users.

van Noordwijk and Leimona (2010) state that voluntariness is based on the concept of free and informed choice at the level of the individual. However, it is very difficult to assess these assumptions on the basis of voluntariness. Thus, here we understand voluntariness in participation as no legal obligation (no legal system of command) to participate.

Schemes with voluntary user participation are often referred to as ‘user-financed’. The other two participation mechanisms are known as ‘government-financed’ schemes (Engel et al., 2008). Firstly, user participation can be compulsory. Command-based methods are regularly used to organise users’ participation in PES schemes (Farley and Costanza, 2010). Secondly, although governments are never the final users of the schemes analysed, they participate as users’ representatives. Governments depend on their ability to command to act as buyers in PES schemes (Vatn, 2015). In this case, users do not participate directly. We thus distinguish this participation mechanism from the previous compulsory participation.

Provider participation

Voluntary participation of providers is based on offering positive incentives to change land-use decisions and not to induce this change through regulations and administrative decisions. Voluntariness of provider's participation as a market characteristic is considered important to ensure that products or services are supplied at a price that producers are willing to accept and would thus not create an undersupply of the good or service (Jack et al., 2008). Pagiola et al. (2008) and Tacconi (2012) state that voluntariness of providers' participation is a key characteristic of PES. As with the user participation classification, classes are (1) voluntary, (2) compulsory, and (3) the government.

Voluntariness should (in theory) differ from providers being the object of compulsory limitations on their land-use decisions (van Noordwijk and Leimona, 2010). It is argued that PES should only be used to pay for practices that are additional to the law (ten Brink, 2011), or conversely that PES are a more cost-effective way of obtaining an environmental outcome than by regulatory approaches (Wunder, 2008). Yet, regulations already exist that directly or indirectly provide or conserve ES (Engel et al., 2008). Governments limit the ability of landholders to choose freely what they can do with their land. For cutting down trees it is often necessary to obtain a license, while in some areas with high biodiversity or in watersheds deforestation can be forbidden by law (Arriagada, 2009; Casas, 2008; Echavarria et al., 2004). Sommerville et al. (2009, p. 2) state "The one criterion that we believe is not crucial to all PES interventions is that they must be voluntary. Although we agree that PES are voluntary at the level of the transaction (i.e., service providers can decide whether or not to accept payment), service providers do not necessarily have the choice whether or not to provide the service, such as in cases where land-use change is illegal.".

It is thus difficult to separate regulations and the importance of administrative decisions from the incentive impact of payments. As both coincide, a hybrid mechanism emerges that combines both, but not one that solely relies on high incentive intensity. According to Laurans et al. (2012), in certain schemes payments are meant to persuade landholders to stop committing illegal acts, thereby countering, to a certain extent, any failures of regulatory approaches. This implies that landholders get paid for obeying the law (Engel et al., 2008). Wunder (2007) states that in many cases it may be rational to use the stick-and-carrot approach (i.e. to supplement weakly enforced laws with PES compensations), especially when top-down protection declarations had been unfair to local users in the first place. When compensation is the idea, then issues of fairness and equity come into play (Wunder, 2008). What you don't have is a market mechanism that provides conservation incentives. It is then a compensation mechanism which aims at making the implementation of regulations more acceptable (Pagiola, 2008). Engel et al. (2008) and Wunder and Alban (2008) try to separate PES from environmental regulation by mentioning that there is an interaction between

PES and command-based measures. If the law had been effectively enforced from the beginning, there would have been no need for PES, at least not for environmental purposes. However they add that even modest law-enforcement efforts can stimulate greater participation of providers by decreasing expected return from alternative, illegal land uses, thus making PES participation more attractive. This makes a full separation between the positive incentive aspect of PES and the command part impossible, which is key to the distinction between participation in PES and regular markets, even when PES participation is voluntary. As Muradian (2013) notes, incentives are one of the numerous possible coordination mechanisms that can emerge in hybrid structures, exactly because they mix monetary signals and control elements.

Here we focus on voluntary versus (command-based) compulsory participation in the scheme, not the existence of land-use (and other) regulations. As with users, even when participation in a PES scheme is voluntary, particular subgroups may be forced to participate (Sommerville et al., 2009).

Following the above, we understand compulsory participation as the legal obligation to participate in a scheme. The latter should not be confused with the obligation to abide to existing environmental regulations. Finally, although there is an ongoing discussion whether public land should be eligible for PES (Sattler et al., 2013), governments could potentially participate as providers. Participation in the scheme can be voluntary, but what allows to distinguish governments from participation by the other actors, is the role governments play as intermediaries (see Section 3.4.5) and as implementers of land-use regulations.

3.3.5. Payment setting mechanism

Markets coordinate decisions between actors using information provided through the pricing system (Begg et al., 2003). For PES, this would imply that – ideally - payments should be determined through demand and supply interactions (Van Hecken and Bastiaensen, 2010).

Users' payment mechanisms

We distinguish between (i) possible voluntary means of establishing users' payment levels (bilateral agreements and payment levels set by intermediaries); (ii) payments set and obtained through command (earmarked taxes/tariffs); (iii) and internal determination of payments (government budget allocation).

We apply the category of bilateral agreements when payments are negotiated between users and providers or intermediaries. This could typify small markets from the users' side. We understand that the price users pay is set by an intermediary when intermediaries offer an ES (product) for purchase at a specific price. Carbon credits are a typical example. This is a market mechanism based on a supply side offer, with no obligation behind the purchase of the ES[14]. Earmarked taxes/tariffs are obligatory payments that are set specifically to finance PES. They are based on command, and do thus not characterize a market from the

[14] In this analysis the credits are (were) sold on the voluntary carbon market.

users' side. These taxes/tariffs are however linked to the quantity of the good or service consumed. For example in Costa Rica's PSA, the more gasoline one purchases, the more one will contribute to the PES scheme through an earmarked tax and thus the more one will contribute to financing the scheme, and to the provision of ES such as carbon sequestration. Similarly, for environmental tariffs used in watershed schemes: the more water consumed, the more one will contribute to the maintenance or improvement of watershed services; this works as a command-based mechanism directly related to demand. Government budget allocation is established internally and through administrative control, characterizing a hierarchy. Government budget allocation is a user payment mechanism when a government specifically assigns a budget to pay for ES provision through a scheme, but this is not directly linked with ES use levels. We did not include government budget used to cover management costs, or additional costs governments can bear by being intermediaries.

Providers' payment setting mechanism

The underlying assumption of supplier price setting through the market is that markets can reveal hidden information, leading to the most efficient provider being chosen or setting the price for ES provision (Arriagada and Perrings, 2009). As no marketplace exists where providers can offer their ES for sale, market price setting in PES could entail the creation of auctions (Börner et al., 2010; Ferraro, 2008). However, in none of the cases studied the price paid to providers resulted from auctioning schemes.

Similar as for users' payment setting mechanisms, providers' payment levels can be the result of bilateral agreements between providers and users or intermediaries. These negotiations can reflect the willingness to accept of providers to change land use(s). When payments are set top down by the intermediary/user, and participation is voluntary, prices again should reflect ES providers' willingness to accept. However, as prices are set top down there is no coordination through the price system. Vatn (2010, p. 1247) writes about this as "cases where the intermediary is the only really active party in defining the price, and where the price is a flat rate it is still a payment mechanism, but not a standard market". Finally, internal budget allocation is a hierarchical governance system (Vatn, 2015).

3.3.6. Conditionality

For an exchange to occur through an ES market, property rights could be established (e.g. carbon credits), this is however not a prerequisite. Markets of many services do not exchange property rights (Wunder, 2015), but payment and service delivery should be conditional. Conditionality in PES is mostly related to the performance of providers. According to Wunder (2012, p. 2) "the arguably decisive PES criterion is whether providers receive conditional payments". Payments should be conditional on whether ES are provided or not (Martin-Ortega et al., 2013). If providers fail to deliver the ES as agreed then payments should be withdrawn or reduced (Milne and Niesten, 2009). It is this conditionality that is considered the fundamental difference between direct payments and activities of another approach, still implemented, and referred to as the generic family of integrated conservation and development projects (ICDPs) (Engel et al., 2008; Milne and Niesten, 2009; Wunder, 2012). In ICDPs incentives are delivered up-front with no further conditionality attached to them. There is thus no fixed mechanism to assure that exchange will take place. Because of conditionality, PES schemes are considered an improvement over the ICDP approach (Ferraro and Kiss, 2002).

Payments can only be conditional on ES if these can be monitored, while also sanctions for non-compliance should be implemented (Sommerville et al., 2011; Wunder et al., 2008). Input-based PES programmes normally divide monitoring on the one hand into monitoring whether the applied land use is actually generating the targeted ES (see Section 3.5.1.), and on the other hand monitoring whether providers are complying with the contractually agreed upon land uses (Engel et al., 2008). As data on non-compliance and sanctions were not provided by the literature review or the experts, it was not possible to obtain sufficient answers on conditionality so as to assess to what degree conditionality was implemented and whether this changed over time. However, as a proxy for conditionality, we assess the use of contractual agreements with the providers, as contracts should stipulate under what conditions (e.g. implementing specific land uses) payments can be received, i.e. the conditions for exchange. Conversely the payments (monetary or in-kind) of the schemes can be non-conditional, ICDPs[15] in this case.

3.3.7. Role of intermediaries

The main role of intermediaries in a market should be to facilitate exchange (Ahn et al., 2011). For PES, a distinction can be made between 'financial intermediaries' (buyers), who collect funds from various sources and allocate them to providers; and 'management intermediaries', which take care of a scheme's implementation (Laurans et al., 2012).

Intermediaries can consist of one single intermediary (e.g. a government agency), several constituents of the same intermediary organisation, or several levels of intermediaries (e.g. a local intermediary implementing the scheme and a national or international intermediary managing the scheme's funding). In PROFAFOR the financial intermediary sold the carbon credits outside of Ecuador, while the management intermediary operates within Ecuador. In *Scolel Té*, credits are sold by an international organisation (Plan Vivo), while the project is administered by the *Fondo Bioclimático* (a non-profit trust fund) and coordinated by AMBIO, a Mexican NGO. In FORAGUA there is the water fund with its management, while the municipalities that are the fund's constituents collect water charges and implement

[15] Although ICDPs could be considered as characterizing non-PES mechanisms, they are still included in this analysis. This is only one of the multiple characteristics used to understand the diversity of hybrid governance structures found in schemes that have been discussed or interpreted as PES. The latter includes schemes that implement ICDPs (e.g. Goldman-Benner et al., 2012; Kauffman, 2013).

activities related to watershed conservation financed through the fund. The latter is similar to the Colombian water fund, where the fund manages the resources, but it are the fund's members that implement activities. The other schemes do not have these two clearly separated intermediary levels. Although schemes such as PSA and Socio Bosque receive aid from NGOs to facilitate signing agreements with providers.
Finally, intermediaries could potentially act as providers. In this case the role of the intermediary merges with that of a provider, i.e. it becomes a hierarchical characteristic.

In summary, in a pure market set-up we would expect that ES trade would take place when a commoditized ES is voluntarily produced by private providers in return for a price negotiated with private users. The payment will be conditional on the level of ES production and intermediaries would only act to facilitate the interaction between users and providers. Conversely, in a set-up based on hierarchical coordination, in our case a government bureaucracy, the budget would come solely from government revenue not directly related to the level of ES use by users. The budget would be allocated based on internal budget distribution mechanisms to government-managed land and government-funded projects, whereby funding for projects is not conditional on ES delivery. Hybrids will include a mix of characteristics that will put them in between these two extremes.

3.3.8. Different categories of schemes

Key reasons for the hybrid structure of PES have previously been identified in the literature. We thus incorporated them to analyse to what extent they provide an explanation of the mode of governance and the observed changes in schemes' configuration (Figure 3.1). These reasons are (1) the size of the user and provider groups; (2) the ES involved; and (3) resistance against PES based on the idea that it is a market-based approach.
Group Size: when only a small number of actors are involved in a scheme, transaction costs of coordination are relatively low (Engel et al., 2008). The more users and/or providers a scheme has, the higher the transaction costs of organising them and thus the higher the costs of using a market mechanism (Vatn, 2010). The complexity of organising large numbers can often not be addressed through markets, so governments step in, using instruments such as taxes, or by acting directly on behalf of users and/or providers (Vatn, 2015, 2010). Hence, the larger the size of the user or provider group, the less market characteristics a scheme should have. We classify schemes in three categories of group sizes: (i) small (when there are only a few and clearly identifiable users or providers); (ii) medium (users or providers are

not easily identified individually, but can be found within a specific area within a country); (iii) large (users or providers are not easily identified, and can be found nation-wide[16]) (Table 3.2).

Characteristics of ES as economic goods: ES are inherently difficult to commoditize (Muradian and Gómez-Baggethun, 2013). The logic behind market creation for ES is that the economic good aspects of rivalry and excludability are changed through the creation of new institutions (OECD, 2004). However, according to Farley and Costanza (2010), rivalry is a purely physical characteristic that depends on the service, and not a dynamic variable that can be changed, while excludability can be a dynamic policy variable, but some ES are inherently non-excludable. Only certain ES, such as carbon sequestration, have the possibility of being commoditized and sold through a market. Biodiversity and landscape beauty are public goods, while watershed services are often characterized as club goods (Engel et al., 2008) or as common-pool resources (Fisher et al., 2010), which can be considered as non-market goods (Hagedorn, 2008). Moreover, the provision of ES has a high level of uncertainty and fluctuates over time (Ascough et al., 2008), while ES are often provided by ecological functions that have a high level of complexity (Cardinale et al., 2012; Norgaard, 2010). Schemes are classified according to the focal ES considered in the scheme (Table 3.1). Schemes that focus on more than one ES are grouped together and classified as schemes focusing on bundled ES (Table 3.2). We would expect that carbon schemes are closer to markets, whereas schemes that focus on biodiversity should rather be organised through governments. Watershed schemes should lie in between these two forms of organisation.

Denomination: in Latin America, there exists an opposition to PES based on the belief that PES need monetary valuation of ES (although it does not e.g. Wunder, 2013). PES are considered a means to achieve the commoditization of nature, and are seen as mechanisms for indirect privatization (Balvanera et al., 2012; Southgate and Wunder, 2009). Resistance against the PES concept can thus be a factor that influences PES (or PES-like) scheme's design and how it evolved over time. As a proxy for resistance against PES, we classify schemes according to whether the schemes themselves use the term PES. We would expect that schemes that do not consider themselves PES are less market-oriented.

León et al. (2010, p. 9), when discussing the process of setting up the Alto Mayo scheme in Peru, mention that "the terminology used to describe the mechanism has changed as a result of the process, from PES to compensation for ecosystem services. The reason is the negative perception of the term 'payment'…" Also, the concept of reciprocal watershed agreements (RWA) creates a shift in conceptualization based on how the term PES is perceived. Although RWA schemes are not necessarily different from PES designs, according to Martinez et al. (2013, p. 43), "RWAs are based on Wunder's (2007) principles of PES , but with a focus on local norms of reciprocity amongst upstream and downstream

[16] In addition, there can be global ES users.

communities". This shift in discourse on schemes can be observed in literature about the well-known scheme in Los Negros, Bolivia. The scheme has been described as a market mechanism for watershed management (Asquith and Vargas, 2007) and as a PES scheme (Asquith et al., 2008), while now rather the differences between RWA and PES are highlighted (Asquith, 2011). Also Socio Bosque does not define itself as PES (Ministry of the Environment, Ecuador, 2012a). A number of water funds, which have been analysed in the context of PES (Balvanera et al., 2012; Goldman-Benner et al., 2012; Kauffman, 2013), use the term water fund as well as compensation for ecosystem services (CES), the latter referring to contractual agreements with individual providers. Finally, NK-CAP was originally conceptualized as a PES (Pereira, 2010), but now undergoes changes following resistance against what is perceived as the commoditization of nature in Bolivia (Bryner et al., 2012).

Other schemes, however, do (continue to) use the term PES to describe what they are. In the cases analysed in Andean countries, only the Pimampiro scheme uses the PES term (Municipality of Pimampiro, 2013). In Mesoamerica on the other hand, all schemes use the PES denomination (Table 3.2).

Table 3.2: Categorization of the schemes according to size, ES and denomination

Case name	Users' group size	Providers' group size	ES	Denomination
Alto Mayo Water Initiative	Medium	Medium	Watershed	Alternative
ESPH –PROCUENCAS	Medium	Medium	Watershed	PES
FONAG	Medium	Medium	Watershed	Alternative
FORAGUA	Medium	Medium	Watershed	Alternative
NK-CAP[a,b]	Small	Few	Carbon	Alternative
PES La Esperanza	Small	Few	Watershed	PES
PES Pimampiro	Medium	Medium	Watershed	PES
PES San Pedro del Norte	Medium	Medium	Watershed	PES
PHES Gil Gonzalez	Small	Medium	Watershed	PES
Plan Vivo Scolel Té	Small	Medium	Carbon	PES
PROFAFOR[b]	Small	Large	Carbon	PES
PRONAFOR	Large	Large	Bundled	PES
PSA	Large	Large	Bundled	PES
RWA Los Negros	Medium	Medium	Watershed	Alternative
Socio Bosque Programme	Large	Large	Bundled	Alternative
Water Fund for Life and Sustainability	Medium	Medium	Watershed	Alternative

3.4. Method

In our analysis, for each characteristic, the number of PES (or PES-like) schemes that include the specific class (as identified in 3.3.4 to 3.3.7) initially and/or currently were summed. A comparison could thus be made of the number of schemes that incorporated a specific class of one market, hybrid or hierarchy characteristic, and the overall evolution of the schemes' design. One scheme can contain several classes, and thus be summed several times for each characteristic. Consequently, the sum of all classes can be higher than the total number of schemes analysed.

As a first step we analysed the initial design and changes for all PES schemes and compared the number of schemes that incorporate specific classes of a characteristic. Cells in the tables below where changes occur are filled in grey. We analysed the significance of these changes using Pearson's Chi-square tests (Agresti, 2007). This was followed by an analysis of the number of schemes per category (see 3.3.8) and between categories that incorporate the different classes of each characteristic. The data on each scheme separately can be found back in the supplementary tables of Annex 4.

3.5. Results

3.5.1. Degree of commoditization

Overall degree of ES commoditization

In only a few schemes commoditized ES (ES units) can be purchased by users (Table 3.3). The number of schemes that include ES units has slightly decreased, as in two carbon credits are no longer sold. In one scheme ES units have now been incorporated. Overall, these changes are not statistically significant. Most schemes used, and continue to use direct ES, while one more scheme added direct ES to its scope. There is thus overall a low level of commoditization of users' ES.

If we look at the providers' side, we can observe that in their initial design most schemes were input-based. This focus on inputs rather than outputs has not changed. Only one scheme (NK-CAP) changed from being output-based to input-based, decreasing its degree of commoditization. Also here changes are not significant.

Table 3.3: Degree of ES commoditization for all schemes

		Initially		Currently		Change	Chi-Square
		N	*% of schemes*	N	% of schemes		
User side	ES Units	*4*	*25*	3	19	↓	0.5780
	Direct ES	*10*	*63*	11	69	↑	
	Indirect ES	*3*	*19*	3	19	↔	
Provider side	Output-based	*3*	*19*	2	13	↓	0.3199
	Input-based	*13*	*81*	14	88	↑	

Degree of ES commoditization for users per category

When the degree of commoditization of ES is analysed according to the categories of user group size, we observe changes in schemes with a small user group (two schemes less incorporate ES units - carbon credits). There are also changes in schemes with large groups of users, focusing on bundled ES, through the inclusion of direct ES and ES units. Finally, while ES units are no longer used in alternatively denominated schemes, they are still part of some PES-denominated schemes. The latter category also includes one additional scheme using direct ES (Table 3.4).

In general, in the schemes with small user groups, users pay for ES units and direct ES. Schemes with medium-sized user groups focus specifically on direct ES. When the group of users is large, the scope are mostly indirect ES. The latter are the schemes considering bundled ES. Carbon schemes focus on ES units, while most watershed schemes incorporate direct ES. Finally, schemes with an alternative denomination do not use ES Units.

Degree of ES commoditization for providers per category

The only change that is observed when analysing providers' ES, is a switch of a small (previously) carbon scheme that was denominated PES (NK-CAP), from output-based to input-based (Table 3.4).

Most schemes focus on input-based schemes irrespective of the size of the provider group, the ES targeted, or their denomination. The only two schemes that are currently output-based are both termed 'PES'. One is a carbon scheme with a medium-sized group of providers, the other a watershed service scheme with a single provider.

Table 3.4: Degree of ES commoditization per category of schemes

		Users' ES									Providers' ES					
		ES Units			Direct ES			Indirect ES			Output-based			Input-based		
	[a]	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch	*I.*	C	Ch	*I*	C	Ch
Group size	Small	*4*	2	↓	*1*	1	↔	0	0	↔	*2*	1	↓	*0*	1	↑
	Medium	*0*	1	↑	*8*	8	↔	0	0	↔	*1*	1	↔	*9*	9	↔
	Large	*0*	0	↔	*1*	2	↑	3	3	↔	*0*	0	↔	*4*	4	↔
ES	Carbon	*3*	1	↓	*0*	0	↔	0	0	↔	*2*	1	↓	*1*	2	↑
	Watershed	*1*	1	↔	*9*	9	↔	0	0	↔	*1*	1	↔	*9*	9	↔
	Bundled	*0*	1	↑	*1*	2	↑	3	3	↔	*0*	0	↔	*3*	3	↔
Denomination	PES	*3*	3	↔	*6*	7	↑	2	2	↔	*2*	2	↔	*7*	7	↔
	Alternative	*1*	0	↓	*5*	5	↔	1	1	↔	*1*	0	↓	*6*	7	↑

[a] *I: Initially;* C: Currently; Ch: Change

3.5.2. Voluntariness of participation

Results user participation

The changes observed in the voluntariness of user participation are statistically significant. There is a small decrease in voluntary user participation, through a decrease of NGOs acting as buyers[17]. Although two more schemes incorporated voluntary participation of businesses, in two other schemes voluntary participation of these actors disappeared (Table 3.5). Since their implementation five schemes already relied on non-voluntary participation. This number has now doubled. The main source of funding for PES were and are governments. Overall, there has been a clear increase in command-based users' participation mechanisms.

Table 3.5: Voluntariness of users' participation for all schemes

		Initially		Currently		Change	Chi-Square
		N	*% of schemes*	N	% of schemes		
Voluntary	NGO	*8*	*50*	6	38	↓	0.0046***
	Household	*2*	*13*	2	13	↔	
	Business	*7*	*44*	7	44	↔	
Compulsory	Household	*5*	*31*	10	63	↑	
	Business	*5*	*31*	10	63	↑	
Government		*10*	*63*	12	75	↑	

***Significant at 1%

Some differences can be observed when looking at how users participate in the schemes categorized according to group size (Table 3.6). When the group of users is small, voluntary participation, especially

[17] NGOs as voluntary buyers are only taken into account if they pay for ES provision, not management costs they can (additionally) bear.

of businesses, is most common. With the exception of two schemes that no longer receive financing through the sale of ES units, this has not changed. However, two schemes now also have the government directly participating. In medium-sized schemes, a slight decrease in the number of NGOs acting as buyers is observed. The biggest change here is the sharp increase in compulsory direct participation. In the three schemes with a large user group, national governments are the buyers. Additionally, in two of them compulsory user participation is used. In these schemes, an increase in the diversification of participation mechanisms, through an increase in voluntary mechanisms, can be observed.

Schemes with large user groups are those which target bundled ES. Two of the carbon service schemes disappeared, but the one remaining is still mainly based on voluntary participation of private entities. However, this scheme has diversified the users' participation mechanisms employed. The strong growth in compulsory participation is entirely due to an increase of command-based methods in schemes focusing on watershed services. The other important funding source in these schemes remain governments acting directly as buyers. In addition, in four of the schemes businesses continue to participate voluntarily. The schemes with an alternative denomination see the strongest increase in compulsory participation and almost all of them (six out of seven) have governments operating as buyers. This is slightly lower in the other category. In addition, households never participate voluntarily in the alternatively denominated schemes.

Table 3.6: Voluntariness of users' participation per category

		Non-Government						Government		
		Voluntary			Compulsory					
	[a]	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch
Group size	Small	*5*	3	↓	*0*	0	↔	*1*	2	↑
	Medium	*6*	4	↓	*3*	8	↑	*6*	7	↑
	Large	*1*	2	↑	*2*	2	↔	*3*	3	↔
ES	Carbon	*3*	1	↓	*0*	0	↔	*0*	1	↑
	Watershed	*8*	6	↓	*3*	8	↑	*7*	8	↑
	Bundled	*1*	2	↓	*2*	2	↔	*3*	3	↔
Denomination	PES	*7*	4	↓	*4*	5	↑	*5*	6	↑
	Alternative	*5*	5	↔	*1*	5	↑	*5*	6	↑

[a] *I: Initially;* C: Currently; Ch: Change

Results provider participation

The participation mechanisms of providers did only change slightly, and this change is not statistically significant (Table 3.7). Schemes are mainly based on voluntary participation of households (and communities). Governments and NGOs participate in some schemes as providers through protected areas, while two more schemes now incorporate businesses as providers. However, the latter are water users, part of the intermediary, and purchased land in watersheds (see also Section 3.4.5.).

Table 3.7: Voluntariness of providers' participation for all schemes

		Initially		Currently		Change	Chi-Square
		N	*% of schemes*	N	% of schemes		
Voluntary	NGO	*3*	*19*	2	13	↓	0.1868
	Household	*15*	*94*	15	94	↔	
	Business	*2*	*13*	4	25	↑	
Compulsory	Household	*0*	*0*	0	0	↔	
	Business	*0*	*0*	0	0	↔	
Government		*5*	*31*	5	31	↔	

Comparing schemes according to the different categories, we observe that an NGO is no longer a provider in a PES-denominated carbon scheme with a medium-sized provider group. In two medium-sized watershed service schemes (one PES and one alternatively denominated) businesses are now operating as providers. Moreover, there is one less watershed scheme and one more carbon scheme, both PES-denominated with medium-sized provider groups, where the government now acts as a provider (Table 3.8).

There are currently only two schemes with a small group of providers. One has an NGO as a provider, the other a government. In schemes with large and medium-sized provider groups, providers are mostly households and communities. Governments participate as providers in schemes focusing on watershed services through municipal protected areas, and in two carbon schemes, also through protected areas. Governments never participate as providers in the schemes focusing on bundled services. Lastly, the biggest difference that can be observed between schemes when classifying them according to denomination, is that governments participate more as providers in schemes with an alternative denomination (four schemes), than in the PES-denominated schemes (only one).

Table 3.8: Voluntariness of providers' participation per category

		Non-Government						Government		
		Voluntary			Compulsory					
	[a]	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch
Group size	Small	*2*	2	↔	*0*	0	↔	*1*	1	↔
	Medium	*10*	10	↔	*0*	0	↔	*4*	4	↔
	Large	*4*	4	↔	*0*	0	↔	*0*	0	↔
ES	Carbon	*3*	3	↔	*0*	0	↔	*1*	2	↑
	Watershed	*10*	10	↔	*0*	0	↔	*4*	3	↓
	Bundled	*3*	3	↔	*0*	0	↔	*0*	0	↔
Denomination	PES	*9*	9	↔	*0*	0	↔	*1*	1	↔
	Alternative	*7*	7	↔	*0*	0	↔	*4*	4	↔

[a] *I: Initially;* C: Currently; Ch: Change

3.5.3. Payment setting mechanisms

***Results users' payment setting*[18]**

The users' payment setting mechanisms show results similar to those obtained when analysing users' participation. Also here changes are statistically significant. There is a strong increase in the use of

[18] Initially 10 schemes also received donations for ES provision, currently 9 schemes receive them directly.

earmarked taxes/tariffs to finance PES, and an increase in government budget allocation (Table 3.9). In addition, there is a small decrease in bilateral agreements, while one more scheme has incorporated the use of payment levels set by the intermediary.

Table 3.9: Users' payment setting mechanisms for all schemes

	Initially		Currently		Change	Chi-Square
	N	*% of schemes*	N	% of schemes		
Bilateral agreements	*10*	*63*	7	44	↓	0.0017***
Price set by intermediary	*1*	*6*	2	13	↑	
Earmarked Taxes/Tariffs	*5*	*31*	10	63	↑	
Government budget allocation	*10*	*63*	12	75	↑	

***Significant at 1%

Schemes with a small number of users show a decrease in the use of bilateral agreements. In one scheme a payment setting mechanism no longer exists (PROFAFOR), while the other one (NK-CAP) now relies on government budget allocation. These are two (former) carbon schemes. Schemes with a medium-sized user group show a decrease in the use of bilateral agreements and payment levels set by the intermediary. However, four schemes, which focus on watershed services, now additionally use earmarked taxes. One more of these schemes currently also relies on government budget allocation. For the schemes with a large user group (focusing on bundled services), two have incorporated the use of bilateral agreements and one payment levels determined by the intermediary. Moreover, we see that three watershed service schemes no longer use bilateral agreements, and two no longer payment levels set by the intermediary. PES-denominated schemes show a decrease in the use of bilateral agreements and of intermediaries determining payment levels, and a small increase in earmarked taxes and government budget allocation. Alternatively named schemes on the other hand show a small increase in the number of schemes using payment levels set by intermediaries, a strong increase in earmarked taxes, and a small increase in government budget allocation (Table 3.10).

Overall, for user payment setting, when group size is small, payments are determined through bilateral agreements. However, additional payment setting mechanisms are used, with the exception of earmarked taxes. Conversely, in all schemes with medium-sized user groups focusing on watershed services, earmarked taxes are used, followed by government budget allocation. The latter payment setting mechanism is the most important for the schemes with a large user group (bundled ES). However, these schemes use a diverse set of payment mechanisms, including also bilateral agreements and earmarked taxes/tariffs. Furthermore, carbon schemes have never incorporated earmarked taxes as a financing mechanism. There are no big differences between schemes classified by denomination. Except, as was observed previously, an 'ES product' offered at a price set by the intermediary is sold by two schemes that use the PES term.

Table 3.10: Users' payment setting mechanisms per category

	[a]	Bilateral agreements			Price set by intermediary			Earmarked taxes/tariffs			Government budget		
		I	C	Ch	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch
Group size	Small	*5*	3	↓	*2*	2	↔	*0*	0	↔	*1*	2	↑
	Medium	*4*	2	↓	*6*	4	↓	*3*	8	↑	*6*	7	↑
	Large	*0*	2	↑	*2*	3	↑	*2*	2	↔	*3*	3	↔
ES	Carbon	*2*	1	↓	*1*	1	↔	*0*	0	↔	*0*	1	↑
	Watershed	*7*	4	↓	*7*	5	↓	*3*	8	↑	*7*	8	↑
	Bundled	*0*	2	↑	*2*	3	↑	*2*	2	↔	*3*	3	↔
Denomination	PES	*7*	4	↑	*6*	4	↓	*4*	5	↑	*5*	6	↑
	Alternative	*3*	3	↔	*4*	5	↑	*1*	5	↑	*5*	6	↑

[a] *I: Initially;* C: Currently; Ch: Change

Results providers' payment setting

There is only a small change in how providers' payments are set, due to the change in the structure of one scheme (NK-CAP), which shifted from relying on bilateral agreements to internal budget allocation. Overall, the observed change is not statistically significant. In general, most of the schemes analysed rely on bilateral agreements followed by top-down payment setting, while some schemes additionally use internal budget allocation (Table 3.11).

Table 3.11: Providers' payment setting mechanisms for all schemes

	Initially		Currently		Change	Chi-Square
	N	*% of schemes*	N	% of schemes		
Bilateral agreements	*10*	*63*	9	56	↓	0.5418
Price set by intermediary	*6*	*38*	6	38	↔	
Government budget allocation	*3*	*19*	4	25	↑	

If we observe the different categories, we see small changes due to change in a carbon scheme with only one provider that is not named PES. This scheme switched from using bilateral agreement to internal budget allocation (Table 3.12).

Overall, in schemes with large provider groups, only payment levels set by the intermediary are used. Whereas in the other two categories, bilateral agreements are the most important mechanism to set providers' payment levels. Although some rely on internal budget allocation, bilateral agreements are by far the most important payment setting mechanism used in watershed service schemes. In carbon schemes intermediaries determine providers' payments. The more hierarchical nature of many of the schemes that do not use the PES denomination can be observed by their exclusive use of internal budget allocation. However, while in these schemes bilateral agreements with providers are more important, for the PES-denominated schemes it are payments set by the intermediary

Table 3.12: Providers' payment setting mechanisms per category

		Bilateral agreements			Price set by intermediary			Government budget		
	[a]	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch
Group size	Small	*2*	1	↓	*0*	0	↔	*0*	1	↑
	Medium	*8*	8	↔	*2*	2	↔	*3*	3	↔
	Large	*0*	0	↔	*4*	4	↔	*0*	0	↔
ES	Carbon	*1*	0	↓	*2*	2	↔	*0*	1	↑
	Watershed	*9*	9	↔	*1*	1	↔	*3*	3	↔
	Bundled	*0*	0	↔	*3*	3	↔	*0*	0	↔
Denomination	PES	*4*	4	↔	*5*	5	↔	*0*	0	↔
	Alternative	*6*	5	↓	*1*	1	↔	*3*	4	↑

[a] *I: Initially;* C: Currently; Ch: Change

3.5.4. Implementation of conditionality

No changes in the use of contracts and/or ICDPs were found, so their use is given only once (Table 3.13). If a contractual agreement with providers (conditionality) is the key characteristic of PES schemes, than most schemes fulfil this criterion. Three schemes analysed can then not be considered PES, because they do not use contracts with providers. Three other schemes would then have a PES component (contracts) within their broader design (including also non-contractual payments, ICDPs).

Schemes with a small or medium-sized group of providers use mostly contracts, although some implement ICDPs (Table 3.13). Schemes with large providers' groups use only contracts. The category of schemes focusing on watershed services (and one carbon scheme) do include schemes that implement ICDPs. Schemes focusing on bundled services never implemented ICDPs. Finally, it are only those schemes that do not use the PES term that execute ICDPs.

Table 3.13: Use of contracts and ICDPs by category

		Total		Contracts with providers		ICDPs	
		N	% of schemes	N	% of schemes per category	N	% of schemes per category
All schemes		16	100	13	81	6	38
Group size	Small	2	12.5	1	50	1	50
	Medium	10	62.5	8	80	5	50
	Large	4	25	4	100	0	0
ES	Watershed services	10	62.5	8	80	5	50
	Carbon services	3	18.8	2	67	1	33
	Bundled Services	3	18.8	3	100	0	0
Name	PES	9	56	9	100	0	0
	Alternative	7	44	4	57	6	86

3.5.5. Results role of intermediaries

Overall, the change in intermediaries and their roles is statistically significant (Table 3.14). We observe a decrease in non-government financial and management intermediaries. One more scheme now has a government acting as a financial intermediary, and three more have governments operating as management intermediaries. Furthermore, three additional schemes now have the intermediary (or one of its constituents) as a provider.

Only one scheme has no intermediary (La Esperanza). In most schemes governments assume the role of intermediary or form part of the intermediary body. Additionally, in four schemes the intermediary or one of its constituents became a provider through land purchases (establishment of reserves). According to Rojas and Aylward (2003), buying land can be distinguished from PES based on its degree of permanency. The authors state that buying land is more an investment in ES than a payment. The purchase of land makes the intermediary the provider. It creates an internal organisation (hierarchy) as opposed to contracts with (external) providers. In addition, the intermediary may also be the buyer, creating an almost complete hierarchical structure.

Table 3.14: The role of intermediaries in all schemes

		Initially		Currently		Change	Chi-Square
		N	*% of schemes*	N	% of schemes		
Non-Government	Financial	*6*	*38*	4	25	↓	0.0021***
Intermediary	Management	*11*	*69*	9	56	↓	
Government	Financial	*10*	*63*	11	69	↑	
Intermediary	Management	*7*	*44*	10	63	↑	
Intermediary as:	Buyer	*11*	*69*	11	69	↔	
	Provider	*1*	*6*	4	25	↑	

***Significant at 1%

When we compare the different categories, we observe that two schemes with small provider and user groups no longer have a non-governmental financial intermediary (Table 3.15). Two schemes with medium-sized user groups no longer have a non-governmental management intermediary, while one more scheme of this category now has a government acting as financial intermediary. In addition, one more scheme with a small, and two more schemes with medium-sized groups of users now have a governmental management intermediary. Furthermore, in schemes with medium-sized user groups there is an increase of intermediaries acting as providers. When we look at the categories according to the size of the provider group, we observe the disappearance of the non-governmental financial intermediary in one scheme with a small, and one with a large number of providers. The other observed changes are all for schemes with medium-sized groups of providers, and are equal to those found for schemes with medium-sized user groups. The disappearance of the sale of carbon credits in two schemes is the reason for the disappearance of two non-governmental financial intermediaries in small sized schemes. The other observed differences are changes in the structure of medium-sized watershed schemes.

In general, in schemes with a large group of users, governments are the main intermediary. In schemes with a small group this is the opposite, as there it are non-governmental intermediaries (businesses and NGOs). In schemes with a medium-sized user group, both government and non-government organisations are part of the intermediary body. Furthermore, only schemes with medium-sized user and provider groups have intermediaries as providers. These are schemes that focus on watershed services, and purchase or have purchased land in watersheds that are important for water provision and/or in seriously degraded watersheds. In carbon schemes only non-government entities play a role as intermediaries, and never as buyer or provider. When services are bundled, then the intermediary, which for the three schemes analysed is a national government agency, always takes up the role of buyer. When comparing PES schemes and alternatively denominated schemes, no major differences are observed, although in the latter schemes governments act more as intermediaries, and intermediaries also operate more often as providers. The alternatively denominated schemes show thus more often a hierarchical structure.

Table 3.15: The role of intermediaries per category

		Non-Government intermediary						Government intermediary						Intermediary as					
		Financial			Management			Financial			Management			Buyer			Provider		
	[a]	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch	*I*	C	Ch
User group size	Small	*3*	1	↓	*4*	4	↔	*1*	1	↔	*1*	1	↔	*1*	1	↔	*0*	0	↔
	Medium	*3*	3	↔	*7*	5	↓	*6*	7	↑	*3*	6	↑	*7*	7	↔	*1*	4	↑
	Large	*0*	0	↔	*0*	0	↔	*3*	3	↔	*3*	3	↔	*3*	3	↔	*0*	0	↔
Provider group size	Small	*1*	0	↓	*1*	1	↔	*0*	0	↔	*0*	0	↔	*0*	0	↔	*0*	0	↔
	Medium	*4*	4	↔	*9*	7	↓	*7*	8	↑	*4*	7	↑	*8*	8	↔	*1*	4	↑
	Large	*1*	0	↓	*1*	1	↔	*3*	3	↔	*3*	3	↔	*3*	3	↔	*0*	0	↔
ES	Carbon	*3*	1	↓	*3*	3	↔	*0*	0	↔	*0*	0	↔	*0*	0	↔	*0*	0	↔
	Watershed	*3*	3	↔	*8*	6	↓	*7*	8	↑	*4*	7	↑	*8*	8	↔	*1*	4	↑
	Bundled	*0*	0	↔	*0*	0	↔	*3*	3	↔	*3*	3	↔	*3*	3	↔	*0*	0	↔
Denomination	PES	*3*	2	↓	*6*	4	↓	*5*	5	↔	*3*	5	↑	*6*	6	↔	*0*	1	↑
	Alternative	*3*	2	↓	*5*	5	↔	*5*	6	↑	*4*	5	↑	*5*	5	↔	*1*	3	↑

[a] *I: Initially;* C: Currently; Ch: Change

3.6. Discussion

3.6.1. Overall change

In line with Schomers and Matzdorf (2013), the schemes we analysed focus mostly on direct ES and are input-based. Over time there has been a strong increase in the use of command-based methods to organise users' participation and set payment levels. From the providers' side the initial, more market-oriented goevrnance, with voluntary participation and payments organised through bilateral agreements or set by the intermediary has remained. Governments were already an important actor, but their role has increased over time, not only from the users' side, but also as management intermediaries, and in some cases even as providers. The schemes analysed thus rely even more on government involvement. Although more characteristics from a hierarchy are observed from the users' side, and more characteristics of markets when analysing providers, additionally a wide variety of modes of governance are found back. Most schemes now have a more complex organisation compared to their initial design. Finally, those schemes using contractual arrangements with providers and those that implement ICDPs still do so today.

Considering the ES, there have thus only been small changes in the low degree of commoditization. As measuring ES provision for users and of providers is costly, it is usually less expensive to focus on non-commoditized ES and land uses, instead of on actual ES provision (Wunder, 2015). In addition, according to Engel et al. (2008), for providers it can be easier to focus on inputs as the level of ES provision can often not be perceived by them. Another reason for the continued focus on direct ES and input-based schemes is the partial conservation orientation of most PES schemes. Schemes are often implemented (and managed) by conservation NGOs, and government departments with conservation goals. The main aim is to maintain or improve a specific land use (natural ecosystems), and not merely enhancing the provision of ES. As such also users continue to be linked with land uses through Direct ES, but normally not through ES Units.

In line with Pirard et al. (2010), our analysis confirms that, over the long term neither markets nor governments alone can ensure financial sustainability of payments. Government budget allocation (and donations) is an indication that markets do not create enough funds to pay for ES provision. While negotiations with interested service users (businesses) can also be a way to improve financial sustainability (Engel et al., 2008; Wunder et al., 2008), it seems that Latin American PES experiences are increasingly enhancing financial sustainability (and stability) through command-based participation and earmarked taxes/tariffs.

Providers' participation and payment level setting in the schemes analysed did not become more hierarchically organised. Although governments could use PES to finance protected areas (e.g. FAO, 2009) this is often not possible, due to legal restrictions and regulations on government financing.

Finally intermediaries are involved in all schemes analysed, with the exception of one. This could be explained by the fact that intermediaries can decrease transaction costs by reducing the number of actors involved as it can be very costly when each user has to coordinate with the providers (Pirard et al., 2010; Vatn, 2015). As intermediaries in the analysed schemes are most often government, we can argue, following Vatn (2010), that PES are primary a reconfiguration of the role of public bodies who are transformed mostly in intermediaries or buyers.

Additional explanations for the observed changes and differences of the schemes analysed can be found when analysing the schemes categorized according to group size, ES and denomination term used.

3.6.2. Changes per class

Group size

Small group size: Although of the five schemes with a small group of users only three still remain (Gil Gonzalez, Esperanza, and Scolel Té schemes), it can be observed that they were more market-oriented, and that this hasn't changed, except for NK-CAP. Small schemes also include two of the three schemes that focus on ES units, and one of the two remaining output-based schemes.

Medium group size: As group size increases, command-based participation and direct participation of governments, and consequently the use of earmarked taxes and government budget allocation become more important. These mechanisms have strongly increased the use of command mechanisms since the initial stages. Furthermore, almost all schemes focus on direct ES, and are input-based.

Given their local scale, it can be relatively easy to identify users, but there are too many to negotiate how to link payments to ES units. In addition, according to Vatn (2015, 2010), if transaction costs are high using markets, a command system such as a tax may be preferable. So although in their initial stage some of the schemes did not rely on taxes, financial sustainability and lower transaction costs provide an explanation for the increased use of command-based approaches in the schemes analysed with medium-sized user groups.

Also from the providers' side measuring ES on a large scale is not easily achieved, even when the relation between land use and ES is well understood. According to Kolinjivadi et al. (2014), as the number of land users that have an impact on ES provision increases, determining the provision of ES of individual land patches becomes increasingly difficult and costly. Thus focusing on input-based schemes is more economical from a measurement perspective. In the case of the Scolel Té scheme, ES measurement (and monitoring) is partially done by participating communities, decreasing the costs for the intermediary.

Large group size: The national schemes analysed have focused on bundled ES since their implementation. These schemes have large groups of users and providers. They rely strongly on governments from the users' side, but are more market-oriented from the providers' side. Although ES are normally incorporated as indirect ES, while their provision is input-based.

A large group makes linking users to specific ES more difficult. Moreover, when there is a large number of users, compulsory mechanisms for demand generation or government payments for provision will be necessary (Jack et al., 2008). However, the schemes have diversified the focal ES, and participation and users' payment setting mechanisms. As these schemes focus on a large (and diverse) user group, it is possible to obtain additional funding by including more direct ES for specific users. As such these schemes use government budget allocation, earmarked tariffs for watershed services, and voluntary business participation for ES units among others.

Although the large number of providers participate voluntarily, payments are not negotiated in these schemes, but set by the intermediary. Negotiating a price can be very costly when many providers participate. These high transaction costs influence payment setting, with intermediaries proposing the payment instead of using a market (Ferraro, 2011).

The schemes with small groups of actors remain more market-oriented (with the exception of NK-CAP), whereas schemes with medium-sized groups of users and providers evolved towards more hierarchical governance. The national schemes remain strongly government oriented.

In addition to group size, also trust has an impact on transaction costs. If trust between actors is high, transaction costs can be decreased (Bromiley and Cummings, 1995). When trust is low, a hierarchy can reduce transaction costs compared to markets. Trust in providers (fear of exit) has been a key reason for land purchases by the constituents of the water fund FORAGUA, as in the past some providers decided not to renew their contracts (Gordillo, 2013).

Ecosystem services

In addition, also ES have an impact on how schemes are organised.

Carbon schemes: The ES for which commodities are most easily created are carbon services, as clear measurement protocols exist (Wunder et al., 2008). Moreover, carbon credits are sold internationally, and there is thus a long geographical distance between users and providers that tends to make the cost of bureaucracy (hierarchical organisation) extremely high (Muradian, 2013). As users (buyers) are found partially or totally outside of national boundaries within which schemes are implemented, there is no authority that is able to implement taxes/tariffs on this trans-boundary level, and payment levels with the few users are thus negotiated or offered at a specific price. However, from the providers' side payments are not negotiated in the schemes analysed. Due to the international scale, negotiating payments with both users and providers can be very costly. Transaction costs can thus be lowered by setting the payments offered to providers.

Watershed services: Most watershed schemes focus on direct ES without going further in the commoditization process. These schemes have seen a strong increase in command-based methods to organise users, while there is a diversity of mechanisms from the providers' side. Some schemes (the water funds FONAG, FORAGUA and the Water Fund for Life and Sustainability) show a very strong hierarchical structure. In addition watershed schemes include most of the schemes that implement ICDPs.

For watershed services it is possible to identify users, but quantification of the level of ES they use is normally not done as measuring these is costly. Also for the providers, even when a measurement methodology exists (through software such as INVEST or SWAT, Goldman et al., 2010; Quintero et al., 2009), it can still be difficult for providers to actually manage land to obtain a certain level of ES (Engel et al., 2008). Focusing on inputs thus facilitates providers' participation. In addition, according to Wunder (2013, p. 3), for land uses in watersheds "it could also be that water users choose to follow a precautionary principle of widely protecting the pre-existing land cover that has worked well in providing services in the past".

Furthermore, as it is possible to identify water users, they can be organised to pay for service provision (Engel et al., 2008). According to Farley and Costanza (2010), water infrastructure can facilitate the establishment of market-like payment schemes for ES by facilitating exchange. However, infrastructure can be used both to facilitate the identification of specific buyers, but also facilitates compulsory participation in most watershed schemes. When it is difficult to commoditize ES, governments can thus force their citizens to pay for their provision through an additional tax/tariff**.** According to Vatn (2015), compared to markets, this command-based system can greatly reduce transaction costs.

Intermediaries operate as providers in watershed services schemes through land purchases of deteriorated land. This can be an interesting option when paying a private landholder for restoration activities is more expensive. Land purchases could thus in some circumstances lower transaction (and production) costs, and therefore sometimes be a lower cost alternative to PES contracts (Wunder, 2005).
The exception to the more hierarchical structure of watershed schemes (especially from the users' side) is La Esperanza. In this scheme payment levels are set bilaterally, and ES units are used. Payments are calculated through a formula based on the water user's ability to generate hydro-electrical power (Rojas and Aylward, 2002). An agreement that still continues to date. Here there is only one user-buyer and thus only one point where ES have to be measured (and only two actors to negotiate).

Bundled services: These schemes rely mostly on government budget allocation (and an earmarked fuel tax for Costa Rica's PSA) to organise users. In addition, these schemes can include specific payments for certain ES (in the cases analysed earmarked tariffs for watershed services). According to Engel et al. (2008), compulsory mechanisms or government payments will be required when incentives to free ride are high, as ES, such as water quality or biodiversity-related services, are often non-excludable (Wunder, 2015). Governments never participate as providers in the schemes analysed that focus on bundled services (national schemes), as financing and management of nationally protected areas is managed by different regulations than PES financing.

Among the cases analysed water service schemes never evolve towards market governance. Most schemes have actually increased the use of command-based governance over time. Bundled services have seen an increased diversification in schemes, but these three national schemes are mostly organised by the state. The biggest change in the carbon schemes comes from the disappearance of carbon credits as a financing source in two of the schemes. Even when commoditization is possible, there exists a certain level of resistance against this concept, especially in South America.

Denomination

The alternatively denominated schemes show in general less market orientation than the schemes that use the PES term. This can partially be explained by the resistance against ES markets in the countries where these schemes were implemented. In Bolivia, the original NK-CAP sold carbon credits, but this commoditization actually lead to the disappearance of the scheme's initial orientation. The Bolivian 'Law of Mother Earth' states that "All plans and programmes to reduce emissions of greenhouse gases, will focus on the non-commoditization of the environmental functions of the components of Mother Earth, so it will not include funding mechanisms associated with carbon markets" (Plurinational State of Bolivia, 2012). In the case of Ecuador, it is the Constitution that puts limits to the sale of carbon credits, as it states that "Environmental services shall not be subject to appropriation; their production, delivery, use and development shall be regulated by the State" (Art. 74 of the constitution, (Republic of Ecuador, 2008).

In addition, all ICDPs are implemented by those schemes analysed that do not use the PES term. As such the implementers of these schemes seem to agree with Wunder (2012) that implementing ICDPs should not be considered as PES. According to Kauffman (2013), in Ecuador ICDPs are used to overcome concerns about privatization and the commoditization of nature associated with PES schemes. In Bolivia it is the ‘Government Control Act’ that puts limits on many actions that could be included in PES, as it establishes that resources shall not be given directly to natural or juridical persons (Bryner et al., 2012).

Overall, schemes that do not denominate themselves as PES have increased the use of command-based governance and government involvement more than PES-denominated schemes.

3.7. Conclusions

Following Muradian et al. (2010, p. 1206) who consider that “a continuous classification is the most appropriate to describe the existing variety of PES schemes”, we used a multi-level framework based on market and command-based characteristics that was capable of incorporating the wide diversity of schemes implemented. The schemes analysed cover a range of hybrid arrangements going from small markets to almost complete hierarchical organisation. Over time schemes tend to incorporate command-based characteristics to organise users’ side, while voluntary contractual agreements with providers remain the core focus of the schemes. As such - within the existing land-use regulations - from the providers’ side in general a certain market orientation can be found back in the schemes analysed. From the users’ side we observed a diversification through the incorporation of a wide range of participation and payment setting mechanisms. Intermediaries are active in most schemes, with an increase in governments as intermediaries or as members of the intermediary body, and a decrease in the involvement of NGOs. Overall, since their initial implementation, schemes became more complex in their organisation, using a wider range of market and hierarchy mechanisms.

The schemes analysed with small groups of users and/or providers incorporate more market characteristics, while as group size increases, more command-based characteristics are used. In this analysis, carbon services are normally organised through more market-oriented mechanisms, while for watershed services a broad diversity of governance forms can be found. Bundled services rely mostly on governments. The way the schemes analysed evolve is to a certain extent in line with what Wunder (2015) considers the key defining criterion of PES, namely addressing an environmental externality through a payment, that is voluntary for providers and with payments conditional on pre-agreed land uses. As such, the main factor that distinguishes PES schemes from schemes using an alternative denomination is the incorporation of ICDPs and intermediaries as providers in the latter. These schemes are in general more hierarchically organised and never incorporate commoditized ES.

Chapter 4: Analysis of the cost-effectiveness for ecosystem service provision and rural income generation: a comparison of three different programmes in Southern Ecuador

Based on: Raes, L., Aguirre, N., D'Haese, M., Huylenbroeck, G.V., 2014. Analysis of the cost-effectiveness for ecosystem service provision and rural income generation: a comparison of three different programs in Southern Ecuador. Environ. Dev. Sustain. 16, 471–498. doi:10.1007/s10668-013-9489-2

Abstract

In recent years, new tools for funding nature conservation have been designed. Because poverty is often significant in areas with high biodiversity, the improvement of local livelihoods is frequently considered as a secondary goal of new financing mechanisms besides nature conservation. The buffer zone of the Podocarpus National Park in Ecuador is such a high biodiversity zone. In this chapter, we compare the cost-effectiveness and development potential of three different mechanisms to finance nature conservation implemented in this buffer zone, namely (a) an organic coffee label, (b) the Socio Bosque Programme, a nationwide payment scheme for private forest conservation, and (c) FORAGUA, a regional water fund. This chapter describes the functioning and the scope of the mechanisms and analyses their environmental and socio-economic impacts which are compared to the total costs. Results show that the water fund has the highest additionality in ecosystem service provision, while the payment scheme is the most cost-effective both for current as for increased ecosystem service provision and for extra rural job creation. Organic coffee certification has the highest positive impact on rural income creation.

4.1. Introduction

Ecosystem services (ES) are benefits that people derive from nature. Some benefits are tangible, such as crop production, fish or fresh water, while others such as pollination, erosion control, climate regulation, and aesthetic and spiritual fulfilment are less tangible (MA, 2005b). Damage inflicted on the environment by man is reducing the ability of ecosystems to provide such tangible and intangible services for human populations (Chavas, 2009; MA, 2005b). Man's impact on the environment is widely acknowledged and evidence exists to suggest that this may lead to a decrease in well-being in some areas. However, whether or not deterioration of the global biosphere might reduce collective well-being on a global scale and how this might occur is poorly understood (Raudsepp-Hearne et al., 2010). Yet, as suggested by, e.g., Rockström et al. (2009) and Norgaard (2010), future well-being may be at risk if the environment continues to degrade.

One of the (many) institutional causes of ecosystem and ES degradation is that most (economic) decision makers have so far largely ignored the non-market benefits provided by nature (Farley, 2008). Markets normally favour the conversion of ecosystem structures to economic production, rather than conservation of ecosystems for the provision of ES, even when the non-monetary benefits of conservation are greater than the monetary benefits of conversion (Farley, 2008). Moreover, even when the conversion of ecological assets may be desirable, various forms of market failure (e.g., non-tangible benefits, public good nature of many ES, imperfect property rights) lead to more rapid depletion of natural capital than is socially optimal (Tietenberg, 2006). The concept of ES and an understanding of their presumed importance for human well-being (or inversely the potential impact of ES decline on well-being) has incited agents to design economic instruments for nature conservation that take ES into account.

Two instruments that have increasingly been implemented in the last two decades for the promotion of ES conservation are eco-labelling (or eco-certification) of products and payments for environmental services (PES) schemes (Le Coq et al., 2011). Both PES and eco-labelling have attracted interest as mechanisms to translate the external, non-market value of the environment into real financial incentives for local actors to provide such services (Engel et al., 2008). According to Wunder (2006), both PES and eco-labelling share two common characteristics: positive economic incentives and direct conservation orientation.

Muradian et al. (2010, p. 1205) (supported by Vatn, 2010; and Farley and Costanza, 2010 among others) define PES schemes as ‘‘a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources’’. PES programmes to secure ES provision, such as climate regulation through enhanced carbon dioxide fixation by forests, water quality provision through the maintenance of forest cover in critical watersheds, or species and genetic pool conservation through the protection of standing forests in key biodiversity hotspots, have been established in countries such as Mexico, Nicaragua, Costa

Rica, South Africa, and Indonesia (Corbera et al., 2009; Miranda et al., 2007; Muñoz-Piña et al., 2008; Pagiola et al., 2005b; Pirard, 2012b; Rojas and Aylward, 2003; Turpie et al., 2008).

The Global Ecolabelling Network (2012) defines an eco-label as ‘‘a label which identifies overall environmental preference of a product or service within a specific product/service category, based on life cycle considerations’’. An eco-label serves to differentiate the product from other products and to assure consumers that a product is produced in accordance with specific standards (Boström and Klintman, 2006; Jordan et al., 2005). Eco-labelling is a way to increase financial incentives for the provision of public goods. Hence, a labelled product is a private good sold on the market, but bundled with a jointly produced public good (Ferraro et al., 2005). The best-known example of such labelling is organic certification.

Not all authors consider PES and labelling to be different instruments. According to some classifications specific PES tools include direct public payments, direct private payments, tax incentives, cap and trade markets, voluntary markets, as well as labelling programmes (FAO, 2012; Scherr et al., 2006). In this chapter, it is assumed that the difference is based on a payment per unit of land for specific management activities for PES, while eco-labelling is based on a payment per unit of product produced under specific practices. Eco-labels are subject to commodity chain governance, whereas PES mechanisms are based on territorial governance (Le Coq et al., 2011).

One particular question is whether PES and eco-labelling also address equity issues in terms of the distribution of conservation costs and benefits. The conversion of natural resources benefits those who convert the resource, whereas the costs are shared by the rest of the world (Farley, 2008). The benefits of conservation are often widely dispersed and enjoyed to a large extent by non-local (often wealthier) users, while the costs of conservation are mainly borne by local authorities and local communities (Balmford and Whitten, 2003; GEF, 2006; Hein et al., 2006; Kremen et al., 2000). The main sources of local costs are those incurred due to restricted access to the resources in question, local opportunity costs of conservation, and costs due to wildlife damage (Gadd, 2005; Ghimire et al., 1997; Naughton-Treves et al., 2003; Newmark et al., 1993; Ninan et al., 2007; Schaafsma et al., 2011; Vedeld et al., 2007).

Payments for environmental services (PES) and eco-labelling have the potential to redistribute the costs by providing income for nature conservation. Nonetheless, Pagiola et al. (2005b) emphasize that PES is not intended to be a mechanism for poverty reduction, although it is expected to have some impact on this issue. Because low-income households and communities throughout the developing world live in many of the most biodiverse and threatened lands, they potentially stand to gain from PES and eco-labelling (Milder et al., 2010; Molnar et al., 2004).

The aim of this chapter is to explore the cost-effectiveness of PES and eco-labelling in delivering ES provision and in contributing to rural income creation. The chapter provides empirical evidence on the theoretical framework developed by Le Coq et al. (2011), who give an overview of the differences and similarities between PES and eco-labelling. Ferraro and Simpson (2002) demonstrated that direct payments for ecosystem protection can be more cost-effective than promoting commercial activities that generate ecosystem protection indirectly. By comparing conservation contracts with organic certification and with land sales, this chapter also adds to the work of Clements et al. (2010) who compared community-based ecotourism, wildlife friendly products, and direct contracts for bird nest protection in Cambodia. The ES considered here are hydrological services, carbon stocking and sequestration, and biodiversity. In addition, the suitability of the different land uses in terms of connectivity is analysed. Rural income creation is assessed by the impact of PES and eco-labelling on household income, inclusion of smallholders, and job creation.

The next section outlines our methodology. Section 4.3 describes the results of the different indicators for ES and rural livelihoods. Section 4.4 measures the costs of the mechanisms. Section 4.5 studies the cost-effectiveness, the distribution of the costs and the origin of the revenues for the different mechanisms, and Section 4.6 provides some conclusions.

4.2. Methodology

4.2.1. Research area

This research took place in the area surrounding Podocarpus National Park in the province of Loja in Ecuador (Figures. 4.1 and 4.2). This national park lies in the southern part of the Ecuadorian Andes and extends over 146,280 hectares of mainly mountain forests and several thousand hectares of Andean tundra ecosystem, *páramo* (Keating, 2000). Podocarpus National Park is part of the El Condor-Podocarpus Biosphere Reserve, which has one of the richest ecosystems on earth (Josse, 2001; Mutke and Barthlott, 2005), and hence, its conservation should be a priority (Myers et al., 2000).

The rationale behind focusing on the buffer zone of a national park is that protected areas constantly interact with their surroundings. Hence, within protected areas, the destiny of biodiversity is intricately linked to the wider landscape (Wallace et al., 2003). Landscapes located near to protected areas or within key biological corridors represent the areas where integration is most likely to be successful, because in these locations there is an overlap between conservation and rural development priorities (Harvey et al., 2008). According to TEEB (2010b), people in buffer zones of protected areas should have secure incomes from environmentally friendly resource use to support conservation.

4.2.2. PES and eco-labelling schemes

Three different mechanisms for financing the provision of ES were identified within the study area. The implementation and management of the mechanisms are undertaken on a regional/national level. The mechanisms are all managed from offices in the city of Loja, but are implemented in different municipalities and provinces within the Southern Region of Ecuador, which comprises the provinces of El Oro, Loja, and Zamora Chinchipe (Figure 4.1).

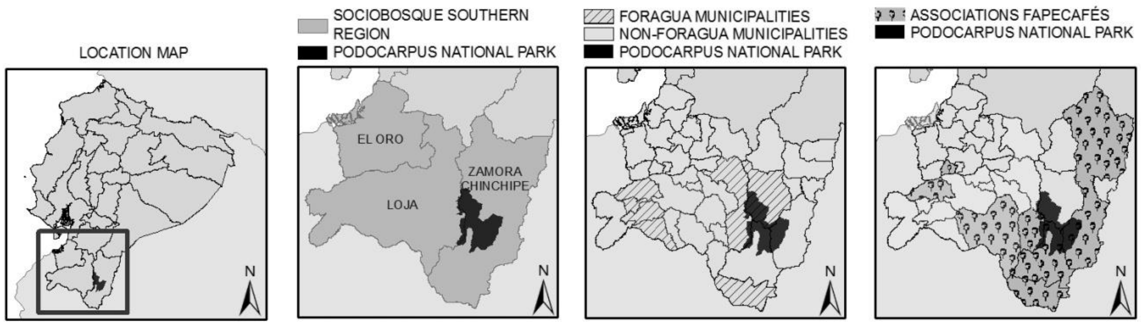


Figure 4.1: Implementation of the PES and eco-labelling schemes in the Southern Region of Ecuador (Source: Cevallos and Raes)

Three projects are studied: one is an eco-labelling programme for coffee developed by the Agro-artisanal Association of Ecological Producers of Specialty Coffee from Loja (APECAEL). The PES schemes considered are Socio Bosque and FORAGUA. These comprise, respectively, a land conservation programme and a regional water fund that invests in land conservation.

The members of APECAEL produce coffee in accordance with the principles of organic agriculture and they practice agroforestry. APECAEL is a member of the regional coffee-growers umbrella organisation FAPECAFES. FAPECAFES incorporates seven coffee-growers' associations, with a total of 2,000 members. APECAEL has 120 members, 43 of which have farms located within the buffer zone. So far only 37 farmers in the buffer zone have commercialized their coffee through FAPECAFES (Table 4.1). FAPECAFES has its own standards for organic production, but some of its members already have organic certification. APECAEL is currently in the process of obtaining international organic certification. The coffee farms of APECAEL members produce polyculture-shade-grown organic coffee.

The Socio Bosque Programme is a state-funded, nationwide programme that at the time of this research provided participants with a financial reward for every hectare of land they conserve under the programme. Individuals receive US$ 30/ha annually for the first 20 ha enrolled in the programme. This amount reduces gradually to US$ 0.50/ha for land included in excess of 10,000 ha. For communities, it starts at US$ 35/ha for the first 100 ha and gradually reduces to US$ 0.70/ha for land exceeding 10,000 ha (the payment amount has since changed, see Chapter 5). Contracts last for a period of 20 years. If conservation is suspended before the expiry of this period, then the money received has to be repaid (for more information see de Koning et al., 2011, and Annex 1). In the Southern Region of Ecuador, 30,450 ha is conserved through 295 individual and six communal agreements. The participants jointly receive US$ 607,411 per annum. Within the Municipality of Loja, 2,450 ha is conserved by 17 individuals (Table 4.1). Among them, eight participants have their property within the study area; this constitutes seven individuals and one property that is part of a nature reserve (Figure 4.2). Socio Bosque describes itself as a system of compensation for ES as opposed to a PES scheme. Their reasoning is that payments are not based on the opportunity costs of the land, but on an equal payment per hectare for all participants, and therefore, they should not be considered a PES scheme (Ministry of the Environment, Ecuador, 2010). However, as ES provision is used to prioritize areas for programme implementation and as this is linked to a financial transfer, it still falls within the broader range of PES schemes.

FORAGUA, the Regional Water Fund established in 2009, uses the revenue from an extra charge on the water tariff to purchase land in the watersheds that supply water for the municipalities in the Southern Region of Ecuador. The land purchased is then transferred to the municipality, and it becomes part of the municipal reserves. Currently, seven municipalities participate in the fund, with annual revenues of around US$ 500,000, covering over 33,000 ha of municipal reserves. In the municipality of Loja, US$ 350,000 is collected annually. EMAAL-EP, the municipal water company, ultimately decides how the money is to be spent, with approval by FORAGUA and the Municipal council. EMAAL-EP is also responsible for management of the protected areas, after the land purchase (Annex 2). Currently, 4,856 ha of land (52 % of the watersheds) is protected within the municipality of Loja. Of this area, 1,645 ha lies within the study area (Figure 4.2, Table 4.1). The concept of ES is used here as the rationale behind an increase in the water tariff for consumers of potable water. This is a payment by ES users to secure and improve the provision of hydrological ecosystem services, and could thus be considered a PES-like scheme (see also Chapter 3).

The organic coffee project is implemented by a NGO, the Socio Bosque Programme by the national government, and FORAGUA by a NGO in conjunction with the municipal authorities.

Table 4.1: Participants and area for the cases in the Municipality of Loja and area within the buffer zone

	APECAEL – Coffee label	Socio Bosque	FORAGUA (EMAAL-EP)
N° of participants in the municipal area of Loja	74 (120[a])	17	1
N° of participants in research area	37 (43[a])	8	1
Hectares in the municipal area of Loja	93	2,450	4,856
Hectares in the buffer zone	51	991[b] (1,641[b])	831 (1,565[c])

[a] According to FAPECAFES, APECAEL has 120 members, 43 of which are within the buffer zone. In 2010 and 2011 only 74 of the coffee producers of APECAEL, 37 in the buffer zone, sold coffee through FAPECAFES.

[b] Given that there is a continuous interaction between a national park and its surroundings, i.e. a national park is not an area separated from the land it borders (DeFries et al., 2010; Hansen et al., 2011), we do include land incorporated in Socio Bosque that lies within the National Park that was privately owned before the park's establishment. However, the area within the Park is not additional in terms of ES provision (see Table 4.3), but as a compensation mechanism for the earlier establishment of the park will have an economic impact on the (previous) owners of this land (see Table 4.5).

[b] Total Socio Bosque area is 1,641 ha, but 650 ha are part of a privately owned reserve "El Bosque". It forms part of the national system of protected areas and so differs from the rest of the land incorporated into the programme. Furthermore, its additionality would be zero, based on the criteria we used here.

[c] The watershed of Pizarro (734 ha) is also a protected watershed, but since 1993 and so not through FORAGUA. In this chapter only the watershed of El Carmen is considered (total area: 912 ha, with 88 ha still to be purchased).

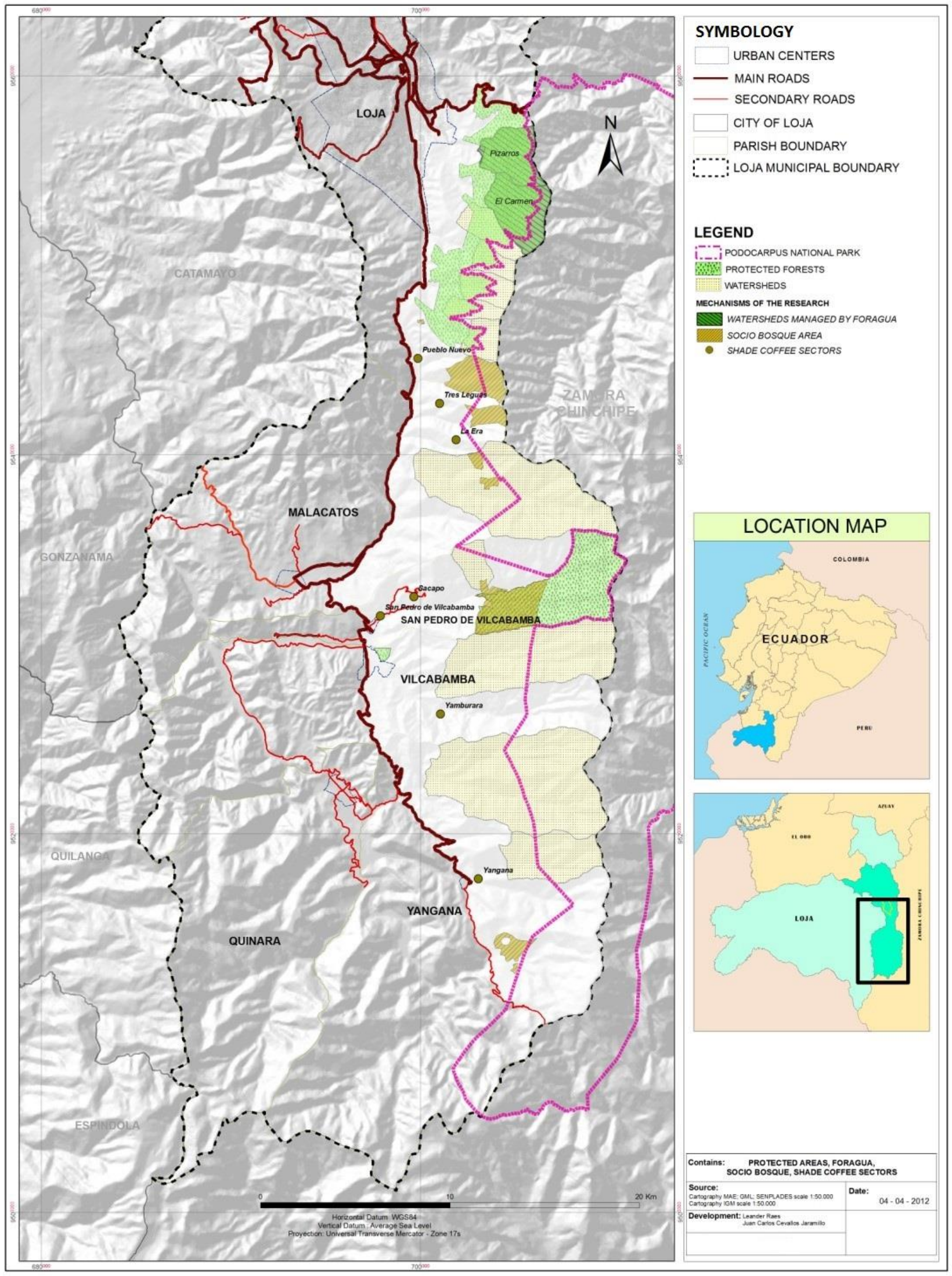


Figure 4.2: Study area and location of the PES and eco-labelling schemes (Source: Cevallos and Raes with data from Ministry of the Environment of Ecuador, 2013; GLM, 2013; SENPLADES, 2013; IGM , 2013)

4.2.3. Research framework

Two indicators for ES provision are used. The first indicator is the ecosystem service index (ESI) developed by the World Bank (Pagiola et al., 2007). The ESI was calculated based on the opinion of experts who ranked the various ES provided by different land uses. However, for the ESI indicator, only additional ES directly related to the programme are considered. Because the ESI does not distinguish between the previous and the new conservation status of the land and does not indicate whether regeneration is taking place, a second indicator was developed, namely the increased conservation and ecosystem regeneration indicator (ICERI). This indicator comprises two parts. First, the increased conservation indicator captures how much protected area is created through the different mechanisms. Second, the ecosystem regeneration indicator accords one point to land that is in the process of regeneration and zero to an already existing mature habitat.

Rural income creation is measured by the impacts of the programmes on both household income and the extra jobs created. These indicators were quantified using field data from survey results.

The cost-effectiveness of the three mechanisms is calculated by taking into account the different indicators, the cost estimates, as well as the distribution of the costs between the different actors. Finally, the different sources of funding for the programmes are analysed. The research framework is illustrated in Figure 4.3.

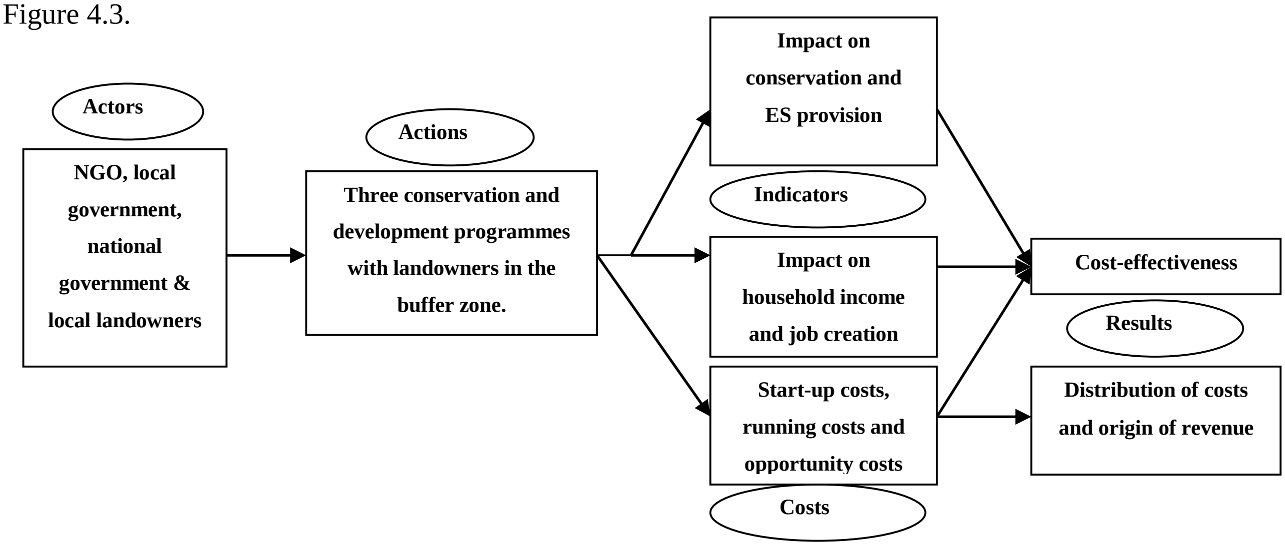


Figure 4.3: Research framework

4.2.4. Data collection

Although in most cases it is possible to list which services an ecosystem produces, it is difficult to quantify the level of service provision (Brauman et al., 2007). Few studies exist that quantify the ES provision by the different ecosystems within the study area. As indicated above, in this chapter, we followed the World Bank's ESI methodology used for quantifying the ES provided by a silvopastoral project in Nicaragua (Pagiola et al., 2007). Experts were contacted and requested to assign points to the ES provided by different land uses. Under the ESI methodology developed to assess the Nicaragua project, indices for biodiversity conservation and carbon sequestration are aggregated. The biodiversity conservation index is normalized and results in a scale from the most biodiversity-poor land uses (zero) to the most biodiversity-rich land uses (one). Similarly, the carbon sequestration index assigns points to different land uses according to their capacity to sequester carbon. In this study, the ES taken into account are hydrological services (mainly watershed services), erosion control, carbon stock, carbon sequestration, biodiversity, and connectivity. A panel of 15 experts[19] was asked to assign points to each ES provided by the different land uses, on a scale from zero to one. The points for each ES provided were summed by specific land use. Only additional ES were taken into account. Of the total ESI for each programme, the ESI for land that was already conserved, but that is now incorporated into the programme, is subtracted from the total ESI, as well as the ESI for the alternative land uses for the newly incorporated land not previously protected (Table 4.3).

To gain additional insights into the improved provision of ES and the increase in the conservation area, maps with the different ecosystems found in the FORAGUA watershed and surveys on the state of the forest conserved under Socio Bosque were used. These data were used to calculate the increased conservation and ecosystem regeneration indicator (ICERI).

Landowners were surveyed from August 2011 until January 2012 in five parishes of the municipality of Loja. Respondents were selected on the basis of owning land inside the buffer zone and being involved in one of the three programmes investigated. In total, 43 landowners were interviewed, including 37 coffee farmers, and seven participants of Socio Bosque[20]. In addition, data from 27 non-organic coffee farmers were used to analyse differences between organic and non-organic coffee farmers.

As one of the aims of this chapter is to measure the impact that the mechanisms have on rural households, measuring household income and land productivity was crucial. Household income was computed by adding the income the respondents said they received from different sources (net income from agricultural, forest, and dairy production; livestock sales; and off-farm income). Farm production consumed by the household was valued using farm-gate prices. Using these data, the household income and participation indicator (HIPI) was developed. Data on additional labour input costs for coffee and additional hired labour for the other programmes were used to obtain the extra rural work indicator (ERWI).

In-depth interviews were undertaken with five key individuals from FORAGUA (including people from the NGO Nature and Culture International), three informants from FAPECAFES, three informants from Socio Bosque, and three informants from Colinas Verdes (a local NGO working with APECAEL). The aim of these interviews was to gain additional insight into the mechanism design and to obtain data on start-up, management, and transaction costs.

[19] A contact list of people working on ES in Ecuador was developed. It comprised 25 researchers from Ecuadorian universities and project managers working for NGOs and the government. Of the 25 researchers contacted, 10 completed the ESI individually and five did it together.

[20] One coffee producer also participates in the Socio Bosque Programme.

4.3. Indicators of ecosystem service provision and rural income generation

4.3.1. Ecosystem services provided

As indicated above, the ES considered in the study are hydrological services, erosion control, carbon services, biodiversity, and connectivity. Hydrological services include improvements to water supply, mitigation of water damage, provision of water-related cultural services, and water-associated supporting services (Brauman et al., 2007). By protecting soils from wind and water erosion, forest ecosystems provide a soil erosion control service—one of the fundamental ES that could potentially contribute to well-being (Fu et al., 2011). Carbon stock and carbon sequestration are the two categories of ES provided by forests that can help to reduce atmospheric carbon (Montagnini and Nair, 2004). FAO (2004) defines a carbon stock as the quantity of carbon contained in a reservoir or system. Carbon sequestration refers to the process of removing gaseous carbon from the atmosphere and fixing it in soil or woody material on land. While carbon stocks are regarded as having the greatest potential for rapid mitigation of climate change, carbon sequestration takes place over a much longer period of time (Montagnini and Nair, 2004). Biodiversity provides both direct (provisioning, regulating, and cultural) and indirect (supporting) ES (Loreau and Hector, 2001; MA, 2005b; Naeem et al., 1994). The Podocarpus cloud forest has been identified as one of the most species-rich forest ecosystems in the world. The forests in this area are believed to contain the highest number of plant species of all the world's tropical forests (CI, 1997). Finally, connectivity is not an ES, but habitat fragmentation is widely recognized as a major threat to biodiversity (Bennett et al., 2006; Gardner et al., 2009; Yerena et al., 2003). This implies that in a homogenous landscape, land uses that tend to improve connectivity are of importance in a buffer zone (Philpott et al., 2008).

The literature concludes that mature forests will provide the highest values for hydrological services, biodiversity conservation, connectivity, and carbon stocks (Beck and Richter, 2008; Bruijnzeel, 2004; Castelle and Johnson, 2000; Fehse et al., 2002; Harvey et al., 2006; Kessler and Kluge, 2008; Pan et al., 2011; Restrepo and Alvarez, 2006; Roa-García et al., 2011). It is generally such mature forests that are conserved through Socio Bosque and through some of the land purchased by FORAGUA (Table 4.3).

Forest regeneration can help to re-establish the provision of some ES and will provide the highest carbon sequestration service (Bathurst et al., 2010; Chandler, 2006; Günter et al., 2009; Hernández-Hernández et al., 2008; Olschewski and Benítez, 2005; Preece et al., 2012; Rhoades et al., 2000). Recently established forests are found on the land purchased by FORAGUA and some of the forests conserved under Socio Bosque (Table 4.3).

Agroforestry systems, such as shade-grown coffee, can also provide important ES that resemble those provided by forests, including biodiversity, carbon sequestration, prevention of soil erosion and aquifer recharge (Blackman et al., 2007; Gómez-Delgado et al., 2011; Harvey et al., 2005; Harvey and Villalobos, 2007; Heidkamp et al., 2008; Philpott et al., 2008; Philpott and Bichier, 2012; Ponette-González et al., 2010; Siles et al., 2010; Soto-Pinto et al., 2010). Nevertheless, these systems typically provide lower levels of ES compared to native forests (Ávalos-Sartorio and Blackman, 2010; Rappole et al., 2003).

Ecosystem service index

The average value of the rankings provided by the experts for ES provision by the different land uses is shown in Table 4.2 (Supplementary Table 5.1 provides an overview of the standard deviation of the rankings). The ESI estimation shows that experts believe that primary forests and páramo provide more ES compared to the other land uses.

Table 4.2: Ecosystem Service Indicator (ESI)

Ecosystem	Hydrological Index	Erosion Control Index	Carbon Stock Index	Carbon Sequestration Index	Biodiversity Index	Connectivity Index	Ecosystem Service Indicator
Primary forest	1.0	1.0	1.0	0.2	1.0	1.0	**5.2**
Páramo	1.0	1.0	1.0	0.2	1.0	1.0	**5.2**
Mature secondary forest	0.8	0.8	0.9	0.5	0.8	0.8	**4.6**
Young secondary forest	0.7	0.6	0.6	1.0	0.7	0.7	**4.3**
Forestry plantation	0.5	0.5	0.4	0.8	0.5	0.6	**3.3**
Organic coffee system	0.6	0.6	0.5	0.5	0.5	0.5	**3.2**
Abandoned pasture	0.4	0.5	0.3	0.4	0.3	0.3	**2.2**
Unmanaged pine/eucalyptus plantation	0.4	0.4	0.5	0.8	0.3	0.3	**2.7**
Conventional coffee	0.4	0.3	0.2	0.3	0.3	0.2	**1.7**
Pasture[a]	0.1	0.3	0.1	0.0	0.1	0.0	**0.6**
Annual crops[a]	0.0	0.0	0.0	0.1	0.0	0.0	**0.1**

[a] Added for comparison

To gain an overview of the quantity of ES provided by the different land uses in the buffer zone, the ESI should be multiplied by the number of hectares covered by each mechanism in Table 4.3. However, this figure does not provide insights into the additionality of the programme. For this, the ES provided by previously protected land has to be deducted from the ES provided by the total land area for each programme. This mainly constitutes land that lies within the Podocarpus National Park, as is the case with Socio Bosque, and land that was already owned by the municipality before the implementation of FORAGUA (see Figure 4.2).

Socio Bosque works in part as a compensation mechanism for landowners who bought land in what is now the Podocarpus National Park prior to its establishment. Of the 991 ha analysed, 498 ha lies within the boundaries of the national park and so the ES provided by this land will be deducted from the total ESI. In the case of FORAGUA, 383 ha was already in the possession of the municipality.

The ES from the alternative land use also have to be deducted to obtain the indicator for additional ES. The alternative land use was obtained by asking Socio Bosque participants for what purpose they originally bought their land. In the case of FORAGUA, this is based on their previous use, while for organic coffee, it is conventional coffee production. For Socio Bosque, 85 ha is part of a private nature reserve, although this is not officially recognized as such, while another 100 ha is used for beekeeping. These 185 ha would remain forest under the alternative land-use scenario without the programme. From the remaining 307 ha of Socio Bosque land area, 93 ha would originally have been used for sustainable forestry. The remainder of the area could have been converted to pasture (Table 4.3). FORAGUA accounts for 448 ha of land which have been newly incorporated into the programme. This land was previously included within dairy farms. For coffee, the ESI looks at the additional ES provided by organic coffee versus conventional coffee.

Table 4.3: ESI for the different land uses

	Coffee label	Socio Bosque	FORAGUA	ESI/ha
Primary forest (ha)	0	763	296	5.2
Páramo (ha)	0	0	78	5.2
Mature secondary forest (ha)	0	182	26	4.6
Young secondary forest (ha)	0	20	302	4.3
Unmanaged pine/eucalyptus plantation (ha)	0	25	0	2.7
Organic coffee system (ha)	51	0	0	3.2
Abandoned pasture (ha)	0	0	129	2.2
*ESI*ha*	*163*	*4,958*	*3,647*	
ES of land conserved prior to the programme				
Primary forest (ha)	0	498	111	5.2
Páramo (ha)	0	0	63	5.2
Young secondary forest (ha)	0	0	206	4.3
Abandoned pasture (ha)	0	0	3	2.2
*ESI*ha conserved earlier*	*0*	*2,590*	*1,797*	
*ESI*ha additional land*	*163*	*2,369*	*1,850*	
Non-additional ES of the alternative land-uses				
Primary forest (ha)		185		5.2
Conventional coffee (ha)	51			1.7
Pasture (ha)		214	448	0.6
Forestry plantation (ha)		93		3.3
*ESI*ha alternative land-uses*	*87*	*1,397*	*269*	
Additional ES of the land area				
Additional Total ESI	*76*	*971*	*1,581*	
Additional ESI/ha	1.5	1.0	1.9	

As the programme purchases land from dairy farmers to incorporate it within a municipal protected area, FORAGUA has the highest additional ESI in total and per hectare. The eco-label provides the lowest total ESI, due to a smaller land area. The lowest additional ESI per hectare is for Socio Bosque, as the programme incorporates a large area of forest that was already conserved within the national park, as well as forest that would not have been converted under the alternative scenario without the programme. As such, the ESI shows current ES provision. It does not take into account increased provision of ES in the future, nor the change in the legal status of the land. For this, a second indicator is used.

Increased conservation and ecosystem regeneration indicator (ICERI)

The ICERI comprises two parts, namely protected/conserved area increase and area under regeneration.

The increased conservation indicator has a score of one for each hectare that is declared a new reserve, as in the case of the El Carmen watershed where land is purchased by FORAGUA. The Socio Bosque compensation for previously established protected area does not increase the total area under conservation. The other land under Socio Bosque is currently only protected for the duration of the contract. A value of 0.5 is assigned to each of these hectares conserved under the programme. The area under organic coffee does not add any land to the area under conservation and so receives a score of zero, just as the land that lies within the national park and land that was already owned by the municipality.

The regeneration indicator gives a value of one to each hectare that is being regenerated and zero to already mature ecosystems. One of the differences between Socio Bosque and FORAGUA is that the former focuses on existing forest, while the latter scheme purchases degraded land for regeneration and reforestation. Hence, FORAGUA has a higher proportion of young forest and land in regeneration within the municipal reserve. In the new organic coffee production systems, no pesticides are used and trees are planted. Yet because it is a productive system, full regeneration is not possible. Based on the calculated biodiversity index, organic coffee has been given a value of 0.5/ha for the regeneration indicator (Table 4.4).

The ICERI is the sum of the ecosystem regeneration indicator and the increased conservation indicator.

Table 4.4: Increased Conservation and Ecosystem Regeneration Indicator (ICERI)

	Total Area (ha)	Increased Conservation Indicator	Mature Habitat (ha)	Land in Regeneration (ha)	Organic coffee (ha)	Ecosystem Regeneration Indicator	ICERI	ICERI/ha
Coffee label	51	0	0	0	51	26	26	0.50
Socio Bosque	991	246	946	45	0	45	291	0.29
FORAGUA	831	448	400	431	0	431	879	1.06

FORAGUA has the highest score for the ICERI, both in total and per hectare. The low score for Socio Bosque actually reflects the dual aim of the programme, namely to conserve existing forest as well as to compensate landowners who lost access to part of their land when the national park was created. The fact that the main aim of the organic coffee system is production, albeit with a conservation aspect, is reflected in its lower score.

4.3.2. Socio-economic impact of the different mechanisms

Household income and participation indicator (HIPI)

The different mechanisms aim not only to finance nature conservation, but also to improve rural livelihoods. This is particularly so for the coffee project, which is aimed specifically at the improvement of smallholders' livelihoods. Although to a lesser extent, Socio Bosque also seeks to contribute to rural development as it incorporates a poverty alleviation component. The direct effects of the two projects are the additional income received by the participants through the mechanisms employed and the extent to which smallholders can participate.

Coffee label. The mean land area for coffee producers in the buffer zone is 12 ha: cropland (2 ha), pasture (5 ha), and forest (5 ha). Households receive a mean income of US$ 609/year for coffee sold through FAPECAFES and a mean income of US$ 147/year for coffee sold to intermediaries. Apart from coffee production, farmers will diversify their agricultural production into other crops, either for consumption or for sale. Farmers produce a wide variety of crops ranging from lime and corn, mainly for subsistence, to passion fruit which they sell. Animal production includes chickens, eggs, and guinea pigs for subsistence as well as cattle and dairy products for sale. Pigs are kept both for subsistence and for sale. Mean non-coffee household farm income is US$ 1,978/year. An important proportion of the coffee producer households' income is generated off-farm[21], with a mean income of US$ 3,262/year[22].

To calculate the additional income generated through eco-labelling, the profit farmers would make if they sold all their coffee as non-certified to an intermediary is subtracted from the profit they actually make[23]. The total additional profit generated by the 51 ha of certified coffee for the 37 households included in the programme is estimated at US$ 7,408 or US$ 146/ha per year.

Socio Bosque. Participants of Socio Bosque tend to have more diverse, and highly variable, income patterns compared to the coffee producers. Their household incomes range from a minimum of US$ 2,951 to over US$ 25,000 per year. The contribution of the programme payments to the household income varies from less than 10 percent to more than 70 percent for one of the participants. The total profit generated through participation in Socio Bosque for the seven households in the study area is US$ 13,170 or US$ 13/ha per year.

FORAGUA. The eight households that sold land to FORAGUA were already living outside the study area or moved when selling their land. Most of them currently live in Loja city. This implies that although it may have been profitable for the landowners to sell their land through FORAGUA, this income did not remain within the study area, and so the multiplier effects of the income on rural income are zero.

The overall additional profit for the participants in Socio Bosque is greater, while on a per hectare basis, the greatest profit is generated through the organic coffee programme. It should, however, be noted that the benefits from the eco-labelling depend on world market prices and will only last as long as coffee is produced. The benefits from the Socio Bosque Programme will last for the duration of the project, namely

[21] Off-farm income was calculated based on INEC (2012b)
[22] Standard deviations: income coffee association: 662; income coffee intermediary: 343; non-coffee household farm income: 2,222; households' off-farm income: 4,417.
[23] Total current profit coffee: US$ 26,605; total coffee profit selling the total harvest to intermediaries: US$ 19,197.

20 years. Moreover, using profit per hectare as an indicator does not show the number of people in receipt of this profit, or whether smallholders have the opportunity to participate. The correction factor for inclusion used here is the number of families that participate per hectare of land under the mechanism. For the coffee system, this is 0.73 families/ha, and for Socio Bosque, this is 0.007 families/ha[24]. An overview of the value of the indicator for the different mechanisms is given in Table 4.5.

Table 4.5: Household Income and Participation Indicator (HIPI)

	Profit US$/ha for participants	Participation Correction Factor	HIPI/ha
Coffee label	146	0.73	106.58
Socio Bosque	13	0.007	0.09
FORAGUA	0	0	0

The coffee production system clearly has the highest impact on rural income creation. This confirms that diverse agro-ecosystems can provide alternatives and more sustainable livelihoods for families (Blackman et al., 2003). Nonetheless, as the poor have limited endowments and poor access to key production factors, additional income from eco-labels is limited by the low level of coffee production due to restricted land access and/or production intensity (Le Coq et al., 2011; Valkila, 2009). While PES schemes are often not designed to include poverty alleviation strategies, they can contribute to more sustainable livelihoods through the provision of cash or in-kind benefits to participants (Pagiola et al., 2005b; Pagiola and Platais, 2002). However, payments received by smallholders in PES schemes (particularly in areas such as the study area) are limited due to the relatively small landholdings and the low level of payments (Le Coq et al., 2011). This can be observed in Socio Bosque, where a minimum amount of forest area is needed to make participation profitable. This explains why the difference between the coffee label and Socio Bosque is even greater when a correction is made for participation. A further difference is that to participate in Socio Bosque, one has to own the land. This is not a prerequisite in the coffee growing system, where renting land offers an alternative way to participate. Yet, given that, at the time of research, participants in the buffer zone only rent land for raising cattle and not for coffee, the property rights issue has not been taken into account in the indicator.

Extra rural work indicator (ERWI)

Indirect effects of the programmes are the additional jobs created within the study area. This includes the people employed by the participants, or by the programme itself, but inside the buffer zone. This does not include jobs created for those who work for the administration of the different programmes. Hence, a second indicator for rural income creation is the labour that is generated inside the study area. For organic coffee, this is the increased use of wage labour for weeding. To obtain the proxy for additional labour, the average labour cost per hectare for weeding on non-organic farms (US$ 89/ha) was subtracted from the individual

[24] For coffee 37 families use 51 ha of land; for Socio Bosque seven families use 991 ha of land.

cost per hectare for the organic farmers[25]. In Socio Bosque, two participants pay someone to protect the forest. In the case of FORAGUA, one person is hired to protect the watershed against trespassers and fire (Table 4.6).

The coffee label has the highest value for this indicator, as organic coffee seems to require a great deal of additional labour.

Table 4.6: Extra Rural Work Indicator (ERWI)

	Total US$ paid for work in buffer zone	US$/ha paid for work	ERWI/ha
Coffee label	10,747	297	297
Socio Bosque	3,474	3.5	3.5
FORAGUA	3,598	4	4

4.4. Costs of conservation actions

In this section, we differentiate between several types of costs that were incurred during the design and implementation of the project. On the one hand, costs are necessarily incurred to (1) design and decide on specific conservation actions and (2) implement the agreed conservation actions, such as the financial cost of land purchase and land management. On the other hand, setting land aside for conservation or restricting specific activities invariably incurs opportunity costs for forgone production (Carwardine et al., 2008; Frazee et al., 2003). Hence, based on Wunder et al. (2008), the costs considered in this study are (a) start-up costs, (b) recurrent costs (management and additional production costs; and ongoing transaction costs), and (c) opportunity costs.

The conservation costs are assessed to estimate the cost of supplying a range of ES on a per hectare basis. To quantify the cost of conserving rainforest or of implementing an agroforestry system, a cost analysis is based on a simple calculation of the current value of the costs. Costs incurred during previous years were compounded to 2011 values using the Ecuadorian annual inflation rate (IMF, 2011). Costs are based on the costs reported by several NGOs and by the different administrations. Costs for Socio Bosque and FORAGUA are often given/published as aggregates at regional or national level. To obtain cost estimates for the buffer zone for these two programmes, average costs per hectare were calculated, and these were then multiplied by the size of the area inside the buffer zone. One should note that although the procedure we followed implies that costs are probably not fully accurate, a per hectare estimate is more reliable than an estimate per participant, as costs differ according to the size of the different properties, at least for the FORAGUA and Socio Bosque Programme. The cost estimation for the coffee programme was based on the cost per participant as most start-up costs were based on a specific number of households

[25] The difference between weeding costs for both groups is significant at the 1 % level (p value is 0.000 with Independent samples Mann–Whitney test).

participating in the project, independent of the size of their land, while recurrent costs were based on the costs per bag of coffee[26].

4.4.1. Start-up costs

In this study, start-up costs are all costs that have been incurred to set up the different programmes. These costs are reported in financial statements of the conservation projects. They also include costs incurred to obtain the necessary information to establish the mechanisms, as well as the costs participants incur in fulfilling all the requirements necessary to participate in a programme.

Coffee label. Although coffee has been produced in the region for a long time, many coffee-growers abandoned their plantations after the collapse of world coffee prices in the nineties. One of the strategies to generate a more sustainable income from coffee has been the establishment of organic and/or specialty coffee brands and coffee producer associations. In Loja, this was done through a local NGO, Colinas Verdes, with financial aid from GTZ, CORPEI, CORECAF, and Manos Unidas. The first project ran from 2003 to 2005, and the second from 2009 to 2011. The aim of the projects was to establish a coffee association, install post-harvest infrastructure, renew the coffee plantations, and establish organic production systems[27]. The cost of the time spent applying for funding and the donors' monitoring costs were not taken into account. Given that the infrastructure for coffee was donated, these costs are included as start-up costs and not as annual amortization under recurrent costs.

Socio Bosque. This programme started paying incentives from 2009 onwards. The political costs for negotiating the establishment of Socio Bosque were difficult to quantify and have not been taken into account. As a proxy for start-up costs, the reported costs of the programme for the year 2008 were used, as well as annual costs for the diffusion of the incentive mechanisms on a national and regional level, and the administrative costs of incorporating land both for the secretariat and for the participants.

FORAGUA. The initial idea to establish FORAGUA dates back to 2006. It officially started operating in 2009. In this study, only cost data for setting up FORAGUA in the Municipality of Loja are included. These are the costs for establishing the secretariat, as outlined in the testimony of Attorney for the trust fund, the costs incurred by the NGO Nature and Culture International and by the Municipality of Loja in studies, consultancy, and the purchase of land. The costs incurred setting up the municipal regulations to collect the environmental charge on water, and for establishing the municipal protected watersheds, have not been

[26] One bag of coffee weighs approximately 46 kg.

[27] Coffee farmers previously produced coffee following the dry processing system, where coffee beans are harvested and dried in the sun. To sell organic coffee, farmers had to switch to wet processing. In this system, coffee berries are pulped, fermented, washed and then dried in special infrastructure. Although this process is not specific to organic coffee (see e.g. specialty coffees), this shift was a necessary condition to be able to sell organically certified coffee. The (high) start-up costs thus include both the costs of changing from dry to wet processing as the shift from conventional to organic coffee.

taken into account (Table 4.7). Further details on the start-up costs can be found in Supplementary Tables 5.2, 5.3 and 5.4.

4.4.2. Recurrent costs

Recurrent costs are divided into management and production costs and transaction costs.

Management and production costs

Carret and Loyer (2003), in their study on the cost of managing a network of protected areas, included operational costs for the head office, regional offices, site operation and daily activities, and investment costs associated with managing biodiversity, and environmental education. Production costs are the combined costs for raw materials and labour incurred in the production of goods and services. For this part, only the recurrent costs for 2011 are used, as obtained from the surveys and the financial statements of the different programmes.

Coffee label. In the coffee system, the costs incurred are administrative costs for the presidency of APECAEL (US$ 1/bag) and additional production costs incurred by the farmers. These additional costs for farmers are the costs for quality control and additional weeding. To calculate the latter, we used a similar approach to that applied to the calculation of the HIPI. Costs for herbicides were added to the average weeding costs per hectare for non-organic farmers. These costs for non-organic farmers were subtracted from the weeding costs for organic farmers in order to obtain the additional organic production costs.

Socio Bosque. The management costs for the programme are the administrative costs of the national secretariat in Quito. Costs incurred by the participants, such as monitoring their protected areas and managing their participation, are considered as transaction costs. Costs incurred by the programme to pay the participants are not taken into account in this study, as these were considered to be revenues from the point of view of participants. Even if these costs were included, Socio Bosque still has the lowest management costs and total overall costs (see also Sect. 4.4).

FORAGUA. The management costs incurred by the Municipality for participating in FORAGUA include an annual payment of 10 percent of the environmental water tariff to cover running the secretariat, the administrative costs for the unit within the municipal water company responsible for the management of the watershed, as well as costs for managing the watershed itself and for the different reforestation and other activities undertaken within the watershed (Table 4.7). More details on these costs are provided in Supplementary Tables 5.5, 5.6 and 5.7.

Recurrent transaction costs

Numerous definitions of transaction costs (TC) are available, which makes TC often a vague concept. Holloway et al. (2000, p. 280) define TCs as ‘‘the costs of searching for a partner (or group) with whom to exchange, screening potential partners to ascertain their trustworthiness, bargaining with potential partners and\or officials to reach an agreement, transferring the products, monitoring the agreement to see that its conditions are fulfilled, and enforcing the exchange agreement''. From an environmental perspective, TCs are incurred to enforce specific conservation actions and to monitor compliance with specific rules implemented to conserve biodiversity.

Coffee label. Bode (2007, p. 23) defines TCs in a study on FAPECAFES as ''all the costs, generated by organisational and communicative actions''. They include telephone and internet costs, as well as the costs of all meetings and assemblies held at different levels of the federation. In this study, TCs are those costs related to the transport costs for members to attend meetings, the opportunity cost for time spent participating in meetings of APECAEL, and local coffee groups, as well as the certification costs incurred by the farmers and by FAPECAFES. Data were obtained through the surveys.

Socio Bosque. The following TCs are considered: costs incurred to establish the deforestation baseline as well as the cost of monitoring participants' compliance with the regulations. Costs also include the costs participants incur to ensure no deforestation occurs on the land they have enrolled under the programme, as well as the costs they have incurred making a legal statement concerning their fulfilment of the programme's requirements.

FORAGUA. TCs incurred by FORAGUA and the Municipality of Loja are those associated with monitoring the watersheds and those that are incurred annually to approve the investment plan proposed by the watershed unit of the municipal water company. This plan has to be approved by the municipal council as well as by the secretariat and the board of FORAGUA (Table 4.7). Further details on the transaction costs can be found in Supplementary Tables 5.8, 5.9 and 5.10.

4.4.3. Opportunity costs

Markandya et al. (2001) define the opportunity cost of an activity that uses scarce resources as the value of the next most economically productive use to which those resources could have been put. The opportunity costs of conservation typically relate to restrictions on development, land-use change, land management practices, exploitation of resources, and the ability to control species which can damage economic interests. For instance, in the case of a protected forest, the opportunity cost of preserving the forest may be the net income per hectare per year from deforestation and conversion of the land to agricultural uses (Olsen et al., 2011).

Coffee label. No opportunity costs have been calculated for the coffee label. As opposed to conservation, the land is used for productive activities. In addition, although restrictions apply for organic labelling and agroforestry, these can also provide multiple benefits for production, such as the control of specific pests, the potential for improved productivity at high altitudes, non-coffee products and improved pollination

(Ataroff and Monasterio, 1997; Borkhataria et al., 2006; Philpott et al., 2009; Ricketts et al., 2004; Torres-Lezama et al., 2012).

Socio Bosque. Under this programme the participants do incur opportunity costs by applying to conserve part of their land under the programme. This is only the case for land that does not lie within the national park, land that is a private protected area or forest used for beekeeping. Opportunity costs are calculated for 307, 93 ha originally destined for sustainable forestry and 124 ha for pasture. Calculations of the opportunity costs are based on Knoke et al. (2009a). For pasture conversion, a rate of 1 ha/year/farmer was taken into account. One participant entered the programme at the end of 2009 and two at the end of 2011. Therefore, the total conversion since the start of the programme would have been four hectares—two hectares with revenue forgone over the following 19 years and two hectares with revenue forgone for the subsequent 20 years. For forestry, the 93 ha was taken into account with revenue forgone for 1 year. The forgone revenue for cattle and dairy production is US$ 2,920, and for sustainable forestry, this amounts to US$ 3,146 (Table 4.7). Annex 5 provides more detail on the opportunity cost calculations.

FORAGUA. For FORAGUA, the opportunity costs are zero. As landowners are paid for their land, they do not necessarily suffer an economic loss. Including the opportunity costs for the land purchased would constitute double counting, as the cost of the land has already been included within the start-up costs.

4.4.4. Total costs

Table 4.7 gives an overview of the costs per category and costs per hectare, as well as total costs.

FORAGUA clearly has the highest total costs. This is mainly due to the high cost of purchasing land. Compared to the other programmes, Socio Bosque has both the lowest start-up and recurrent costs. These low costs are a reflection of the benefit of economies of scale for a national programme that offers incentives for the private conservation of forests. This is in line with Clements et al. (2010) who, in their comparison of three PES schemes in Cambodia, found that the most direct, individual contracts had the lowest administrative costs. For the organic coffee label, costs on a per hectare basis are much higher compared to the other programmes. These high costs per hectare for coffee are due to the production orientation of the programme, because of which costs for implementing the coffee processing infrastructure, additional production costs, costs for capacity building, and costs for certifying the product are included. On a per hectare basis, the costs for Socio Bosque are very low. Even if the payments per hectare to participants (US$ 13/ha) were included as costs, the programme would still have the lowest costs both in total and in per hectare.

Table 4.7: Total costs for the different programmes

	Total Cost inside study area (US$)	Cost US$/ha
	Start-up Costs	
Coffee label	83,723	1,648
Socio Bosque	7,676	8
FORAGUA	297,880	358
	Management and Production Costs	
Coffee label	14,236	281
Socio Bosque	277	0.3
FORAGUA	53,413	64
	Recurrent Transaction Costs	
Coffee label	2,711	53
Socio Bosque	4,742	5
FORAGUA	8,049	10
	Opportunity Costs	
Socio Bosque	6,492	7
	Total Costs	
Coffee label	100,670	1,982
Socio Bosque	19,187	19
FORAGUA	359,343	432

4.5. Cost-effectiveness, cost distribution, and revenue origin

4.5.1. Cost-effectiveness of the mechanisms

As indicated in Tables 4.3 and 4.4, FORAGUA generated the largest additional ES provision as well as the highest ICERI because it establishes newly protected areas and allows the regeneration of degraded land. The organic coffee system has the greatest impact on rural income because it generates profit per hectare with the widest inclusion, while also creating additional rural labour opportunities. Using the cost estimates, it is now possible to look at the cost-effectiveness of the different mechanisms.

Cost-effectiveness to achieve ecosystem service provision and conservation

Figure 4.4 shows the cost in US$ per point increase for the two environmental indicators (ESI and ICERI) (see also Supplementary Table 5.13).

According to both indicators, Socio Bosque is relatively more cost-effective. This is mainly due to its low costs and to a lesser extent because of the programme's contribution to ES provision. This is in line with Brauman et al. (2007), who compared eco-friendly subsidies with direct conservation payments and found that direct investment in conservation may be the most cost-effective mechanism for ensuring ES provision. Costs incurred in the FORAGUA programme include a high initial cost for land purchase. If only recurrent costs had been taken into account, Socio Bosque would still be slightly more cost-effective (US$ 12/ESI and US$ 40/ICERI versus US$ 39/ESI and US$ 70/ICERI for FORAGUA).

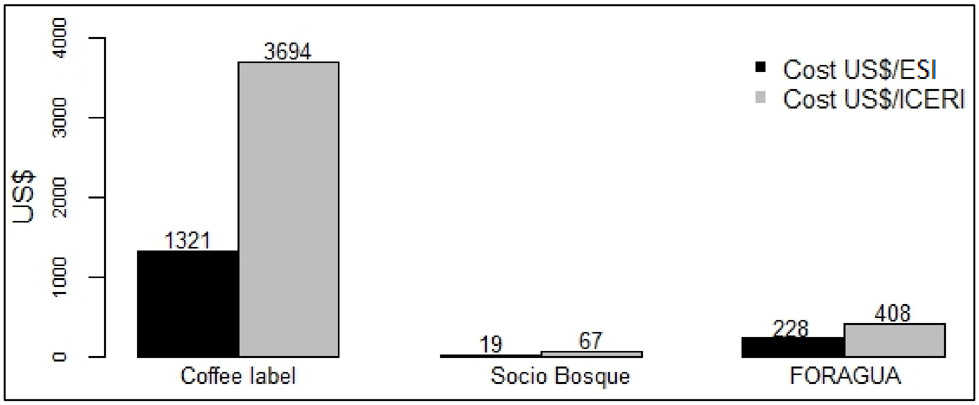


Figure 4.4: Cost-effectiveness of ecosystem service provision and increased conservation

Cost-effectiveness for rural income creation

Figure 4.5 compares the cost-effectiveness of the different programmes in achieving rural income creation (see also Supplementary Table 5.14). For the first indicator, the production of coffee is the most cost-effective when compared to FORAGUA and Socio Bosque. This is due to the high score for the HIP indicator, as it creates higher additional revenue per hectare for the largest group of rural landowners. The lower cost-effectiveness for Socio Bosque with regard to the HIP indicator is more a reflection of the limited number of participating households within the buffer zone than of the costs incurred per dollar of revenue in terms of payments to the participants (costs of US$ 1.5 per dollar income versus US$ 14 for the coffee label). Socio Bosque is the most cost-effective for the second indicator. However, this no longer holds when only the recurrent costs are taken into account (US$ 2/ERWI for coffee, US$ 3/ERWI for Socio Bosque and US$ 17/ERWI for FORAGUA). FORAGUA is the least cost-effective in contributing to rural income, which is not surprising as the programme's main aim is to increase the size of the municipal protected area.

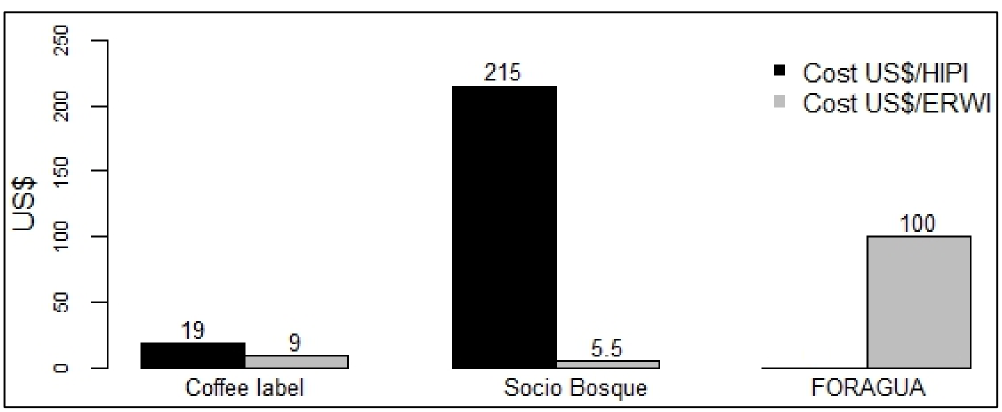


Figure 4.5: Cost-effectiveness for rural income creation

4.5.2. Distribution of the cost

Apart from the cost-effectiveness, it is interesting to investigate the distribution of the costs for the different programmes (Figure 4.6) (see also Supplementary Table 5.15).

Donors (NGOs and development cooperation) financed the start-up costs of the organic coffee system, while transaction and labelling costs are paid by FAPECAFES and the coffee producers. The producers obviously pay the additional production costs. In Socio Bosque, most of the costs are borne by the participants. These comprise the cost of applying for participation and the recurrent TCs. In FORAGUA, most costs are paid through money generated by the environmental tax on water. Most of the revenue is spent on purchasing land. Donors also funded land purchases and hydrological studies, while the local and national government pay part of the management costs.

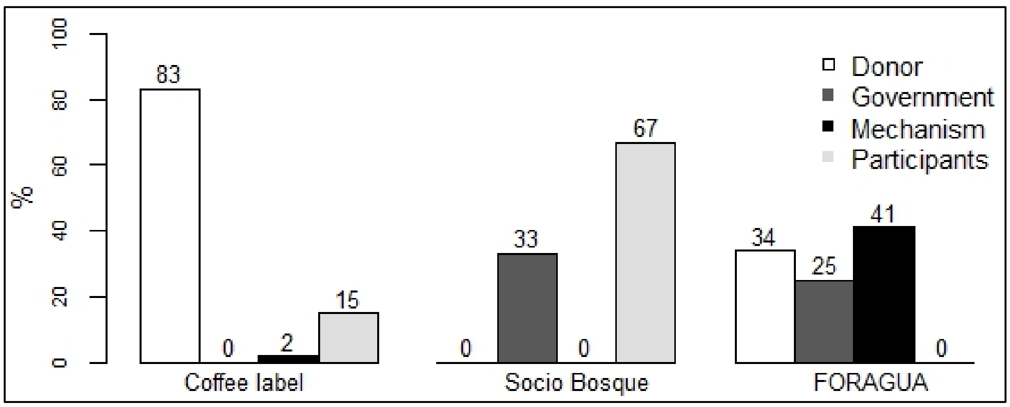


Figure 4.6: Distribution of the costs between different actors (%)

4.5.3. Origin of the payments

The final aspect considered in this study is sustainability in terms of funding. Figure 4.7 shows who directly or indirectly paid the costs (see also Supplementary Table 5.16). Donors fund most of the costs in the coffee system, yet these are start-up costs. Coffee consumers, who purchase coffee from the buffer zone, incur the other costs. Socio Bosque is fully government-funded through money that is allocated on an annual basis and that is not specifically earmarked for this purpose. Although the participants incur costs, these are completely covered by the payment they receive from the government. From 2012 onwards, the German Development Bank is to provide a total of 11.5 million Euros to Socio Bosque within the framework of a REDD initiative. For FORAGUA, the environmental charge on water provides the main revenues, but government funding is still needed to cover administrative costs, while donor funding was fundamental to cover start-up costs.

Both the origin of the payments and Section 4.5.2. on the distribution of costs, illustrate the differences between considering all costs, as we did in the cost-effectiveness analysis, versus considering costs related to agents carrying the costs. In the latter case cost-effectiveness should be calculated separately for the users of ES (who can be directly involved or represented by the government as in Socio Bosque, see

Figure 4.7), the costs for providers of ES (the participants and FORAGUA's mechanism in Figure 4.6), or only the costs for the national and local government or donors funding the programmes' implementation (see Figures 4.6 and 4.7). In addition, a difference could be made between private costs, social costs and overall costs.

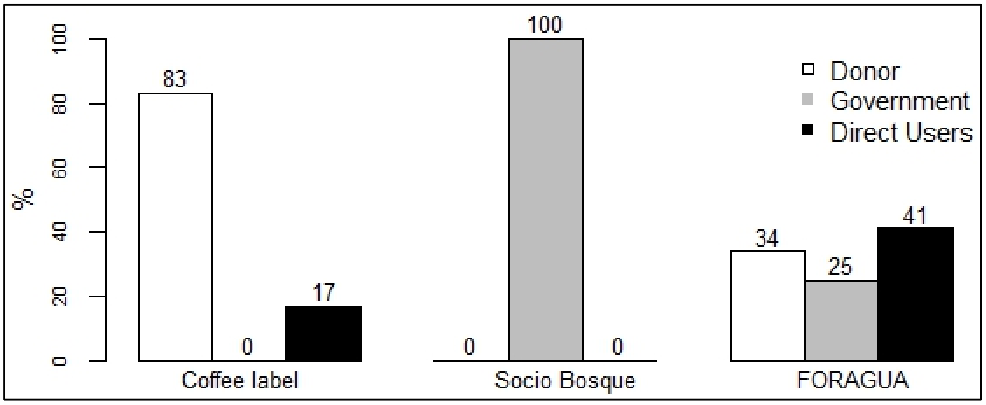


Figure 4.7: Origin of the revenue to pay programme costs (%)

As FORAGUA focuses on hydrological services with a compulsory payment part of the water bill, it has probably the most sustainable source of revenue. The environmental charge on water is set within municipal regulations, making it difficult to divert the money collected to other uses than watershed conservation programmes. Water users can easily be targeted and consumption of water is unlikely to cease. This makes continued funding very certain, both currently and over the longer term.

It is the opinion of many experts that markets for specialty coffees will continue to expand at a much higher rate than regular coffee markets and that they will continue to attract a price premium (FAO, 2009b; Lewin et al., 2004). Therefore, the coffee system has also found a relatively sustainable source of revenue. The price premium is, however, dependent on the quality of the coffee, the productivity, the coffee association, and on the world market prices. Nonetheless, the potential level of payment can still be increased as production volumes and quality are currently low.

The sustainability of funding for Socio Bosque is less straightforward. This does not mean that this programme will disappear if the national government changes. However, unlike the national PES scheme in Costa Rica, where money is set aside by law to fund the scheme, Socio Bosque currently has no other specific sources of income and funds may therefore run out or be diverted to other uses. The results are in line with Le Coq et al. (2011) who mention that an eco-label is more likely to be efficient in capturing funding from distant areas to support difficult to measure ES provision, whereas PES mechanisms may be more efficient at capturing local funding for well-identifiable and measurable ES provision, because PES schemes are based on contractual agreements that should be honoured.

If our findings here are added to the increased conservation indicator of the ICERI, it can be observed that both from a conservation perspective and from the funding side, FORAGUA creates the highest certainty by adding legally protected land to the municipal protected areas that are funded through a municipal decree. In the case of Socio Bosque, conservation is weaker as a contract can be broken (although this implies a financial penalty for the participant), while funding depends on government budget allocation. Uncertainty in ES provision seems highest for organic coffee, since it is more tied to the world market price than the actual incentive payment (the additional income from labelled coffee sales).

4.6. Final discussion and conclusions

This chapter showed that designing an optimum strategy that allows for the maximization of nature conservation and the services it provides, as well as the income of poorer households, is very difficult. A trade-off is often necessary between increasing ES provision and improving rural livelihoods. Moreover, further trade-offs are required in terms of the scope and duration of the programmes. Purchasing land will increase the provision of ecosystem services. However, taking land out of production is expensive and can have negative effects on local development. Paying private landowners for conservation increases the total rural income, but poorer households will often be unable to participate, and when they do participate, payments will be low due to small land areas and the limited payments they receive per hectare. Mainly due to their lower access to land and higher opportunity costs, smallholders will find it difficult to participate in conservation schemes, and it will cost more to include them. Nonetheless, PES and PES-like schemes are cheaper to implement than a programme that aims to increase agricultural production.

Hence, from a regional perspective, the combination of different instruments that address different areas which have different characteristics seems the best way forward. The mix of buying up land in degraded watersheds, or other key areas, paying landowners who have primary forest on their land and implementing agroforestry production systems for smallholders can create a balance between increasing conservation, ES provision, and rural livelihoods. Furthermore, targeting the right programme on specific areas and rural communities is the key to achieving such balance. Alternatively different aspects of the different programmes could be better integrated. A mixture of different instruments also permits access to different funding sources: funds from inside the region for local benefits, funds of national origin for protection of ES of national interest, and funds from international donors or international consumers for ES that have an international interest or that can be linked to private goods which are internationally marketed.

Although PES schemes for private conservation can be successful in conserving forests, they could have a greater positive impact on rural areas and livelihoods if they include payments for a greater diversity of sustainable land uses. Some PES programmes, such as the Mexican PES scheme ‘‘Programa PRONAFOR’’ and the Costa Rican national PES scheme, pay agroforestry coffee farmers on a per hectare basis for ES provided. As such PES could serve the same aim as coffee price premiums in protecting agroforestry systems.

As indicated above, a price premium for coffee is highly dependent on international consumers' willingness to pay and is strongly influenced by international coffee prices. Besides, organic coffee certification only targets ES provision indirectly and its effect on conservation will only last as long as farmers produce in an organic way. Given the low costs of per hectare payments one can argue that it is more effective to pay landowners to implement coffee agroforestry practices. This could provide them with a fixed income which is less dependent on niche markets, while foreign revenue would still enter the country from coffee exports.

Chapter 5: A portfolio analysis of incentive programmes for conservation, restoration and timber plantations

Based on: Raes, L., D'Haese, M., Aguirre, N., Knoke, T., 2016. A portfolio analysis of incentive programmes for conservation, restoration and timber plantations in Southern Ecuador. Land Use Policy 51, 244–259. doi:10.1016/j.landusepol.2015.11.019

Abstract

This paper uses portfolio analysis to study how the Ecuadorian incentive programme for forest conservation and restoration (*Socio Bosque*), and an incentive programme for timber plantations, may reduce income risk and/or maximise returns for a given level of risk for farmers in the municipality of Loja. The main existing land use in the research area is milk production on pasture, with some farmers having forest land. Our results suggest that most farmers would significantly increase the area under conservation and/or restoration through the incentive programmes as part of their risk reduction strategies, compared to a decision based solely on expected returns. However, the portfolios (land use combinations) including the incentive programmes would only lead to small areas of tree plantations being established. None of the land use combinations analysed would increase the income of all households to above the poverty line, as the monetary incentives are too low and many farms are too small. For forest holders all the land use combinations we studied would have a positive impact on income, but we observed a negative impact on household income for milk producers without forest. For producers without any forest, there seems to be a trade-off between maximising household income and risk reduction through combining incentives for restoration and tree plantations.

5. 1. Introduction

The undervaluation of ecosystem services provided by forests and other natural ecosystems is considered one of the main causes of their ongoing degradation (Pearce, 2007; Swallow et al., 2009; Tacconi, 2000). One solution that has been proposed to address to this, is to pay private and communal landholders to maintain and restore forests and other ecosystems under their stewardship (Engel et al., 2008; Rodríguez de Francisco et al., 2013). These incentives, broadly categorised as payments for ecosystem services (PES) or PES-like schemes, aim to maintain the current level of ecosystem services, or to restore or increase them (Sommerville et al., 2009; Tacconi, 2012; Wunder, 2008).

A recurrent question is how important the expected returns are in encouraging farmers to keep land under conservation management instead of converting it to other income-generating activities. Similarly, expected returns can play a role in a farmer's decision to continue with existing land-use practices, or adopt land uses that could improve ecosystem service provision. When payment levels from PES schemes are at least as high as any alternative gains from the land, one would expect the scheme to be more successful in gaining and retaining participants (Farley et al., 2011; Secchi et al., 2009). However, in PES schemes payments are often lower than the opportunity costs faced by participants (Mahanty et al., 2013; Wunder and Alban, 2008). In addition, PES participation is also influenced by intangible factors (Kosoy et al., 2007), such as the value participants place on ecosystem services (Bremer et al., 2014a) or their general concern for the environment (Zanella et al., 2014).

A broad range of factors affect farmers' land use decisions, including environmental and agricultural policies and regulations, markets, climatic conditions, the physical attributes of land and socio-economic and personal factors (e.g. beliefs, values), demographics and gender (Gasson, 1973; Lambin et al., 2001; Malawska et al., 2014; Villamor et al., 2014). Farmers' decisions about land allocation and land management are contingent upon multiple natural (e.g. variable climate) and financial uncertainties (e.g. crop or input price fluctuations) (Jakoby et al., 2014; Knoke et al., 2011). The resulting uncertainty in the expected profit from any given land use makes land use decisions risky (Engle Warnick et al., 2011).

Land owners are generally considered to be risk-averse (Knoke et al., 2008), an observation widely supported both in developed (Bocquého et al., 2014; Bond and Wonder, 1980; Just and Pope, 2002) and developing countries (Antle, 1987; Moscardi and Janvry, 1977; Tanaka and Munro, 2014). Risk-averse farmers tend to choose land uses with the least uncertainty, despite their lower potential rewards (Aimin, 2010). Farmers often diversify their activities and land uses in order to spread risk (Engle Warnick et al., 2011). Hence, in addition to differences in expected returns, comparing potential investments in different land uses requires analysing the trade-offs between the uncertainties and profitability associated with different activities. Similarly when analysing PES, the effect of the contracts on income uncertainty should be taken into account.

In addition to their primary aim of conserving or increasing ecosystem service provision, PES schemes are often implemented with an eye on poverty reduction (Ingram et al., 2014; Rodríguez de Francisco et al., 2013). Through their impact on income, consumption, labour and land markets, PES can have positive effects on livelihoods even if the programmes are not explicitly designed to reduce poverty (Kollmair and Rasul, 2010; Wunder, 2006). Yet, the success of PES programmes in reducing poverty depends on the equitable distribution of benefits and on the size of the compensation payments (Grieg-Gran et al., 2005; Jack et al., 2008). Wunder (2008) argued that PES could potentially trap poor landowners if payments are lower than actual or potential revenues from alternative income-generating land uses. However, PES can also be a stable income source and a valuable way to diversify income (Grieg-Gran et al., 2005). The balance of these effects depends on how land use restrictions impact on people's livelihoods.

Against this background, this chapter examines the extent to which PES schemes for conservation and restoration, and incentives for timber production could be desirable land uses compared to pasture for milk production. This chapter does so for farmers in an area of southern Ecuador important for hydrological services and biodiversity. The PES and PES-like programmes considered in the study are conservation and restoration incentive schemes which form part of the Ecuadorian Socio Bosque Programme, and the government's Economic Incentives for Afforestation and Reforestation programme, which encourages the establishment of timber plantations. The latter programme is not a PES *per se*, but has similar outcomes in terms of the provision of certain ecosystem services. Farmers can combine participation in these programmes with commercial milk production, which is an important agricultural activity in the study area. These land use combinations are compared to the current situation, and to combinations of potential land uses in the absence of the incentive programmes (i.e. milk production and tree plantations without incentives).

In this chapter we use a portfolio approach to calculate the shares each land use should have, subject to economic return and uncertainties, while allowing for a mixture of different land uses. Following Markowitz (1952), the selection of a land use portfolio is a minimization or maximization problem based on two criteria, namely the activity's expected return and the risk involved. After Knoke (2008), this study views specific land uses as single investments. As such, one area of land can be divided into sections with different uses, creating a combination of land uses that produce an optimum relation between revenue and risk. This chapter builds on the research of Knoke et al. (2011), who analysed an 'Optimized Land-Use Diversification' to study the effects of carbon payments as an instrument to reduce deforestation. Instead of using a portfolio approach to estimate payment levels for a PES scheme, we use it to analyse the potential impacts on income and uncertainty of individual farmers diversifying in several PES and PES-like programmes.

Following Knoke et al. (2011) we use the terms risk and uncertainty interchangeably; and agree that "the phenomenon of uncertainty was simply seen as our inability to predict something (market prices in our case) with certainty" (Knoke et al., 2011, p. 1142). In our portfolio analyses, 'risk' is (narrowly) defined by the standard deviation of the expected returns, and risk reduction is defined as the minimization of this standard deviation. Land uses for which the returns have high standard deviations are considered more risky or more uncertain. With risk avoidance, various combinations of expected returns and risk may generate an identical utility, because less uncertainty may compensate for a lower expected return and vice versa (Knoke et al., 2008). Following Castro et al. (2015), we understand that our analyses show how land should be allocated to achieve risk minimization or to maximize the expected return for a given level of risk. However, as Castro et al. (2015, p. 4) state "this does not necessarily mean that the model output is a proper prediction of future land allocation. […] It may just help risk averse land owners to achieve their economic objectives in a consistent way".

The chapter consists of the following analytical steps: (i) net present value calculations of the expected revenue of milk production, of forest conversion to pasture for milk production, of monetary incentives for conservation of forests and the restoration of native vegetation on pastures, and of the establishment of timber (pine and Andean alder) plantations with and without the incentives; (ii) portfolio analysis of the different land-use combinations; (iii) the impact on household income of adopting different land use portfolios. The methodology is detailed in the next section.

This chapter contributes to both the PES and forest economics literature. Portfolio diversification and optimization has been increasingly used to analyse PES (e.g. Benitez et al., 2006; Castro et al., 2013; Knoke et al., 2011). However, as far as we know, it has not been used to analyse the potential impact of existing PES programmes, nor to study the impact of combining several programmes. In addition, empirical evidence of the potential contribution of PES to joint social and environmental goals remains limited (Bremer et al., 2014b; Ingram et al., 2014).

5.2. Methodology

5.2.1 Research area

This research focuses on commercial milk producers who own land in the buffer zone of the Podocarpus National Park in Loja municipality in the province of Loja, Southern Ecuador (figure 5.1). The research area consists of patches of forest and pasture and covers 5,475 ha, excluding the area belonging to the National Park and two protected watersheds. In addition to its role as a buffer zone for the National Park, the area's Andean ecosystems are important providers of hydrological services (Ataroff and Rada, 2000; Celleri et al., 2007). To improve the conservation of biodiversity and maintain and enhance hydrological services (water quality and dry season flow), Loja municipality is a member of the Regional Water Fund

(FORAGUA). The municipality has the power to protect areas of hydrological importance through land purchases, using funds obtained through the water fund, and by establishing municipal reserves (Municipality of Loja, 2007). In addition, the municipal water company of Loja has reforested the El Carmen watershed, mainly with native Andean alder species.

Two watersheds in the area that provide water to the city of Loja (El Carmen and Pizarros) are now protected as municipal reserves, while the Curitroje watershed also provides drinking water to Loja city, but is currently not protected. Additional water catchments are planned, but not yet operational, in the Namanda, Monica and Santa Urco watersheds. Further measures to conserve and improve the ecosystem services provided by the lands of the buffer zone are thus needed. Iñiguez–Armijos et al. (2014) suggest that it is necessary to retain at least 70% native vegetation cover within the water catchments in order to conserve biodiversity and water quality in the Ecuadorian Andes. Other measures they propose include the exclusion of livestock and the restoration of riparian vegetation.

While the purchase of land and the declaration of municipal reserves have been the most commonly implemented measures within the research area to date, several national PES and PES-like programmes exist that could potentially be used to engage landholders within the research area in the maintenance and improvement of ecosystem services.

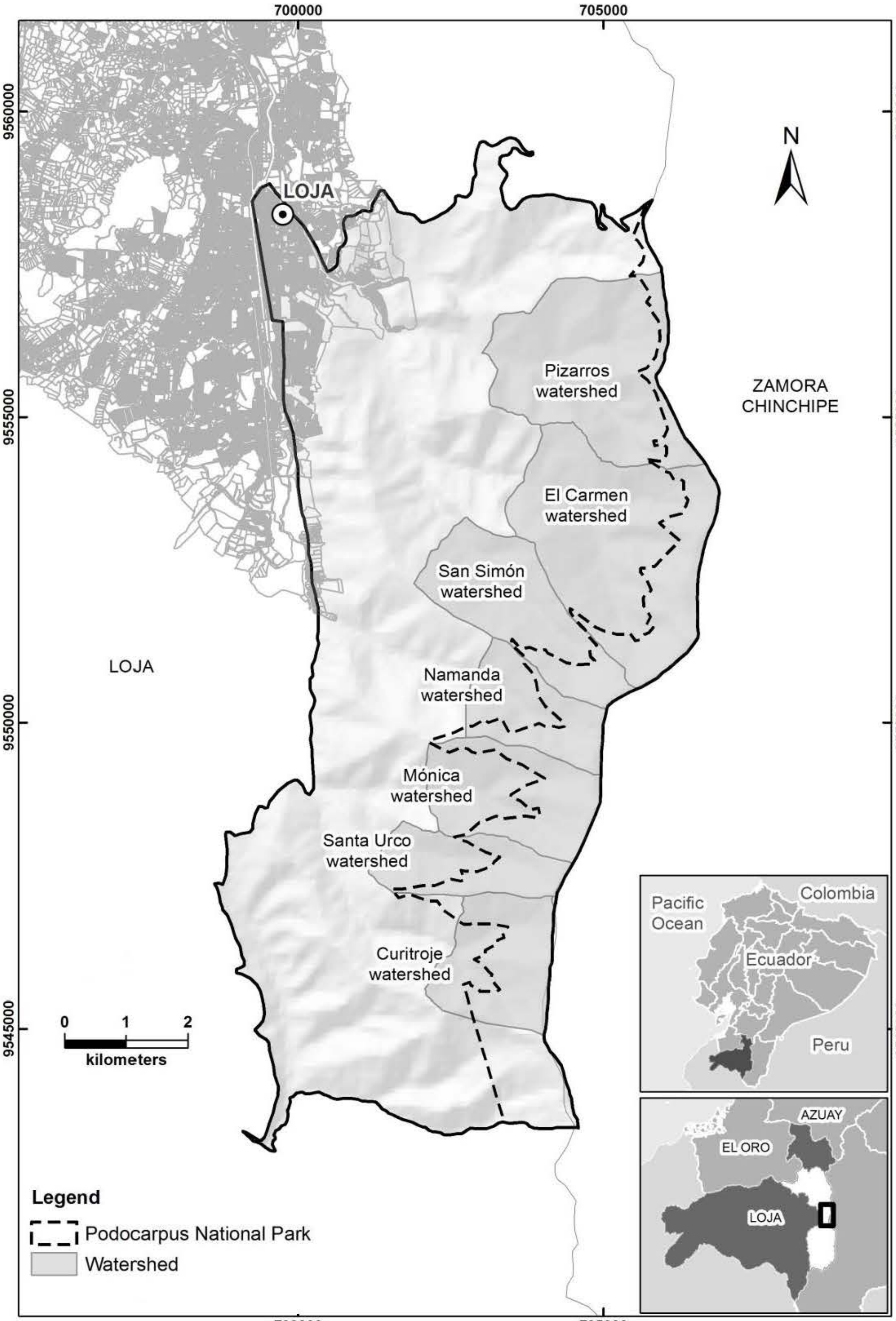


Figure 5.1: Research area (Source: Castillo and Raes)

5.2.2 The programmes

The Socio Bosque programme was established by the Ecuadorian Ministry of the Environment in 2008. It is a government-funded, nationwide programme which provides financial incentives for conservation of forest and/or *páramo* (Andean grasslands) on private and communal lands. The programme's objectives include the conservation of biodiversity, carbon storage, the maintenance of hydrological services, and poverty alleviation. Participation in Socio Bosque is voluntary, but to receive payments landowners must comply with a number of terms and make a twenty-year commitment to so doing. These terms include the prohibition of logging, burning, and commercial hunting or fishing. Furthermore, any activity that could potentially alter the hydrological conditions, reduce carbon storage, or threaten the area's capacity to harbour biodiversity is not allowed (de Koning et al. (2011) provide more details on the programme).

In 2012 the programme added a new component, aiming at restoration instead of conservation. Incentives focus on areas with young forests and abandoned pasture and on increasing the connectivity between protected areas, municipal reserves and areas under Socio Bosque. The restoration programme aims to assist natural regeneration and enrichment with native species and has been actively implemented since 2014 (Ministry of the Environment, Ecuador, 2014a) (see Annex 6 for details).

Whereas Socio Bosque aims at stimulating ecosystem restoration and halting degradation, the Ecuadorian Ministry of Agriculture provides incentives to establish plantations in formally forested areas through the programme *Incentivo Económico para la Forestación y Reforestación* (Economic Incentives for Afforestation and Reforestation). For private individuals, these incentives refund up to 75% of the costs of establishing plantations during the first four years, depending on the trees' survival rate. These incentives are for single species plantations for timber production (Ministry of Agriculture, Ecuador, 2013a, 2013b). We consider pine (*Pinus patula*) and Andean alder (*Alnus acuminata*) plantations in our portfolio calculations. These are fast-growing species (Aguirre et al., 2011). Pine is non-native and is currently the dominant tree species in plantations in the Ecuadorian Andes (Chacón et al., 2009). Pine plantations have been criticised for their negative effect on hydrological services, such as water retention capacity (Farley et al., 2004; Harden et al., 2013). However, the impact on other hydrological services is not always clear (Hofstede et al., 2002), and the negative impacts can partially be explained by the already degraded state of land on which pine is often planted (Chacón et al., 2009). The native Andean alder is preferred by Loja's watershed managers, because they consider this species better for the provision of hydrological services. Andean alder can improve water quality by decreasing erosion through soil improvement, whereas it also has a better water retention capacity than pine (Knoke et al., 2014). Plantations of this species do however decrease total water discharge compared to pine and pastures (Knoke et al., 2014; Windhorst et al., 2013). Andean alder plantations have furthermore been found to contain relatively high levels of biodiversity (Castaño-Villa et al., 2014).

In summary, landholders in our study faced the following land use options: (i) farmers without forest could continue their current milk production, or they could abandon milk production to participate in the Socio Bosque restoration component or use incentives to establish tree plantations; and (ii) farmers with forest have similar options for their pasture lands, but could additionally decide to fell trees and establish pasture[28] for milk production or preserve forests under the Socio Bosque programme (Table 5.1). In our portfolio analysis these land uses could be combined. To be able to analyse the impact of the incentive programmes we compare these portfolios with land use combinations that also generate revenue, but exclude the incentive programmes. In this case the land use options are: (i) farmers without forest can continue to produce milk or establish tree plantations assuming all establishment costs; and (ii) farmers with forest can consider deforestation to expand their pasture[29]. While landholders participate in the Socio Bosque conservation programme in areas adjacent to our research area (see Chapter 4), at the time of research no landholders had participated started in the restoration component of the programme, and were not (yet) combining these programmes on their properties.

Table 5.1: Land use choices for portfolio analysis

		Milk producers			
		Without forest		With forest	
	Incentives:	Without	With	Without	With
Current land uses	Portfolio options				
Pasture	Continuing milk production	X	X	X	X
	Tree plantations	X		X	
	Incentive programme for tree plantations		X		X
	Socio Bosque restoration incentives		X		X
Forest	Deforestation for new pasture establishment			X	X
	Socio Bosque conservation incentives				X

5.2.3 Data collection

Farmers' household data were collected in the research area through two stages of fieldwork. During the first stage (November-December 2011) 19 farmers were interviewed about their production systems (production of milk, pigs, chicken, and crops). The data collected were used as input in the milk production and non-milk farm income calculations (see Annex 6). In the second stage (March to July 2013) 95 farmers (accounting for about 20% of the land in the research area) were interviewed to gather data on socio-economic characteristics, land uses and milk production systems. The 19 farmers initially interviewed were also part of the second interview round. The portfolio analysis is based on the data from the 95 farmers and utilizes some extra income estimates derived from the first survey.

[28] Under Ecuador's Forestry Law (Law 74, 1981) the conversion of watershed vegetation into farmland and the extraction of timber for commercial purposes are prohibited. Landholders need a permit from the Ministry of the Environment to clear vegetation (possible up to a maximum of 20%). The application procedure is costly and prolonged (Rodríguez de Francisco et al., 2013) and, until recently, this law was rarely enforced (Wunder and Alban, 2008).

[29] In this scenario we did not consider just leaving the forest, as we only considered revenue-generating activities.

Because we lacked data on all the households owning land within the research area, random sampling was not possible. Furthermore, many of the milk producers surveyed lived on the fringes of Loja city. Instead, we used snowball sampling to find respondents. We first approached milk producers to whom we were introduced by watershed managers from Loja's municipal water company. In turn, these farmers introduced us to new respondents, which also helped to gain respondents' trust. Participation was voluntary and verbal prior consent was obtained after respondents were informed about the purpose of the research and use of collected information, and were assured individual anonymity.

Data on government programmes were obtained by contacting government officials and tapping into on-line resources (Socio Bosque, Ministry of Agriculture, FORAGUA and EMALEP).

Milk prices were provided by the Ministry of Agriculture. As data on cattle prices were unavailable we used prices from local cattle markets and data provided by the NGO Nature and Culture International who did a milk production survey in 2006[30]. Historical data on pine and Andean alder prices were unavailable. We collected data on pine prices at 20 local sawmills from 1 to 14 October 2013. None of the sawmills were buying Andean alder wood at the time of research.

5.2.4 Expected return in net present value

As a first criterion to assess impact (Knoke et al., 2008), the expected return (ER) for each of the 95 farms, expressed in net present value (NPV) over 20 years,[31] was calculated for the different land uses by summing the discounted future net revenues from land management:

$$\mathrm{ER_{NPV}} = \sum_{t=0}^{T} r_t (1+i)^{-t} \tag{1}$$

With the expected return expressed as net present value (ER_{NPV}), T as the total period, r_t as the net revenues in time t, and i as the discount rate (Castro et al., 2013). Following Benitez et al. (2006), and Knoke et al. (2009a), a discount rate of 5% was used. From here on, the expected return will be expressed in net present value and presented per ha.

Price volatility is frequently used to model the uncertainty associated with investments in agriculture, because farmers often take production decisions based on prices and their recent fluctuations (Aimin, 2010; Benitez et al., 2006). Hence, we calculated income uncertainty on the basis of price variability, cattle mortality rates and probability of fire damage in tree plantations (see Annex for details). Price variability was based on monthly milk prices from 2000 to 2013, provided by the Ministry of Agriculture. Real prices were compounded using the monthly inflation rate (INEC, 2012d). Prices ranged

[30] This NGO supports FORAGUA's management. The survey was carried out to estimate milk producers' opportunity costs.

[31] The duration of Socio Bosque contracts is 20 years, which is also the production cycle for pine and Andean alder plantations.

from US$ 0.29/l to US$ 0.45/l, with an average of US$ 0.36/l. We assume that the probability of occurrence of future prices is equal to the number of times they occurred in our data series (see Annex 6). For pine, the surveyed sawmills reported prices between US$ 1 and US$ 30/m^3, with an average price of US$ 15/m^3. Prices varied strongly between sawmills. We assumed equal probability of occurrence of these prices. No data on Andean alder were available. As Andean alder is seen as a potential substitute for pine plantations in the research area, we assumed that it would fetch the same prices as pine. Annex 6 gives more detail on data used and income calculations of milk production and tree plantations.

The costs and benefits of deforestation and establishing new pasture were based on compounded data on pasture establishment in the buffer zone of Podocarpus National Park of Knoke et al. (2009b). The costs of having cows on new pastures were estimated from market prices for cattle. Alternatively, if a farmer abandons pasture for restoration or tree plantations, cows could simply be relocated on new pastures. Yet, it was difficult to predict in advance how many head of cattle would be available to move after pasture abandonment. Instead, we included sales income from cattle that were grazing on land that would be abandoned for restoration and tree plantations so as to avoid underestimation of the expected returns of portfolios with deforestation. Calculations of milk production on pasture established after deforestation were similar to those of continuing milk production for the subsequent 19 years.

Dairy farmers with forest could decide to conserve forest through Socio Bosque. The conservation incentives were US$ 60/ha for farms of less than 20 ha. For farms of more than 20 ha, incentives were US$ 30/ha for up to 50 ha of forest. The incentives per ha decreased as more forest was included (Ministry of the Environment, Ecuador, 2012c) (see Annex 1 and 6). One respondent owned 110 ha of forest, and would receive less for each additional hectare after the first 50 hectares. However, to simplify portfolio modelling, this reduced payment was not taken into account.

Because Socio Bosque contracts were fixed for 20 years, the revenues resulting from participation in the scheme should remain constant. However, as the budget was mostly allocated by the government (see Chapter 4), the government could change the programme (for example if oil prices dropped) or even stop payments (for example after elections). These uncertainties are not easily quantifiable. We follow Knoke et al. (2011) who used a variation coefficient of 5% for carbon payments to take into account any fluctuations in the programme payments, as carbon credits are part of the financial sustainability strategy of Socio Bosque (Ministry of the Environment, Ecuador, 2012).

To calculate the expected revenue from milk production and tree plantations, and their standard deviations, simulation models were created in R (R Core Team, 2014). Sampling was based on probability of price occurrences, cattle mortality, and seedling survival and fire damage. The model was run 1,000 times for each farmer. The expected return per farmer used in the further analyses was the sum of the discounted means of the outcomes (net revenue per year) obtained per run. A similar approach was used to calculate the expected return for both types of tree plantation, but here we ran the model 10,000 times each.

5.2.5 Portfolio theory

Minimum variance portfolio

We used portfolio analysis to compare and combine land use activities at different ratios based on their expected financial returns and standard deviations. The standard deviation of expected return indicates the degree of uncertainty of an investment (Knoke et al., 2005). It is reduced when investing in activities with a high covariance (Markowitz, 1952). A large variety of land use combinations are possible, giving different expected returns and standard deviations, creating the 'achievable region', which is delimited by the portfolio frontier curve (Figure 5.2). Following Brealey et al. (2011) the expected return of a portfolio *p* of land uses is given by:

$$ER_p = w_1 ER_1 + \cdots + w_i ER_i \qquad (2)$$

With ER_1 the expected return of land use 1, ER_i the expected return of land use *i*, w_1 weight of land use 1 in the portfolio, and w_i weight of i-th land-use.

The portfolio variance of a portfolio with two assets is given by:

$$\sigma_p^2 = w_1^2\sigma_1^2 + w_2^2\sigma_2^2 + 2w_1w_2\sigma_{1,2} \qquad (3)$$

Where σ_p^2 is the portfolio variance, σ_1^2 and σ_2^2 the variances of assets (land uses), and $\sigma_{1,2}$ the covariance of the two assets.

For multiple assets we use matrix annotation:

$$\sigma_p^2 = [w_1, \dots, w_i]\begin{bmatrix}\sigma_1^2 & \cdots & \sigma_{i,1}\\ \vdots & \ddots & \vdots\\ \sigma_{1,i} & \cdots & \sigma_i^2\end{bmatrix}\begin{bmatrix}w_1\\ \vdots\\ w_i\end{bmatrix}$$; which can be written as

$$\sigma_p^2 = W'\Omega\, W \qquad (4)$$

Where W is an Nx1 vector of asset weights, and Ω is an NxN land use covariance matrix, and N is the number of assets considered.

Covariances were estimated from 20-year simulations of yearly expected returns series (expressed in net present value), bootstrapped using the R package boot, to obtain correlation coefficients (Canty and Ripley, 2015; Davison and Hinkley, 1997).

The covariance is defined as:

$$\sigma_{ij} = \rho_{ij}\sigma_i\sigma_j \qquad (5)$$

Where ρ_{ij} is the correlation coefficient of two assets, and σ_i and σ_j their standard deviations.

Assumptions were that a) expected returns were normally distributed, b) farmers cared only about mean return and variance, c) land uses were interchangeable[32] before the start of the analysis, d) farmers' milk production systems remained stable, and e) transaction costs were zero.

The *minimum variance portfolio* combines land uses to obtain the lowest standard deviation and, therefore, lowest sensitivities to risk. Subject to covariance levels, it makes maximum use of diversification to achieve a risk level that can be lower than each of the individual land uses it contains. Specific portfolios will provide maximum reward (highest expected return) for a given risk, creating the efficient frontier of portfolios (Markowitz, 1952). This frontier is the part of the portfolio curve for which economic returns are larger (or equal) than those of the minimum variance portfolio (Figure 5.2). An investor would be expected to choose a portfolio on the efficient portfolio frontier (Brealey et al., 2011).

Weights of different land uses in a minimum variance portfolio of two land uses are calculated as:

$$w_{1,MVP} = \frac{\sigma_1^2 - \sigma_{1,2}}{\sigma_1^2 + \sigma_2^2 - 2\sigma_{1,2}} \tag{6}$$

$$w_{2,MVP} = 1 - w_{1,MVP} \tag{7}$$

Subject to: $w_1 + w_2 = 1$

Where $w_{1,MVP}$ is the portfolio weight of the first asset, $w_{2,MVP}$ of the second asset.

For more than two assets matrix annotations are used. The minimum variance objective function is a minimization of estimated portfolio risk:

$$\sigma_p^2 = W'\Omega\, W \tag{8}$$

The minimum variance portfolio is the solution to the following minimization problem:

$$W_{MVP} = \sigma_{MVp}^2\, \Omega^{-1}\dot{1} \tag{9}$$

where W_{MVP} is a Nx1 vector of portfolio weight, σ_{MVp}^2 is the minimum variance, and $\dot{1}$ is an Nx1 vector of ones (Clarke et al., 2013).

Sharpe ratio and the optimal portfolio

The minimum variance portfolio has been used to analyse risk-reducing land-use combinations (e.g. Estrada et al., 2006; Reeves and Lilieholm, 1993). However, according to Knoke et al. (2011), modelling land use allocations under risk aversion should not only imply land use combinations that minimize variance, but should also consider the combination that gives the highest return per unit of risk. The latter is achieved through an *optimal portfolio land use allocation*. In this analysis we consider both the minimum variance and optimal portfolio land use combinations to illustrate how land should be allocated to the available

[32] At the start of the analysis farmers can potentially choose any of the land uses, once the 20 year period starts, we considered that the land uses can no longer be changed.

practices to achieve absolute risk minimization or maximization of the return per unit of risk, and to analyse the impact of the incentive programmes on land use allocation.

Calculating the portfolio that gives the highest return per unit of risk requires defining a risk-free asset. Following Knoke et al. (2011) we consider income from land sales as risk-free. The selling price is fixed at US$ 400/ha, which equals the compensation payment that Loja municipality disburses when negotiating land acquisition to establish protected watersheds[33] (Cevallos J.C. personal communication, 2013). Obviously, selling land also entails risk as farmers lose a source of future income. Moreover, it is drastically different from the land uses considered. However, even government programmes with long-term fixed payments are uncertain due to possible changes in government policies. Whereas, once land is paid for, the money is cashed and farmers face no uncertainty over future expected revenues, which is why we consider selling land as being without income risk.

Figure 5.2 illustrates the capital allocation line which represents the combination of risk-free and risky activity in which the standard deviation only changes according to the weight of the risky activity in the combination (Bodie et al., 2008). The slope of the capital allocation line is the reward-to-variability ratio or Sharpe ratio. It represents a risk premium because it is the net return of the portfolio for one unit increase in variance/volatility (Sharpe, 1994).

Mathematically, the Sharpe ratio is calculated by (McDonnell, 2008):

$$Sharpe\ Ratio\ (slope) = (ER_p - R_r)/\sigma_p \qquad (10)$$

With ER_p the expected return of the risky portfolio, R_r the return on the risk-free activity, and $\sigma_p$the volatility of the risky portfolio. The variance of the risk-free asset is zero.

The optimal portfolio we considered here is the one on the efficient frontier with the highest Sharpe ratio (Prigent, 2007), and is the point where the portfolio frontier is tangent to the capital allocation line (Figure 5.2).

The weight of 1 asset (w_1) in an optimal portfolio of two assets and a risk-free option is given as follows:

$$w_{1,OP} = \frac{(ER_1 - \mathrm{R_r})\sigma_2^2 - (ER_2 - R_r)\sigma_{1,2}}{(ER_1 - \mathrm{R_r})\sigma_2^2 + (ER_2 - \mathrm{R_r})\sigma_1^2 - (ER_1 - R_r + ER_2 - R_r)C\sigma_{1,2}} \qquad (11)$$

$$w_{2,OP} = 1 - w_{1,OP}$$

Matrices and vectors are used when more than two land uses are to be included. Consider ER a vector of the expected returns of the assets (land uses), and W a vector of allocations of the risky assets, the maximization problems is defined by (Campbell and Viceira, 2002):

$$\max_{\boldsymbol{w}} W'\left(ER - R_r\,\dot{1}\right) - \frac{1}{2}W'\Omega\,W \qquad (12)$$

[33] This official land price is relatively new. The price paid in 2009 and 2010 for properties in the El Carmen watershed by the municipality of Loja was on average US$ 435/ha (see also Chapter 4).

Where $\left(ER - R_r\,\dot{1}\right)$ is the vector of excess returns on the N risky assets over the return from the riskless asset. The variance of the portfolio return is $W'\Omega\, W$.

The solution to this maximization problem is:

$$W = \Omega^{-1}\left(ER - R_r\dot{1}\right) \quad (13)$$

The minimization and maximization problems were solved with the R package quadprog (Turlach and Weingessel, 2013).

Portfolio constraints

When analysing land use portfolios, a key requirement is that the portfolio weights are larger than, or equal to, zero, or mathematically that:

$$w_i \geq 0,\ \ i \in N \quad (14)$$

An additional constraint added to the analysis was that a minimal area for each cow owned had to be available. As cows are indivisible, the area farmers could allocate to pasture for milk production was either zero, equal to the area each of them used per cow, or any multiple thereof.

For each milk producer with forest we expressed the expected return, standard deviation and Sharpe ratios of continuing milk production on the existing pastures, as part of a portfolio that included the current forest (Section 5.4.2). The weight of continuing milk production was equal to the percentage of pasture in relation to the total area (forest and pasture). The calculations were carried out using equations 2, 3 and 10, where forest had an expected return and standard deviation of zero. Expressing current milk production as a portfolio facilitated comparing this activity with a portfolio combining several land uses on pasture and forest.

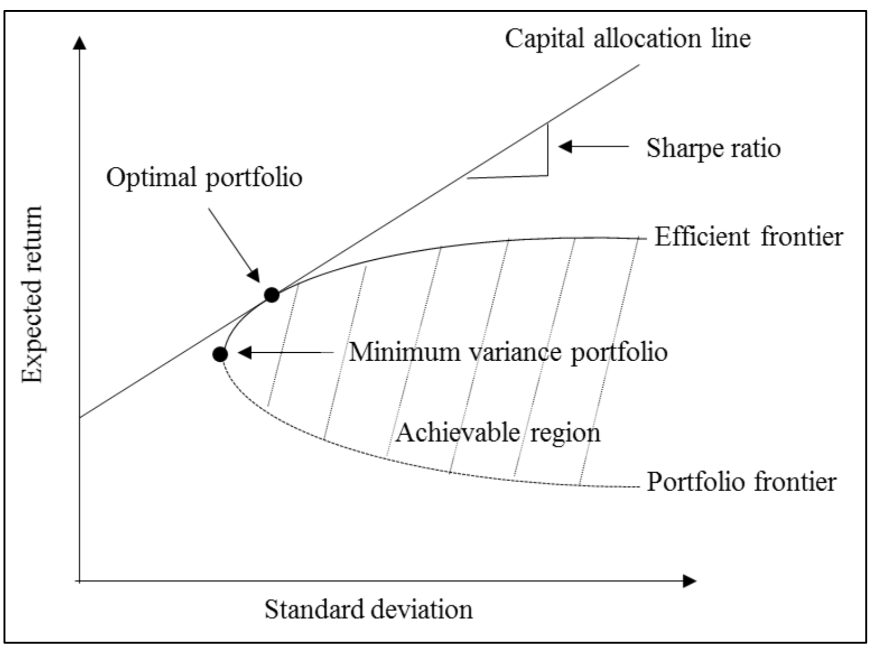


Figure 5.2: Sharpe ratio, portfolio frontier, and minimum variance and optimal portfolio

Household income and poverty

Three indicators were calculated to quantify the impact of the different portfolios on household income, namely: (1) total household income and income per household member; (2) the effect of the portfolio choice on households crossing the poverty line; and (3) change in household income. While the first indicator is straightforward, the second indicator is inspired by the potential contribution PES schemes could have on poverty reduction. While poverty is a multidimensional concept (Davies et al., 2014) that cannot be grasped only by income indicators (Scheidel, 2013), one potential contribution of PES schemes could be to reduce the number of people living below the poverty line. Here we used the 2011 Ecuadorian poverty line of US$ 874 per household member (INEC, 2011b). Finally, we analysed the change in household incomes in each portfolio. For this we divided the milk producers without and with forest into quartiles according to percentage of income received from milk production. For calculating household income including land income from the different portfolios, the annuitized expected return of the different land uses was used and added to the other income sources.

5.3. Results

5.3.1 Socio-economic data

Of the 95 farmers interviewed in 2013, 58 did not have forest on their land, 37 did. The non-forest holders owned a total of 332 ha of pasture. The farmers with forest had a total of 430 ha pasture and 312 ha forest land. Over the total sample, 68% of land was pasture, 28% forest and 4% cropland, which is similar to the figures available for the municipality of Loja (60%; 33% and 7% respectively). The average farm size of our sample was 6.1 ha for milk producers without forest and 20.1 ha for those with forest (see also Supplementary Table 6.1), compared to an average farm size of 9.5 ha for the whole municipality (Ministry of Agriculture, Ecuador, 2012). Average cattle density was below the carrying capacity of 2.3 for native pastures and 3.4 for improved pastures in the Ecuadorian Andes (Barrera et al., 2010), but most farms should still be considered as high-input (Knoke et al., 2014). The daily average production of six litres of milk per cow is below the average for the Ecuadorian Andes (INEC, 2011c). The percentage of households in the sample below the poverty line is the same as the Ecuadorian average of 29% of the population (INEC, 2011b).

Households mainly derive their income from off-farm activities, with non-milk farm income, mainly production of pigs, vegetables, maize and potatoes (sale and subsistence) being second. Milk production accounted on average for around 20% of the income of forest owners and 18% for non-forest holders. On average, farmer households were to be found in the third decile of Ecuadorian household income categories (INEC, 2012c) (see also Supplementary Table 6.2).

Forest owners had on average[34] more pasture, more cows, and a higher total income from milk production than non-forest owners. However, the latter group reported on average a higher milk production per cow, and more income per hectare. On average forest owners derived more farm income from non-dairy sources than farmers with no forest. Off-farm income did not significantly differ between the two groups (Table 5.2).

Table 5.2: Production and household income characteristics

	Milk producers			
	Without forest (N=58)		With forest (N=37)	
	Average	Median	Average	Median
Total land area (ha)	6.1	4.5	20.6	10.5
Pasture area (ha)	5.7	4	11.6	8
Crop area (ha)	0.4	0	0.6	0
Forest area (ha)	/	/	8.4	3
Number of cows	4	3	8	5
Total heads of cattle	9	7	18	11
Milk production (litres)	6.4	6.8	5.8	6.5
Density units (cattle/ha)	1.2	1.2	1.1	1.1
Off-farm income/household (US$/year)	4,307	3,552	3,749	3,504
Non-milk farm income (US$/year)	628	200	1,700	370
Milk income (US$/year)	617	544	1,003	491
Total household income (US$/year)	5,552	4,489	6,453	4,299
Income per household member (US$/year)	1,592	1,183	2,011	1,363

5.3.2 Expected return on activities

Non-forest owners had on average a higher total expected return per hectare from continuing milk production than forest owners, US$ 640/ha more (Table 5.3). For the latter group, the expected return from milk production on existing pastures did not differ much from that on newly established pastures after deforestation. There were no significant differences between the two groups when income from tree plantations (with and without incentives) and restoration incentives were compared.

From the perspective of simply maximizing the expected return, a majority of milk producers without forest would continue with their current milk production, instead of establishing timber plantations when incentives are not included. For this group, without incentives establishing tree plantations on pasture would give an average loss of US$ 457/ha and US$ 311/ha for Andean alder and pine, respectively. However, when the incentives are included, the average loss would be only US$ 9/ha for Andean alder, whereas the establishment of pine plantations would give an average gain of US$ 268/ha. Furthermore, the majority of farmers without forest would not consider restoration incentives, as the average loss of abandoning pasture to receive the restoration payments would be US$ 408/ha.

[34] All differences were analysed using statistical tests with the R software (R Core Team, 2014). When differences are illustrated in the text, this is because they were statistically significant at least at the 10% confidence level.

For the majority of forest owners, the expected return from timber plantations with and without the additional incentives is higher than from continuing their current milk production. Without incentives they would gain on average US$ 138/ha from establishing Andean alder plantations on their pastures, and US$ 285/ha from pine plantations. With incentive the average gain would increase to US$ 588/ha and US$ 863/ha for Andean alder and pine plantations, respectively. Additionally, a large share of forest holders would gain from receiving restoration incentives instead of producing milk, on average US$ 121/ha. However, conservation incentives were lower than the expected return from milk production after deforestation for most forest holders, on average US$ 103/ha lower.

Table 5.3: Average expected returns, average standard deviation of expected returns, and percentage of farmers with a higher expected return for non-milk production activity on pasture or forest

	Milk producers			
	Without forest	With forest	Without forest (N=58)	With forest (N=37)
	Average expected return in US$/ha (average standard deviation)		% farmers with higher expected return than milk production	
Milk production	1,542 (215)	902 (175)	/	/
Andean alder plantation[a]	1,085 (652)	1,040 (652)	31	68
Pine plantation[a]	1,231 (560)	1,187 (560)	36	73
Andean alder plantation[a] with incentives	1,533 (672)	1,490 (672)	55	76
Pine plantation[a] with incentives	1,810 (573)	1,765 (578)	66	86
Restoration payments[a]	1,134 (137)	1,023 (136)	33	68
Milk production after deforestation	/	819 (225)	/	/
Conservation payments	/	716 (47)	/	41

[a] Includes income from selling cattle (on average US$/ha 485)

5.3.3 Portfolio analysis

Pasture land use allocation for milk producers without forest

A first set of portfolios was calculated for every milk producer without forest. In a first step we considered the portfolios with Andean alder (Figure 5.3 and Table 5.4), followed by a comparison of Andean alder and pine portfolios (Supplementary Table 6.11).

Whereas in both minimum variance scenarios a statistically significant area of pasture was abandoned for alternative land uses (tree plantation and/or restoration), this shift was significantly larger when we considered the incentive programmes. In the latter case on average 75% more portfolio weight, or an average of 3.7 ha more pasture, was allocated to alternative land uses than in the scenario without the incentives. However, the majority of this shift went to restoration activities, as the portfolio weight and area dedicated to Andean alder was significantly lower in the minimum variance portfolio with incentives than without. In the minimum variance land use allocation with incentives, around 48% of farmers would not dedicate any land to milk production, but without incentives 66% would continue with existing milk production. The land use change was reflected in the difference in the expected return between both minimum variance portfolios, which was significantly lower (on average US$/ha 355 less) in the allocation with the incentive programmes.

Significant changes in portfolio weights and land area were observed in both optimal land use allocation scenarios. As in the previous observations, both the portfolio weight and area dedicated to milk production were significantly lower when taking the incentive programmes into account, respectively on average 39% less weight or 1.8 ha less pasture. This difference was due to the importance of land allocated to restoration, as the land dedicated to tree plantations was on average much higher (2.2 ha) in the optimal portfolio without incentives. In the optimal portfolio with incentives ten percent of farmers would continue current milk production, as opposed to 66% in the land allocation without the incentives. Furthermore, when incentives are considered around 36% of farmers would not allocate any land to milk production. However, the difference in expected return between both optimal portfolios was not significant.

When we compared the weights, land areas and expected returns of the portfolios with Andean alder plantations with those including pine, we found no significant differences. The exception to this was the portfolio weight and area allocated to tree plantations in the minimum variance scenarios without the incentive programmes. In these scenarios on average more weight (5% more) and area (0.2 ha more) were allocated to pine plantations than to Andean alder plantations.

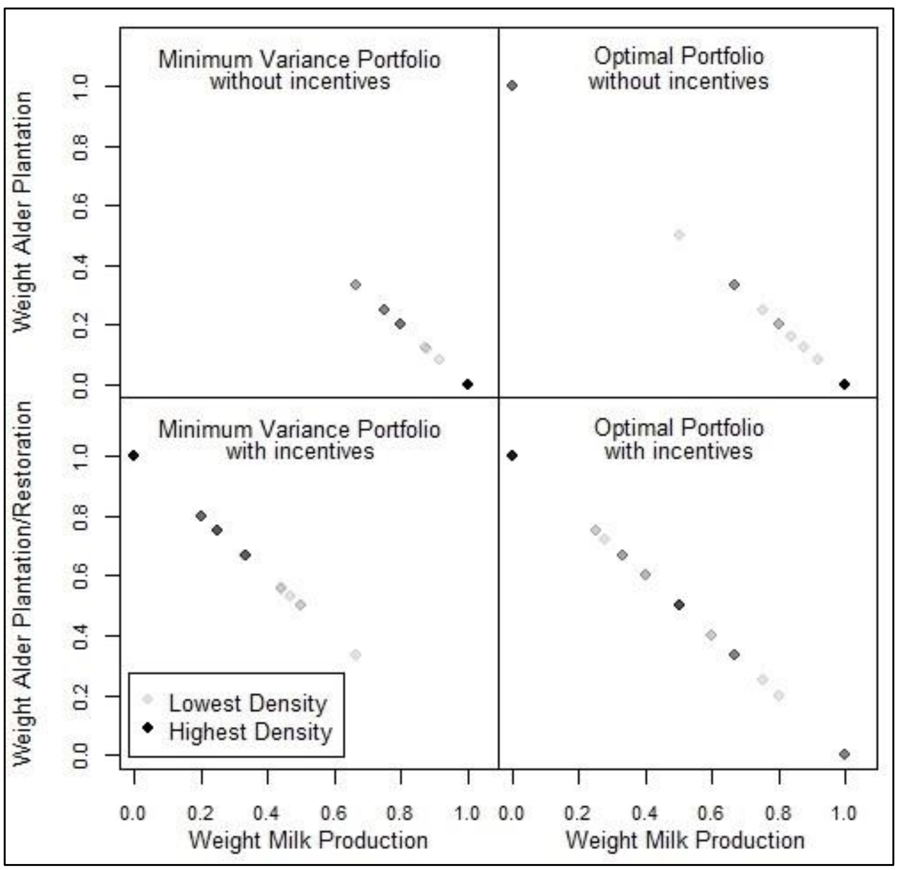


Figure 5.3: Portfolio of land use allocations of non-forest holders with and without incentives

Table 5.4: Average output of portfolios with Andean alder for milk producers without forest

		Current situation	Change through minimum variance portfolio		Change through optimal portfolio	
	Incentives:	Without	Without	With	Without	With
Average area (ha)	Pasture	5.7	- 0.4	- 4.1	- 2.3	- 4.0
	Plantation	/	+ 0.4	+ 0.0	+ 2.3	+ 0.1
	Restoration	/	/	+ 4.1	/	+ 3.9
Average portfolio weights	Pasture	1.00	- 0.08	- 0.83	- 0.24	- 0.63
	Plantation	/	+ 0.08	+ 0.00	+ 0.24	+ 0.01
	Restoration	/	/	+ 0.82	/	+ 0.62
Average expected return (US$/ha)		1,542	- 62	- 417	+ 128	+ 37
Average standard deviation of ER/ha		215	- 7	- 84	+ 91	- 51
Average Sharpe ratio		4.4			+ 1.2	+ 3.0

Pasture and forest land use allocation for milk producers with forest

The different portfolio scenarios considered for every forest owner also created significant changes in land use allocation compared to continuing milk production on existing pastures (Figure 5.4, Table 5.5). The weight and area of pasture for milk production under the minimum variance land use combination with the incentive programmes were significantly lower than in the minimum variance portfolio without incentives: on average 46% and 8.5 ha less, respectively. Similar to the scenarios for milk producers without forest, this difference is due to the significant weight and area allocated to restoration activities, as the weight and area allocated to Andean alder plantations were significantly higher in the minimum variance scenario without incentives; on average 10% (or 1.2 ha) higher. In the scenario with incentives around 24% of farmers would stop all milk production, whereas without incentives 14% would allocate all their land to milk production. The impact of the conservation payments can be observed through the difference in deforestation between both portfolios. This was significantly lower in the portfolio with incentives, with an average reduction in deforestation of 68% of the forest area. However, even with incentive payments, 24% of forest holders would not allocate any land to conservation. These observed differences did not, however, translate into significant differences in the expected return of both minimum variance land use combinations.

The differences between both optimal land use scenarios were similar to those for the minimum variance portfolios. On average significantly more pasture was abandoned under the scenario with incentives (5.3 ha) than without, although a larger area was dedicated to tree plantations (3.2 ha more) in the portfolio without incentives. The optimal portfolio land use combination with incentives reduced deforestation, with an average of 52% of the forest land conserved. Moreover, around 41% of farmers would not allocate any land (pasture or forest) to milk production under the incentive scenario. By contrast, 30% would not conserve any forest, and two farmers of this group would dedicate all their pasture and forest land to milk production. There were no significant differences between the expected return of the two optimal portfolios.

When we compared the weights, land areas and expected returns of the portfolios with Andean alder plantations with those including pine plantations (Supplementary Table 6.12), we found no significant differences.

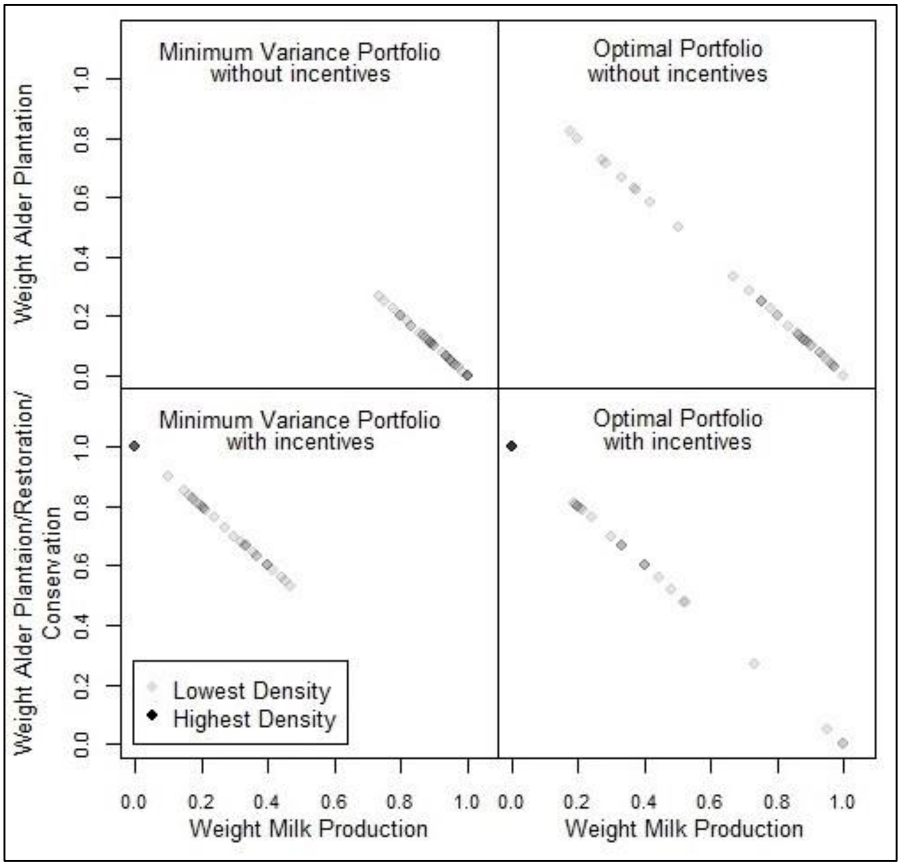


Figure 5.4: Land allocation for milk producers with forest, considering both current forest and pasture land

Table 5.5: Average portfolio output for milk producers with forest considering Andean alder plantations

		Current situation	Changes compared to current situation			
			Minimum variance portfolio		Optimal portfolio	
	Incentives:	Without	Without	With	Without	With
Average area (ha)	Pasture	11.6	- 1.3	- 9.8	- 3.2	- 8.6
	Plantation	/	+ 1.3	+ 0.1	+ 3.2	+ 0.1
	Restoration	/	/	+ 9.7	/	+ 8.4
	Deforestation	/	+ 8.4	+ 2.7	+ 8.4	+ 4.0
	Conservation	8.4[a]	- 8.4	- 2.7	- 8.4	- 4.0
Average portfolio weights	Pasture	0.68	- 0.10	- 0.56	- 0.27	- 0.55
	Plantation	/	+ 0.10	+ 0.01	+ 0.27	+ 0.01
	Restoration	/	/	+ 0.55	/	+ 0.54
	Deforestation	/	+ 0.32	+ 0.10	+ 0.32	+ 0.13
	Conservation	0.32[a]	- 0.32	- 0.10	- 0.32	- 0.13
Average expected return (US$/ha)		613 (902[b])	+ 285	+ 300	+ 442	+ 473
Average standard deviation of ER/ha		107 (159[b])	+ 21	- 45	+ 102	- 33
Average Sharpe ratio		0.19 (2.12[b])			+ 4.5	+ 9.4

[a] Current standing forest, not active conservation through Socio Bosque
[b] Milk production data not taking into account forest area

5.3.4 Sensitivity analysis

As the calculation of the portfolio that maximizes the Sharpe ratio is sensitive to the value of the risk-free asset, we also carried out a sensitivity analysis. We increased the value of land to US$/ha 600. We only considered the portfolios with Andean alder.

A few differences can be observed with respect to the previous optimal land use allocations (Tables 5.4 and 5.5). For milk producers without forest a larger portfolio weight and area was dedicated to milk production under both optimal portfolio scenarios (Table 5.6). Under the optimal scenario without incentives less weight and area were, on average, dedicated to Andean alder plantations than in the previous analysis. However, for the optimal portfolio with incentives the weight and area of tree plantations were larger than in the previous optimal portfolio. The most significant differences that we observed were the decreases in the portfolio weight and area allocated to restoration.

Under the scenario with higher valued risk-free asset, the optimal portfolio without incentives for forest holders included slightly more Andean alder plantations, although there were no changes in the number of farmers who would allocate all their pasture to Andean alder plantations. In the optimal portfolio with incentives, less pasture is abandoned, and significantly more weight and area is allocated to Andean alder plantations. There was further a decrease in the area under restoration and a slight decrease in the area under conservation.

Table 5.6: Output optimal portfolios with higher risk-free value, considering Andean alder plantations

		Current situation	Changes compared to current situation for non-forest holders		Current situation	Changes compared to current situation for forest holders	
	Incentives:	Without	Without	With	Without	Without	With
Average area (ha)	Pasture	5.7	- 2.1	- 3.6	11.6	- 3.3	- 8.1
	Plantation	/	+ 2.1	+ 0.9	/	+ 3.3	+ 1.5
	Restoration	/	/	+ 2.7	/	/	+ 6.6
	Deforestation	/	/	/	/	+ 8.4	+ 6.1
	Conservation	/	/	/	8.4	- 8.4	- 6.1
Average portfolio weights	Pasture	1.0	- 0.21	- 0.56	0.68	- 0.28	- 0.50
	Plantation	/	+ 0.21	+ 0.03	/	+ 0.28	+ 0.05
	Restoration	/	/	+ 0.53	/	/	+ 0.45
	Deforestation	/	/	/	/	+ 0.32	+ 0.15
	Conservation	/	/	/	0.32	- 0.32	- 0.15
Average expected return (US$/ha)		1,542	+ 143	+ 117	613	+ 454	+ 554
Average standard deviation of ER/ha		202	+ 81	- 35	107	+ 114	- 1
Average Sharpe ratio		3.2	+ 1.5	+ 2.7	0.2	+ 5.7	+ 9.3

5.3.5 Impact of land use allocations on household income

We now turn our attention to the impact of different land use combinations on household income. On average, for farmers without and with forests, total household income and income per household member did not differ significantly between the different land use allocation scenarios studied (Table 5.7). However, land use, as predicted by the minimum variance portfolio with incentives, would push a number of milk producers without forest into poverty. By contrast, for milk producers with forest land, all the portfolios were found to have a positive impact on the number of households below the poverty line.

Table 5.7: Total Income, income per household member, % households below the poverty line for different land use portfolios

		Milk producers without forest N=58		Milk Producers with forest N=37	
	Incentives	Without	With	Without	With
Minimum variance portfolio	Total HH income (US$)	5,413	5,299	6,933	6,573
	Income/HH member (US$)	1,546	1,507	2,200	2,078
	% HH below poverty line	29	33	19	19
Optimal portfolio	Total HH income (US$)	5,638	5,480	7,134	6,803
	Income/HH member (US$)	1,622	1,572	2,265	2,145
	% HH below poverty line	28	29	19	19
Continuing milk production	Total HH income (US$)	5,419		6,224	
	Income/HH member (US$)	1,547		1,930	
	% HH below poverty line	29		30	

We also analysed how much the incomes of farmers would change if they were to adopt the land uses predicted by the minimum variance and optimal portfolios. Because these changes may have distributional effects, they were calculated per quartile, according to percentage of income derived from milk production. These quartiles were calculated separately for milk producers with and without forest [35] (Table 5.8).

For households without forest, land use combinations in the minimum variance portfolio with incentives would decrease their income, except for those in the first quartile. The change in household income was not significant under the minimum variance portfolio without incentives. Adoption of the optimal portfolio with incentives would only significantly increase the income of households in the first quartile. This increase would be even higher for this quartile when adopting the optimal portfolio without incentives. For milk producers with forest, positive income effects were found across all portfolios and quartiles and with no significant differences between the portfolios with and without incentives.

Table 5.8: Average change in total household income and number of households below the poverty line per milk income quartile for the land use portfolios

	Producers without forest				Producers with forest			
Quartile	1st	2nd	3rd	4th	1st	2nd	3rd	4th
N	14	15	14	15	10	9	9	9
Change in total household income without incentives (%)								
Minimum variance portfolio	+ 0.0	+ 0.1	- 0.2	- 0.3	+ 5	+ 10	+ 12	+ 27
Optimal portfolio	+ 23	+ 0.2	+ 0.4	- 0.3	+ 17	+ 14	+ 13	+ 27
Change in total household income with incentives (%)								
Minimum variance portfolio	+ 6	- 2	- 4	- 18	+ 13	+ 11	+ 8	+ 6
Optimal portfolio	+ 13	- 1	- 1	- 4	+ 16	+ 13	+ 10	+ 11

5.4. Discussion

The results of the portfolio analysis provide insights into those land use allocations that could have risk reduction effects, and the impact of adopting these land use combinations on household income generation.

First, although introducing risk reduction positively influenced the adoption of PES, opportunity costs had a strong impact on land use combinations maximizing returns per unit of risk. In the optimal portfolio allocation of milk producers without forest, the most productive farmers - with higher opportunity costs - would not consider restoration or tree plantations. Socio Bosque's restoration incentives would be too low for them to contemplate abandoning pastures. Lower opportunity costs have been linked to increased participation in existing PES programmes (e.g. Zanella et al., 2014). Furthermore, we were not able to account for a possible intensification of milk production systems on pasture when part of land would be reallocated to restoration and tree plantations. Farmers could choose to increase production on remaining pasture if they abandoned land. This would affect the land use combinations we obtained through our portfolio approach. With intensification higher payments would be needed to achieve the same land use allocation (see also Carpentier et al., 2000; Knoke et al., 2011).

[35] Quartiles for milk producers without forest were: (1) less than 6%, (2) between 6-12%, (3) between 12-23%, and (4) more than 23% of total income from milk production; and for milk producers with forests: (1) less than 6%, (2) between 6-14%, (3) between 14-24%, and (4) more than 24%.

Producers with forest would also restore a considerable area of pasture land according to the portfolios with incentives. Additionally, they would allocate more forest to conservation than to deforestation. The reduction in deforestation was large compared to the portfolios without incentives, where we assumed that farmers were allowed to deforest all their land. Similarly, Knoke et al. (2011) found that in a land allocation that maximized Sharpe ratios, landholders would cut down all their forests without incentives. However, it is generally only possible to fell about one hectare of forest per year (Knoke et al., 2009b). Furthermore, according to Bremer et al. (2014a) legal restrictions on land use and biophysical constraints often limit the conversion of land conserved through Socio Bosque to other uses.

Second, for milk producers with forest we found indications of a potential leakage effect of PES. Leakage occurs when the implementation of conservation/restoration measures in one area provokes ecosystem degradation elsewhere (Wunder, 2007). In our analysis this occurred within individual farms through the allocation of pasture land for restoration and thus the abandonment of milk production, while simultaneously allocating forest land to deforestation to start milk production on newly established pastures. This negative environmental impact should be considered when implementing both components of the Socio Bosque programme simultaneously.

Third, the current incentive system may only promote pine plantations. In general, we did not find differences between the portfolio allocations of pine and Andean alder. Given that, to date, more pine has been planted and sawmills are currently only using pine, there may thus be no additional incentive to plant Andean alder plantations. Promotion of Andean alder among sawmills, and promotion of the incentive programme itself, could potentially change this situation. In addition, our results showed that the Socio Bosque incentives would have significantly more impact on shifting land uses than incentives for timber plantations. The latter land use had more uncertain expected returns. Moreover, land management that focuses on conservation and restoration through natural succession does not require such specialised technical knowledge as engaging in forestry enterprises (Pagiola et al., 2008; Zbinden and Lee, 2005).

Fourth, the sales value of land and land use regulations influenced the portfolio allocations. Increasing the value of the risk-free asset resulted in less land being allocated to those activities with the lowest expected return. For most farmers this would mainly decrease the areas allocated to Socio Bosque. These observations are in line with Knoke et al., (2011) who found that an increase of the risk-free asset would require higher PES incentives. Although the value we attributed to land (US$/ha 400) may be quite low compared to the expected return from milk production for most farmers, other aspects have to be taken into account. We considered this as a minimum land value, compared to a potentially higher, but more uncertain market value. Additionally, this value has been established by Loja municipality, and is related to the implementation of a municipal ordinance regulating land uses in the watersheds and their surrounding areas (Municipality of Loja, 2007). This ordinance establishes restrictions on activities that can be carried out in the research area and even allows for expropriation of land when deemed necessary. Although the municipality has, to date, always negotiated land sales, the value they put on land may discourage other potential buyers from offering a higher price. In addition, according to Wunder and Alban (2008), even modest law enforcement efforts can stimulate greater participation in a PES scheme by decreasing the expected returns from other land uses. As such, PES can also be understood as a potential compensation mechanism which aims to make the implementation of regulations more acceptable (Pagiola, 2008).

Finally, in the last part of our analysis we investigated the impact on income and poverty of adopting the different portfolios. We found that Socio Bosque's aim of reducing poverty through payments would not be achieved among the farmers we studied. The differential payments based on land size do not seem to be having the desired effect since those with a larger land area have more potential to diversify their land use, especially given our area-per-cow constraint. Milk producers with larger farms also tend to have (more) forest and, as such, receive comparatively larger payments for conservation. Increased farm size has been positively related with PES participation (Duke et al., 2014; Zbinden and Lee, 2005). In our study, the Socio Bosque payments made were not sufficiently high to lift all milk-producing households above the poverty line. This problem has been observed in other PES programmes (e.g. Mahanty et al., 2013). It has been suggested that PES initiatives are often ineffective in involving poor landowners, who lack access to sufficient resources to devote to ecosystem provision (Grieg-Gran et al., 2005). In addition, we noted negative impacts on household income among farmers without forest if the portfolios with the incentive programmes would be adopted. The strongest impact was observed among households of the fourth quartile, whose income depended most on milk production.

These results point towards a trade-off between increased income and the improved provision of ecosystem services within the current programmes. An additional problem with the Socio Bosque programme is that land use is fixed for 20 years, which blocks the option to sell land, and thus decreases the liquidity of this land. Without the programme, farmers could sell some of their land at times of financial stress. Poverty might be addressed more effectively with direct transfers (Rodríguez et al., 2011), as already happens in Ecuador, or by improving productivity and off-farm labour options instead of participation in PES schemes. Additionally, PES schemes that focus on productive land uses may be more interesting for poor landholders (Cao et al., 2009). However, results from elsewhere suggest that PES programme participation may reduce the farm household resources committed to agricultural production (Börner and Wunder, 2012). Chang and Boisvert (2009) found statistical evidence that decisions to participate in conservation/restoration programmes and off-farm work were correlated. For many farm households, a decision to shift household labour from agricultural production to off-farm work may well combine with a decision to remove land from agricultural production.

5.5. Remarks

In our portfolio analysis we considered farmers as investors who base their land use decisions on the expected return and the certainty of that return, influenced by risk aversion. There is however a broader range of aspects that influence such decision making, which also depends on the information farmers have and use (Engle Warnick et al., 2011) and the relative risk and expected utility farmers attach to the different land uses (Hildebrandt and Knoke, 2011). Time preferences of milk producers are another important aspect in this regard and these can be accounted for by using different discount rates: Knoke et al. (2014) used a 5% and 8% discount rate in their analysis of abandoned pasture restoration.

In addition, there is a broad range of risk reduction strategies, such as insurance for forestry production, or contracts to assure a less volatile milk price. However these are currently not used in the research area. Furthermore, optimal portfolios could also be calculated using a different methodology such as minimax portfolio optimization, which uses a risk threshold (Polak et al., 2010), instead of absolute minimum risk or maximizing the Sharpe ratio; or semi-variance, mean absolute deviation and variance with skewness (Chang et al., 2009), instead of Markowitz' mean variance as used in this analysis. Extending the analysis with these methodologies to optimize land use allocations or minimize risk in combination with PES type incentives could be an area of future research.

The issue of land degradation and the current state of the pastures could strongly influence land use decisions. Although our analysis suggested that tree plantations were of little interest, Knoke et al. (2011) argue that reforestation could make economic sense when land degradation is considered. Moreover, the slope of the land could also affect land-use allocations.

We did not take into account all the variability in the production of milk and wood, which can be greatly affected by climatic conditions (Jakoby et al., 2014). Additionally, we only considered the price variations of agricultural output, and left the input prices constant. We thus did not consider all potential sources of uncertainty. However, according to Knoke et al. (2011), increased risks only strengthen the diversification effect. This would further increase the portfolio weight and area allocated to receive Socio Bosque payments (under our assumed uncertainty).

The reliability of our risk estimates depends heavily on the historical data provided by the government and collected by us. It is uncertain how these prices will evolve in the near future. Other methods for price and uncertainty estimation have been proposed and used, such as Monte Carlo simulations (see Hildebrandt and Knoke, 2011) for an overview of the different methodologies to estimate uncertainty). In addition we assumed a normal distribution of expected returns, an assumption that, especially in the case of forestry, may not hold true (Hildebrandt and Knoke, 2011).

We also did not consider the transaction costs involved in participating in one or more of the government programmes. The municipal watershed management and FORAGUA could play a fundamental role here, and act as intermediaries to reduce transaction costs (Hejnowicz et al., 2014). Whereas the three programmes analysed are all programmes of the national government, local government could operate as an intermediary, facilitating and supporting milk producers' applications. This could lower transaction costs for producers, and allow them more possibilities for considering land use combinations.

5.6. Conclusions

This chapter showed how portfolio theory can be used to analyse different conservation, restoration and reforestation programmes. It used a case study of different government programmes in Southern Ecuador to demonstrate that conserving forest and restoring pastures to their natural vegetation increased risk minimization and optimized land use allocations. It was more effective in achieving these aims than portfolios that lack incentive payments. However, the results also illustrated that, under the optimal land use allocation, restoration payments would only have a limited influence on highly productive farmers without forests. We also found evidence of potential leakage effects in land allocation portfolios with incentives among forest owners. Where there are no incentives available, tree plantations played a significant part in land use portfolios. However, significantly less land would be allocated to tree plantations when the incentive programmes were available.

Socio Bosque payments could have a positive impact on household income as they provide an extra source of income, yet we found that the effect of these payments on poverty reduction was limited. The size of the payments is not based on household income nor on the number of people in the household or community receiving the payment. So while Socio Bosque may be an important mechanism for conservation and, more recently, for restoration, there is a trade-off between the dual goals of protecting the largest area of land and reducing poverty. This is because the area that poor landholders can conserve and/or restore is too small to obtain a significantly higher income under the current payment levels. In addition, restoration payments generally would have a negative effect on expected household income, with the largest drop in income observed in those households that rely the most on income from milk production and did not own forest.

Chapter 6: Farmers' preferences for contracts to adopt silvopastoral systems in Southern Ecuador

Based on: Raes, L.; Speelman, S.; Aguirre, N. and Van Damme, P. Farmers' preferences for PES contracts to adopt silvopastoral systems in Southern Ecuador. (Under review).

Abstract

The supply of ecosystem services with public good characteristics from private lands could be improved through the use of positive incentives for the adoption of specific land use practices. This study investigates farmers' willingness to participate in contracts with incentive payments for the adoption of silvopastoral systems, with or without additional management requirements. The research was carried out in the buffer zone of Podocarpus National Park in Southern Ecuador, an area important for ecosystem service provision, but with most land dedicated to raising cattle. A choice experiment was conducted to measure farmers' interest in different types of contracts that varied with respect to the type of silvopastoral system, extra land use requirements, access of cattle to streams, level of payment and contract duration. A latent class model was used to determine how contract attributes, and farm and farmer characteristics influence choice for a certain type of contract. The majority of farmers in the area were interested in the proposed contracts, but they differed in their preferences regarding additional requirements and the importance of payment levels. Based on their preferences three classes of respondents were identified. The results suggest that farmland area, agricultural income, the share of this income in total income, and landowners' perception of environmental problems provide a partial explanation for the heterogeneity observed in the choices for specific contracts. Offering flexible contracts with varying additional requirements within the same scheme may contribute to enhance participation. However, enforcing contract compliance through the threat of withholding payments may be problematic, as for a group of farmers payment levels did not significantly influence contract choice. In addition, one small group of farmers preferred the business as usual scenario. Building of trust and combining PES with integrated conservation and development projects may be better to convince these farmers to adopt silvopastoral systems.

6.1. Introduction

Ecosystems provide a range of products and services such as timber, food, water purification and carbon sequestration on local, national and global scales (MA, 2005c). Land use changes such as deforestation to expand agricultural lands, often degrade soil, water and biological assets. This has affected the ability of ecosystems to supply ecosystem services beyond mere agricultural production (Foley et al., 2011; Power, 2010). A primary reason for this degradation is the failure to internalize environmental costs and benefits associated with land use decisions (Lant et al., 2008). Agricultural producers do not consider all ecosystem services they generate in their production decisions (Dagang and Nair, 2003; Matta et al., 2007).

Direct payment programmes that translate external, non-market ecosystem services into financial incentives, such as payments for ecosystem services (PES), are now commonly used policy instruments to encourage the continued provision of ecosystem services by private landowners (Layton and Siikamäki, 2009; Wünscher et al., 2008). PES schemes offer incentive payments to individuals and communities for the conservation or production of non-market ecosystem services or the implementation of land use practices that would secure those services (Ferraro and Kiss, 2002; Greiner and Stanley, 2013). As such PES should thus internalize the value of ecosystem services in land use decisions (Ojea and Martin-Ortega, 2015).

According to Wunder (2012, p. 2), “the arguably decisive PES criterion is whether providers receive conditional payments”. Payments should be conditional on whether ecosystem services are indeed provided (or land management systems implemented) or not (Martin-Ortega et al., 2013). If providers fail to deliver as agreed, then payments should be withdrawn or reduced (Milne and Niesten, 2009). It is this conditionality factor that is considered the fundamental difference between direct payments and integrated conservation and development projects (ICDPs) (Engel et al., 2008; Milne and Niesten, 2009; Wunder, 2012). ICDPs deliver monetary and other benefits upfront through the implementation of concrete interventions. Yet, these projects do not include longer term conditionality to assure continuation of the actions once a project has ended. However, ICDPs and PES could potentially be combined (Petheram and Campbell, 2010).

Making payments conditional to delivery of concrete terms is important to assure PES effectiveness. However, if payments are perceived to under-compensate landowners, PES may not conserve or improve ecosystem service provisioning due to low participation or high rates of non-compliance. On the other hand, if payments overcompensate landowners, conservation benefits may not be maximized from the given budget (Jack et al., 2009). Hence, the level of payments is also crucial for PES implementation. The opportunity costs resulting from the loss in revenue caused by the provision of additional or new ecosystem services are often proposed as a way to estimate the adequate payment level (Ojea and Martin-Ortega, 2015; Robert and Stenger, 2013).

Apart from establishing payment levels, it is imperative to understand preferences of landowners for alternative configurations of a PES programme when adapting these incentives to particular circumstances (Layton and Siikamäki, 2009; Petheram and Campbell, 2010). The range of heterogeneity within these preferences and whether such heterogeneity is associated with particular farm and farmer characteristics are also important issues (Beharry-Borg et al., 2013).

Although conservation of natural forests is generally considered to guarantee the highest levels of ecosystem services (e.g. Pan et al., 2011; Paul et al., 2008), other land management approaches, such as reforestation or the implementation of agroforestry systems, may also provide a wide range of services (e.g. Fehse et al., 2002; Haug et al., 2010; Philpott et al., 2008). "Agroforestry is a collective name for land use systems and practices where woody perennials are deliberately integrated with crops and/or animals on the same land management unit" (ICRAF, 1993) . When trees and livestock production are combined in an agroforestry system, often the term silvopastoral systems (SPS) is used (Calle et al., 2009).

SPS generate a variety of ecosystem services such as protection of watersheds and biodiversity (Alavalapati et al., 2004; Harvey and Haber, 1998). In addition to their capacity to provide non-market ecosystem services, SPS may also increase livestock production and as such rural household income (Ibrahim et al., 2006; Murgueitio et al., 2006). However, in spite of both positive environmental and economic outcomes, the wider adoption of SPS has been hindered by a lack of capital for its implementation (Ibrahim et al., 2006; Pattanayak et al., 2003). According to Pagiola et al. (2007), a small payment can make SPS more profitable than many current agricultural practices. The World Bank's Regional Integrated Silvopastoral Ecosystem Management Project used five-year PES contracts to encourage the adoption of SPS in degraded pastures in Colombia, Costa Rica and Nicaragua (Pagiola et al., 2010). Other examples of agroforestry practices as part of a PES scheme are Costa Rica's national PES programme and the Mexican carbon PES scheme Scolel Té (FONAFIFO, 2015; Soto-Pinto et al., 2010).

In this study, we use a choice experiment (CE) with farmers in the buffer zone of Podocarpus National Park in Southern Ecuador to analyse and quantify preferences for different components of PES contracts that try to incentivize farmers to adopt SPS. This study aims to quantify (1) farmers' interest in contracts for the implementation of SPS that come with or without additional management requirements; (2) whether farmers' preferences are affected by the payment level and/or contract duration; and (3) the extent of heterogeneity in preferences among farmers and investigate whether this heterogeneity can be attributed to particular farmer and/or farm characteristics.

CEs are a stated preference valuation technique where subjects are asked to choose among different hypothetical goods or outcomes. They allow for the analysis of preferences for policies containing several components. Moreover, because of their hypothetical nature, CEs can be used to analyse preferences for polices that have not yet been implemented (Rabotyagov and Lin, 2013). The application of CE to PES is limited, but a growing number of studies has been done to better inform PES design (Kaczan et al., 2013).

Horne (2006) examined factors that influence choices of private forest owners in Finland for conservation contracts. The results of the CE showed that payments had to be increased considerably when programme characteristics became stricter and contracts longer. Also Ruto and Garrod (2009) showed that payments had to increase for CE participants to choose longer contracts, or contracts with stronger conditionality or higher administrative costs. According to Matta et al. (2009), participation of private forest owners in a conservation programme is lowered as more restrictions on forest management are imposed. Broch and Vedel (2012) used a CE to analyse farmers' preferences for afforestation contracts in Denmark. Contract flexibility and protection of biodiversity and groundwater decreased farmers' need of compensation to participate, while monitoring increased payments. Rabotyagov and Lin (2013) used a CE to examine landowner preferences for the attributes of ongoing forest conservation contracts. In their study shorter contracts and partial enrolment of forests are preferred.

Furthermore, heterogeneity among respondents and respondent characteristics play an important role in preferences. Matta et al. (2009) found that forest owner characteristics significantly influence participation. Broch and Vedel (2012) and Rabotyagov and Lin (2013) also observed high heterogeneity in farmers' preferences. Siebert et al. (2010) found that one group of German farmers interested in the concept of conservation set-aside would accept a small to moderate loss in income to withdraw land from agricultural production. Putten et al. (2011) had similar findings for a conservation incentive programme in Australia.

Several recent studies used CE to evaluate participation in conservation incentive programmes in developing countries. Tesfaye and Brouwer (2012) analysed the interest in contracts that stimulate soil conservation measures; contracts offering additional credit, land use security and extension services could be used to increase participation. Balderas Torres et al. (2013) examined preferences for PES in Mexico. They showed that local development interventions included in the PES design can result in a lower importance of monetary payments. Kaczan et al. (2013) assessed farmers' preferences for participation in a PES programme with different types of payments in Tanzania. They found that an upfront, in-kind payment of fertilizer significantly motivated participation, whereas payments into the village fund did not. Marenya et al. (2014) examined smallholders' preferences for incentives to adopt conservation practices in Malawi. Most farmers preferred cash payments to insurance contracts, even when the insurance contracts offered higher expected returns. Nordén (2014) used a CE in Costa Rica, and found that landowners with a higher dependence on on-farm income had stronger preferences for higher payments.

In this chapter, the focus is on preferences for PES contracts to adopt SPS. For which, as far as we know, no CE studies have been carried out. While different monetary payment levels are included in the

CE, we add extra management requirements to some of the contract options to analyse landholders' preferences, instead of focusing on different payment types. As such, we can find out what payment levels would incentivize farmers to adopt SPS, but also whether farmers choose contracts with extra requirements. Furthermore, such preferences may be highly heterogeneous.

6.2. Methodology

6.2.1. Research Area

This research was carried out in the buffer zone of the Podocarpus National Park close to the city of Loja in Southern Ecuador (Figure 6.1). Excluding the national park itself (1,472 ha within the research area) and the El Carmen and Pizarros watersheds (912 and 734 ha respectively), the study area has a total surface of 5,475 ha. This area is important for the supply of drinking water to the city of Loja. The municipality of Loja is a founding member of the Regional Water Fund (FORAGUA). This fund uses an environmental charge included in the water bill, to finance the protection and improvement of watershed services. The watersheds of El Carmen and Pizarros have been protected through land purchases and the declaration of municipal protected areas. However, in the other watersheds of the research area and the areas surrounding them, people continue to raise livestock, mainly dairy cows for milk production. SPS, forest conservation, pasture abandonment for restoration and tree plantations can decrease the negative impact of cattle raising on water quality and improve the soil's water retention capacity (Iñiguez–Armijos et al., 2014; Knoke et al., 2014).

In Ecuador, several programmes are currently implemented that try to incentivize the adoption of the above-mentioned land management systems. The largest of these programmes is the national, government-funded Socio Bosque Programme. Through this programme, private and communal landowners receive incentives (or rewards) for the conservation of forest and/or páramo (Andean grasslands) (de Koning et al., 2011 and Annex 1 and 6 provide more detail on Socio Bosque).

The conservation component of Socio Bosque started in 2008. In 2014 the programme additionally started the implementation of a new module, focusing on restoration. In the programme restoration can focus on the recovery of natural ecosystems through reforestation or natural regeneration. In addition, the aim can be enrichment of lands with native species (Ministry of the Environment, Ecuador, 2014b). The latter includes planting hedges or trees in pastures, both of which can be part of SPS. The programme refunds establishment costs during the first three years, after which the participants can receive the Socio Bosque incentives.

Furthermore, for the establishment of timber plantations, the Ecuadorian Ministry of Agriculture provides incentives through the *Incentivo Económico para la Forestación y Reforestación* programme (Economic Incentives for Afforestation and Reforestation) (Ministry of Agriculture, Ecuador, 2013a, 2013b). This programme refunds part to all of the investment costs of tree plantations with productive purposes.

The main land management system considered in our analysis is SPS. However, farmers could additionally prefer to implement conservation measures for their forests, restore degraded pastures to their natural state, or establish timber plantations as an additional (long-term) income source.

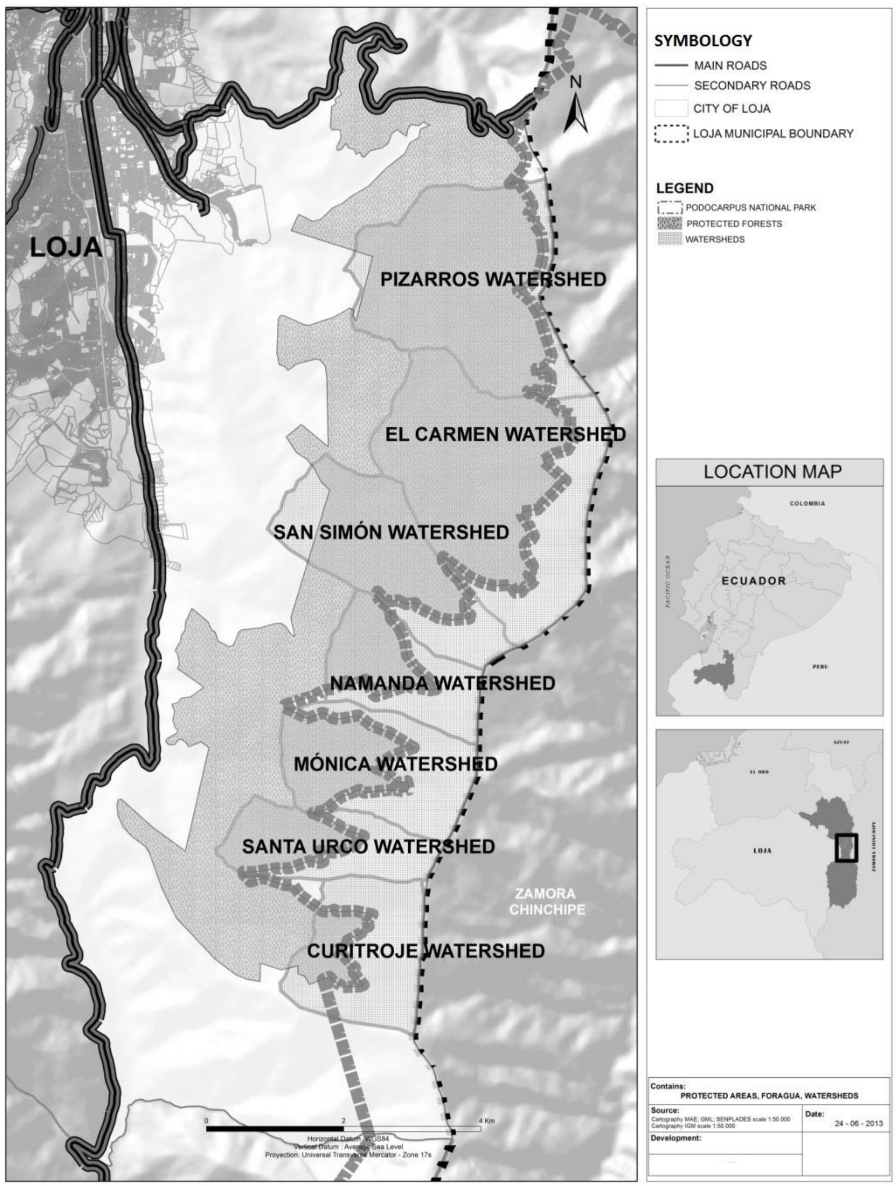


Figure 6.1: Research Area (Source: Cevallos and Raes with data from from Ministry of the Environment of Ecuador, 2013; GLM, 2013; SENPLADES, 2013; IGM, 2013)

6.2.2. CE model design

The basis of CEs is that "any good can be described in terms of its attributes or characteristics and the levels that these take" (Bateman et al., 2002). CEs combine consumer choice theory (Lancaster, 1966), with random utility theory (Ben-Akiva and Lerman, 1985). The first theory postulates that utility from a good comes from the value of the different attributes of that good. Whereas, according to the second, the observation of utility can only be made imperfectly, so the utility of a good consists of deterministic and stochastic elements. CEs can be applied to situations wherein an individual selects one alternative from a set of different options. Each alternative is represented by a bundle of attributes, a series of characteristics. The set of alternatives contained in each question is known as a 'choice set' (Aizaki, 2012). In a CE, respondents are asked to select the most-preferred alternative from a choice set. In this study, the choice set consists of two different contract specifications and a 'business as usual (BAU)' situation. The BAU option is included to ensure that respondents are not forced to choose an unsatisfactory option (Jaeck and Lifran, 2014)

Attribute specification

The good to be analysed in this study is a contract for the adoption of SPS. In this CE, we aimed to focus on these landscape elements that are important for the generation of watershed services and biodiversity conservation, and that can be provided by local landowners, specifically milk producers.

A preliminary list of attributes that described land management practices was derived from a literature review and from semi-structured interviews with watershed managers and researchers, and was additionally based on existing programmes implemented in Ecuador (see 6.2.1).

The proposed contracts consist of two parts: (1) the implementation of an SPS system with or without additional land management requirements, and with or without the obligation to implement additional protection measures for streams on the farmer's property; and (2) payments conditional on the continuation of the contractually stipulated management measures for the duration of the contracts. Whereas the government programmes currently state that they will reimburse establishment costs, in the contracts we presented in the CE, we specified that inputs such as seedlings would be provided to the participants as part of the contract for the implementation of SPS. The provision of seedlings was also included in the reforestation option of the additional land management attribute. Further inputs provided were barbed wire and poles to fence areas for SPS, restoration (natural recovery), and to restrict access of cattle to streams. The supply of these materials can be considered the ICDP part of the contract, as the monetary payments were additional to the supply of inputs. Early withdrawal from a contract, i.e. breaking contract conditions, would result in the termination of further payments[36].

The selected contract attributes and their levels are reported in Table 6.1 and can be presented as:

- *SPS*: (1) trees dispersed within pastures (silvo-pastures) and (2) living fences (Murgueitio et al., 2006). The species to be used are fruit trees and the native Andean alder (*Alnus acuminata*) a fast growing species which can improve water quality and water retention capacity (Knoke et al., 2014), and is well-suited for agroforestry (Dunn et al., 1990). It was specified in the questionnaire that a minimum of 150 trees/ha had to be planted, based on the minimum necessary to receive credits for the implementation of agroforestry systems provided by the National Financial Corporation of Ecuador (CFN, 2015).
- *Additional land management requirements:* (1) conservation and/or restoration of a minimum of 15% of the land area[37], (2) the establishment of Andean alder plantations, or (3) no additional land management system.
- *Access of cows to streams*: protection of water sources from cattle can improve water quality (Chará and Murgueitio, 2005). Farmers can choose (1) to restrict access by fencing the land, or (2) not restricting access.
- *Payments*: the payment vehicle was a per hectare annual cash payment ranging from US$ 30 to US$ 70 with three levels. The current Socio Bosque payments for conservation are 60 US$/ha per year for farmers with less than 20 hectares of overall property, and start at US$/ha 30 per year for landowners who own more than 20 hectares of property (Ministry of the Environment, Ecuador, 2012a).
- *Contract duration:* contract duration ranged from five to 20 years. The latter coincides with the production cycle of Andean alder and the duration of Socio Bosque contracts.

[36] The Socio Bosque programme is stricter by demanding pay back of a certain amount of money depending on the number of years of participation previous to breaking the contract rules (Ministry of the Environment, Ecuador, 2012d).

[37] The questionnaire specified that in first instance land with stronger slopes and riparian areas should be incorporated. These lands are most vulnerable to the impacts of cattle and hence more gains are to be foreseen.

Table 6.1: Attributes and attribute levels

Attributes	Dummy coding	Levels			
		Opt-Out	Level 1	Level 2	Level 3
Silvopastoral system	Yes	None	Silvo-pastures	Hedges	/
Additional land management requirement	Yes	Current state	Conservation and/or restoration	Andean alder plantation	No extra requirement
Access cows to streams	Yes	Current situation	No access	Access allowed	/
Payment (US$/ha/year)	No	0	30	50	70
Contract duration (years)	No	0	5	10	20

Survey design and data collection

Having identified the relevant attributes and levels to be used in the CE, an experimental design was created from which the choice sets were constructed.

A fractional factorial design produced six blocks, each with eight choice sets. The questionnaire consisted of three parts. The first questions were on socio-economic characteristics of participating households and identified farm characteristics such as crops grown, milk production, and area of crop, pasture and forest land. The second part collected information on farmers' assessment of environmental problems and knowledge of existing governmental incentive programmes. Finally, an information section explaining the upcoming CE procedure was followed by the CE exercise itself. The CE included visual aids for each of the land management attribute options. Questionnaires were filled in during face-to-face interviews.

The questionnaire was tested in the research area with ten households in February 2013. Some clarifications had to be added to the CE, but no major changes were made. The CE survey with 120 milk producers was carried out from March to July 2013 by students from the National University of Loja. We lacked a list of all households owning land within the research area. These households were only listed as residents of Loja city, not specifically of the research area. Thus, random sampling was not possible. Instead, snowball sampling was used to find respondents (Biernacki and Waldorf, 1981; Lavrakas, 2008). The watershed managers of Loja's municipal water company introduced us to a few milk producers raising cattle in or around each of the watersheds. These milk producers in turn introduced us to new respondents. Respondents were randomly assigned to one of the six survey versions. Participation in the CE was totally voluntary. Verbal prior consent was obtained after respondents were informed of the purpose of the survey, and were assured individual anonymity.

Income calculation

The survey collected information on farm production activities and off-farm income. Non-milk farm income (crops, pigs and chickens) was calculated using 2011 prices. Subsistence income was calculated with farm gate prices. The official minimum milk price in 2011 of US$/l 0.40, and production cost data from Knoke et al. (2014) were used to calculate milk income. Off-farm income calculations were based on national data of income per job type in Ecuador (INEC, 2012c).

6.2.3. CE model specification

Conditional Logit Model

In a CE, it is assumed that a farmer will choose a contract for which the net utility is higher than either the option of no contract or any of the other choices. The probability of a farmer making a particular choice is assumed to increase as the utility of that choice increases (Louviere et al., 2000). The conditional logit model (CLM) is the most commonly used discrete choice model for the analysis of CE results. The basic model is the standard random utility function (McFadden, 1974).

In each choice occasion, a respondent faces a choice between j=2 alternatives and the BAU option. The vector of observed attributes for the option j in a choice occasion faced by respondent i is labelled as X_{ij}. As it is difficult to completely describe a choice only in terms of its characteristics, the random-utility model accounts for unobservable elements by adding an error term (Bateman et al., 2002). The utility that an individual derives from choosing an alternative in a choice occasion is given by:

$$U_{ij} = C_{BAU}BAU + \beta_i X_{ij} + \varepsilon_{ij} \qquad (1)$$

Where C_{BAU} is a constant term that depict the impact that unobserved effects exert over the selection of the BAU scenario. It takes the value of 0 if one of the contract options is selected and 1 if the BAU scenario is selected. β_i is the coefficient vector associated with the attributes and is deterministic and observable. The second term ε_{ij} is the unobservable error term and is assumed to be independent and identically distributed, following a Gumbel distribution (Louviere et al., 2000; Swait, 1994). The analysis becomes one of probabilistic choice, as predictions cannot be made with certainty since there is an error term in the utility function (Louviere et al., 2000). An individual i, will choose an alternative from a specific choice set, given that the utility of this alternative is greater than the utility of any other alternative in the choice set. Conditional on β_i, he probability (P) that farmer i chooses alternative j in preference to any other alternative k can be expressed as:

$$P_{ij} = \frac{exp(\beta_i X_j)}{\sum_j^J exp(\beta_i X_j)} \qquad (2)$$

Although the relative simplicity of the CLM is an advantage, it has some important limitations. This model has only choice attributes included as explanatory variables. The conditional logit assumes that choices are consistent with the 'independence of irrelevant alternatives' (IIA) property. This property assumes that the relative probability of an option being chosen is unaffected by the introduction or removal of alternatives (Luce, 1959). This model furthermore assumes that preferences are homogeneous across respondents (Hausman and McFadden, 1984). Preferences however, may be heterogeneous and accounting for the preference of heterogeneity enables computations of unbiased estimates of individual preferences. If a violation of the IIA assumption is found, then the assumption can be relaxed by employing latent class model (LCM) (Boxall and Adamowicz, 2002).

Latent Class Model

Survey data were analysed with a Latent Class Model (LCM) because preferences are expected to be heterogeneous. In LCM, heterogeneity follows a discrete distribution, a specification based on the concept of endogenous (or latent) preference segmentation. The latter means that each class c=(1,2,…C) is homogeneous with respect to attribute preferences, but there is heterogeneity across classes (Boxall and Adamowicz, 2002). Except for the number of classes, LCM does not require any assumption on the distribution of the parameters. The detection of respondent classes can be of significant practical use in policy design compared to the identification of a continuous distribution of preferences like in random parameter models (Kaczan et al., 2013).

The overall utility that individual i derives from alternative j in a LCM includes an additional term ϕ that seeks to capture random preferences among individuals. Preferences are assumed to be constant for a given individual across the choices made, but not across the whole sample. The utility function then becomes (Swait, 1994):

$$U_{ij} = C_{BAU}BAU + \beta_i X_{ij} + \varepsilon_{ij} = C_{BAU}BAU + b_i X_{ij} + \phi_i X_{ij} + \varepsilon_{ij} \qquad (3)$$

Where β_i has two terms: b represents the associate parameters of the contract attributes, and ϕ is a vector of deviation parameters representing individuals' characteristics (Swait, 1994). Louviere et al. (2000) and Swait (1994) provide more details on the model specifications.

Now, again conditional on β_i, the probability (P_{ij}) that farmer i chooses alternative j over any other alternative k can be expressed as (Boxall and Adamowicz, 2002):

$$P_{ij} = \sum_c^C \left(P'_{i,c} \cdot \frac{exp(\beta_{ic} X_j)}{\sum_j^J exp(\beta_{ic} X_j)} \right) \qquad (4)$$

Where $P'_{i,c}$ is the probability that an individual i belongs to a latent class c; and β_{ic} are the estimates of the parameters for attribute preference within each class.

These parameter estimates and the specific probabilities of class membership are adjusted iteratively by maximum likelihood methods to maximize the explanatory power of the LCM for a given number of discrete classes C (Boxall and Adamowicz, 2002).

The models were estimated with LIMDEP Version 9 (Greene, 2007). Farm and farmer characteristics were introduced as interaction terms with the choice attributes. Through trial and error, the following covariates were introduced: (1) total forest area; (2) percentage income from milk; and (3) respondent's perception of the quality of their drinking water.

Statistical tests

To analyse differences between the classes of farmers we used several statistical tests. For normality we used the Shapiro-Wilk test of normality, while we also applied the central limit theorem (Rice, 2007). For differences in variances we used F tests for two samples and Bartlett's tests for more than two samples. Depending on these analyses for comparing two classes we used either the Mann-Whitney or the two sample t-tests. For more classes ANOVA and Wilcoxon Rank Sum tests were used. Analyses were carried out with R software (R Core Team, 2014).

6.3. Results

6.3.1. Characterization of respondents

Table 6.2 presents the descriptive statistics of characteristics of our sample of 120 respondents. Male respondents outnumbered females. Respondents were on average 50 years old, and typically completed primary school. Eight respondents earned a university degree, while four received no formal education.

The farmers of our sample managed around 22% of the land area within the research area, excluding the protected areas. The average farm size was 10 ha, although with a high standard deviation (18) within the sample. Figure 6.2 shows the distribution of all sample farms according to size. For comparison we added the distribution of all farms in Loja Province to[38]. The latter were obtained from the Ministry of Agriculture of Ecuador (2012).

Commercially some farmers produced pigs, whereas most crop production was for household consumption. The most important agricultural activity for the majority of farmers was milk production, which could be commercial or for subsistence[39]. The word subsistence may not properly describe all non-commercial milk farms, some farms could rather be considered as 'hobby' farms since to these farm income was not an important source of income (or food). The utility they derived was difficult to assess. Hence, for some subsistence (non-commercial) farms profit could even be negative. A large part of the households of the sample did not depend on milk production for their livelihoods, but rather on off-farm income. 28% of households lived below the poverty line, similar to the Ecuadorian average of 29% of the total population (INEC, 2011b). Figure 6.3 shows how our sample of households was situated within the Ecuadorian income deciles (INEC, 2012c).

[38] Farms smaller than 1 hectare are normally not dedicated to cattle and milk production.
[39] 25 of the 120 milk farms of the sample were non-commercial.

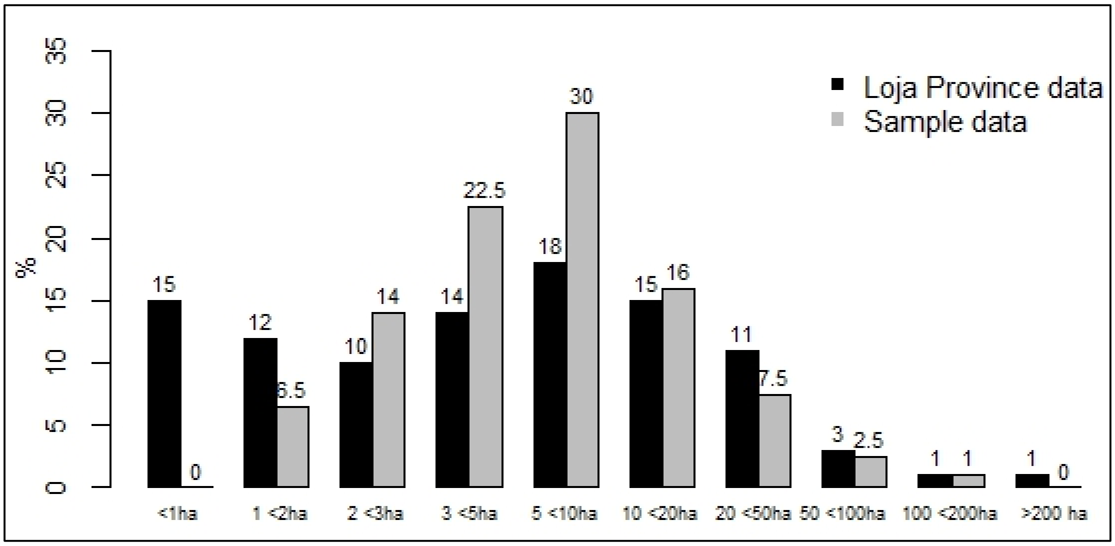


Figure 6.2: Percentage of farms according to farm size for Loja Province and sample, respectively

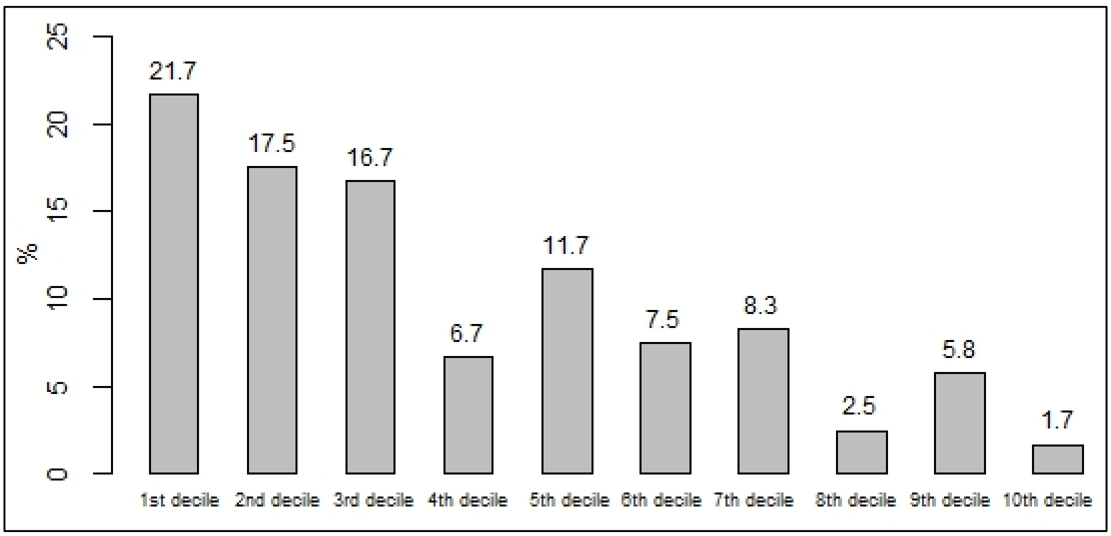


Figure 6.3: Distribution of sampled households according to Ecuadorian per capita income deciles

Table 6.2: Farm and farmer characteristics

Characteristic	Median	Mean	Standard deviation
Age of respondent (years)	47	50	16
Number of household members	3	4	1
Total area (ha)	5	10	18
Pasture area (ha)	4	6	8
Crop area (ha)	0.1	0.5	1
Forest area (ha)	0	3	11
Off-farm income (US$/year)	3,500	4,310	4,030
Crop income (US$/year)	90	330	500
Non-cattle livestock income (US$/year)	0	550	2,040
Non-milk farm income (US$/year)	200	880	2,280
Heads of cattle	4	7	10
Average milk production per cow (l/day)	6	6	2
Total household milk income (US$/year)	310	510	690
Milk income per ha (US$/year)	80	90	100
Total household income (US$/year)	4,350	5,700	4,490
Income per household member (US$/year)	1,260	1,640	1,190
Percentage income from farm (%)	18	28	30
Percentage income from milk (%)	8	14	21

6.3.2. Outcomes of the choice modelling

Model selection

Table 6.3 provides a summary of the different test statistics (log likelihood, Bayesian Information Criterion - BIC, and Akaike Information Criterion - AIC) used for model selection (Hensher et al., 2005). A model is preferred if it has a lower AIC or BIC (Boxall and Adamowicz, 2002). The fit of the CLM was low and therefore it is further discarded from our analysis. This low fit could be due to the considerable heterogeneity in farmers' preferences. This was confirmed by the better test statistics for the LCMs.

Following Boxall and Adamowicz (2002) and Louviere et al. (2000), we defined the number of classes based on the BIC, AIC and the credibility of results given the size of the classes. The four class model showed the highest BIC, the two class model the lowest. However, the three class model had an AIC lower than the two class model. In addition, the two class model had a segmentation that was less representative for individual farmers' preferences than the other LCMs, with most respondents in one class. Whereas the four class model had very small numbers of individuals in three out of the four classes. Given its good fit and meaningful segmentation, the three class LCM was chosen as the most appropriate model.

Table 6.3: Criteria for model selection

Model	CLM	LCM 2 classes	LCM 3 classes	LCM 4 classes
Log likelihood	-769	-700	-678	-653
Number of parameters	7	18	29	40
AIC	2.154	1.997	1.965	1.924
BIC	2.199	2.111	2.150	2.179
Average class probability (%)	/	97	91	94

Results of contract preferences

The LCM demonstrated an uneven distribution of farmers' preferences across the three classes (Table 6.4). Class 1 was strongly associated with 53% of the sampled farmers, class 2 with 17% of the sample and class 3 with 30%. The LCM suggested that different preferences were present with regard to the BAU option. Class 2 showed a strong positive preference for the BAU, while class 1 had a negative preference.

Class 1 contained the largest group of respondents. In addition to having a negative and significant BAU coefficient, indicating a preference for the proposed contracts, this class displayed a significant aversion towards the conservation/restoration attribute versus no additional land management requirements, and for putting fences to protect streams from cattle. However, higher payments increased the likelihood of a contract being chosen by this class. Class 2 had a positive BAU coefficient, indicating that its members were on the whole reluctant to opt for SPS contracts. When contracts were chosen, the additional requirement of restricting access for cattle was the only significant attribute of farmers' contract preferences. Finally, respondents in class 3 displayed a significant preference for the conservation/restoration attribute over no additional land management requirement, and the implementation of fences for access restriction for cattle over a system where fences was not compulsory. This group stood out as the only group with a non-significant parameter estimate for the BAU.

The type of SPS did not appear to be a significant influence in contract choice for any of the classes. The same held true for contract duration. Furthermore, also the establishment of plantations versus no additional land management requirements was not a significant attribute in guiding contract preference for any of the classes.

Table 6.4: Output LCM

Class (probability)	Attributes	Coefficient	SE (coef.)	P-value	
Class 1	Silvo-pastures	-0.1466	0.1742	0.3999	
(0.53)	Conservation/Restoration	-0.5963	0.2248	0.0080	***
	Andean alder plantations	-0.2309	0.2200	0.2941	
	Cows no access to streams	-0.3855	0.2195	0.0790	*
	Payment	0.0088	0.0051	0.0853	*
	Contract duration	0.0058	0.1340	0.6663	
	BAU	-1.1308	0.3706	0.0023	***
ϕ (1)	Forest area	-0.1749	0.1004	0.0815	*
	Percentage income from milk	-1.8572	1.3888	0.1811	
	Water quality perception	-1.1668	0.6825	0.0873	*
Class 2	Silvo-pastures	-0.2560	0.4723	0.5878	
(0.17)	Conservation/Restoration	0.4342	0.7256	0.5495	
	Andean alder plantations	0.0827	0.8052	0.9182	
	Cows no access to streams	1.5460	0.8076	0.0556	*
	Payment	-0.0115	0.0148	0.4360	
	Contract duration	0.0072	0.0519	0.8899	
	BAU	2.4536	1.0710	0.0220	**
ϕ (2)	Forest area	-0.1353	0.1256	0.2812	
	Percentage income from milk	-5.8487	3.0393	0.0543	*
	Water quality perception	0.8771	1.2059	0.4670	
Class 3	Silvo-pastures	-0.0746	0.3328	0.8225	
(0.30)	Conservation/Restoration	1.1881	0.5284	0.0245	**
	Andean alder plantations	0.5891	0.4898	0.2291	
	Cows no access to streams	2.1449	0.5464	0.0001	***
	Payment	-0.0051	0.0092	0.5769	
	Contract duration	-0.0220	0.0239	0.3580	
	BAU	-0.5755	0.7094	0.4173	

* Significant at 0.1
**Significant at 0.05
***Significant at 0.01

Farm and farmer characteristics across classes

Classes did not only differ in terms of contract preferences, but also in terms of farm and farmer characteristics of their members. LCM output showed that forest area and perception of water quality differed significantly between class 1 and 2. The model additionally revealed that farmers in class 2 received a significantly different percentage of their income from milk production than farmers of class 3 (Table 6.4).

Post hoc analysis of farm, respondent and household characteristics of the different classes showed that, on average, farmers in class 1 had significantly more pasture land area than farmers in class 2, but significantly less land than those in class 3 (Table 6.5). Class 1 was also characterized by receiving a significantly higher income share from farming than class 2, but lower than class 3. Farmers in Class 2 were the least dependent on farm income. The income share from both milk and crops was significantly lower in this class compared to that of the other two classes. Class 3 contained households that were more farm-oriented. They obtained a significantly larger share of their income from farming, specifically from milk production, and had on average more land, including larger pasture, crop and forest areas, and more cattle than the other two classes. On the other hand, this class received significantly less income from off-farm sources.

Commercial or non-commercial farming nor the slope of the land provided additional insights into explaining differences between classes (Table 6.6). Also in terms of the educational level of the respondents, no significant differences between classes were found.

Table 6.5: Average farmer and farm characteristics of the different classes

Characteristic	Class 1	Class 2	Class 3	p-value	
Age of respondent (years)	49	50	53	0.2020[a]	
Number of household members	4	4	3	0.2562[a]	
Total area (ha)	6.5	7	17	0.0480[b]	**
Pasture area (ha)	5	5	9	0.0313[b]	**
Crop area (ha)	0.5	0.3	0.6	0.1309[b]	
Forest area (ha)	1	1	7	0.1711[b]	
Off-farm income (US$/year)	4,590	4,800	3,560	0.1333[b]	
Crop Income (US$/year)	320	200	420	0.0647[b]	*
Animal Income (US$/year)	440	90	980	0.6129[b]	
Non-milk farm income (US$/year)	760	300	1,400	0.2269[b]	
Heads of cattle	5	5	13	0.0004[b]	***
Total milk income (US$/year)	420	310	780	0.0350[c]	**
Milk income per ha (US$/year)	90	80	100	0.7230[a]	
Total household income (US$/year)	5,770	5,400	5,730	0.9500[a]	
Income per household member (US$/year)	1,620	1,410	1,790	0.4108[c]	
Percentage income from farm	25	12	41	0.0000[c]	***
Percentage income from milk	12	5	23	0.0671[b]	*

[a] One-way ANOVA with Tukey Honest Significant Differences Test

[b] Kruskal-Wallis rank sum test and Wilcoxon rank sum tests with continuity correction

[c] One-way analysis of means and (Welch) Two Sample t-tests

* Significant at 0.1

** Significant at 0.05

*** Significant at 0.01

Table 6.6: Differences between classes in level of education, type of milk production activity and slope of the land

	Class 1 (N=64)	Class 2 (N=19)	Class 3 (N=37)	p-value[a]
	Percentage of class member			
Level of education				
No formal education	3	0	5	0.5090
Incomplete primary education	25	42	32	
Complete primary education	31	21	27	
Incomplete secondary education	20	21	14	
Complete secondary education	16	16	8	
Higher education (university)	5	0	14	
Type of milk production				
Commercial	78	68	78	0.6509
Non-commercial	22	32	22	
Slope				
Regular/almost flat	14	0	8	0.4273
Irregular with weak slope	45	42	51.5	
Irregular with strong slope	25	37	32.5	
Irregular with weak & strong slope	16	21	8	

[a] Pearson's χ^2 test

Respondents' assessment of environmental problems

Finally, there were a few significant differences between classes as to their respective assessments of several environmental problems (Table 6.7). Generally, members of class 1 did not consider the quality of their drinking water as good. This was opposed to the other two classes, where the majority considered the drinking water to be of good quality. Furthermore, in comparison to the other two classes, significantly more class 1 members did not consider that they had erosion problems on their land. Significantly more respondents in class 1 considered the stream water to be polluted compared to class 3. Moreover, the perception on the negative impact of cattle on water quality differed significantly between class 2 and 3, with more agreement on this negative impact in class 3. Also more members of the class 3 considered that cattle caused erosion compared to the other two classes.

When considering awareness of existing environmental programmes, a majority of members of class 3 did know the Socio Bosque Programme, compared to only a minority in the other two classes. Finally, awareness of the PROFERESTAL[40] programme in class 1 was significantly lower than in the other two classes.

[40] Name of the previous programme of timber plantation subsidies. It was changed to the new programme during the CE.

Table 6.7: Farmers' assessment of environmental problems and awareness of alternative government programmes (in % of class members)

	Class 1 (N= 64)			Class 2 (N= 19)			Class 3 (N= 37)			p-value[a]
	Yes	No	NA	Yes	No	NA	Yes	No	NA	
Is the quality of drinking water good here?	30	70	0	74	26	0	59	41	0	0.0004***
Is the stream water polluted?	70	30	0	53	47	0	54	41	5	0.1144
Erosion problems on your land?	37	52	11	21	37	42	38	38	24	0.0374**
Erosion problems on neighbours' lands?	33	42	25	32	26	42	46	22	32	0.1914
Does cattle husbandry have a negative impact on water quality?	75	17	8	53	37	10	76	11	13	0.1556
Does cattle husbandry cause erosion?	50	47	3	37	58	5	65	24	11	0.0686*
Do you know the Socio Bosque Programme	22	78	0	26	74	0	57	43	0	0.0013***
Do you know the PROFORESTAL Programme	5	95	0	37	63	0	24	76	0	0.0008***

[a] Pearson's $\chi2$ test of differences among the 3 classes
* Significant at 0.1
** Significant at 0.05
*** Significant at 0.01

6.4. Discussion

This study showed that the preference for SPS contracts and the impact of additional environmental requirements varied among farmers, with a relatively large group of farmers preferring SPS contracts that guarantee higher payments. This observed heterogeneity in contract preferences is similar to what was found by Matta et al. (2009), Broch and Vedel (2012), Beharry-Borg et al. (2013) and Rabotyagov and Lin (2013).

Class 1 consisted of households that, in terms of farm and household income characteristics analysed, were situated in between the other two classes. Members of this first class showed a clear aversion for the BAU. This group additionally disliked the option of forest conservation or taking land out of production for restoration, or adding water access requirements to the contract. They only showed a significant positive preference for the payment attribute of the proposed contracts. Members of this class, on average, owned no or only a small forest patch; so conservation may be of little interest. Whereas the lack of interest in abandonment of pastures for restoration could partially be explained by the fact that the majority of the members in class 1 did not consider erosion, and thus one aspect of pasture degradation, to be a problem on their land. In addition, class 1 members did not show any significant preference for one of the two productive land management attributes (timber plantations or no additional requirement beyond SPS). Rosa et al. (2004) note that smaller landowners are more likely to participate in PES schemes that incorporate land management systems with a productive orientation, such as agroforestry (SPS).

That members of class 1 preferred contracts with higher payments, and without additional restrictions is not surprising. In a similar line, Horne (2006) and Ruto and Garrod (2009) found that payments increased with more conditionality, whereas Matta et al. (2009) found that more restrictions lowered participation in conservation programme. Bremer et al. (2014), in their analysis on participation in the Socio Bosque programme for páramo conservation, found that for most participants payments were an important motivation to participate. However, they add that the maintenance or improvement of water supply and biodiversity were also important factors in explaining programme participation.

Most class 1 members considered water and stream quality to be low, and were aware that cattle had negative effects, especially on water quality. However, a majority did not consider erosion to be a problem on their land. Vignola et al. (2010) in their analysis on the adoption of soil conservation in a Costa Rican watershed, found that although farmers can be aware of the impact of agriculture on soil stability, they may underestimate the impacts of their own activities on erosion. Furthermore, members of class 1 could have thought that there was not much they could do to improve water quality and decrease stream pollution.

Class 2 was the least interested in SPS contracts. Farmers in class 2 depended less on income from farming than farmers in the other classes, and would thus experience less of a potential impact on household income of participating in the PES contracts. Their preferences may thus depend less on how well payments would match their opportunity costs, and could be based more on their direct interest or trust in the programme. Mistrust in the government (or other PES implementers) and a fear that PES may lead to expropriation of land are essential factors in PES participation (e.g. Bremer et al., 2014; Southgate and Wunder, 2009; Zanella et al., 2014). The implementation of in-kind payments and extension services in the Bolivian PES scheme in Los Negros, was based on the observation that for participants "paying cash 'smells' more like giving up property rights—whether that fear is rational or not" (Asquith et al., 2008, p. 679).

Class 2 preferred the BAU, but additionally showed willingness to improve environmental conditions by restricting access of cows to streams. Thus, an ICDP where farmers are provided with the necessary materials to decrease the negative impact of cattle on water quality could have been more interesting for this class. The preference for the BAU could also be related to the fact that the majority of the members in this group did not consider that cattle causes erosion, whereas a large group within this class did not know whether or not they had erosion problems on their land. Environmental education as part of an ICDP may better solve these issues than PES.

Class 3 only showed a significant preference for the additional requirement of forest conservation/restoration (versus no additional land management system), and access restriction to streams. Class 3 had more forest, which could explain why in this class there was a strong interest in conservation. However, this does not explain the preference for stream access restriction. Just as class 1, class 3 members agreed that cattle husbandry had a negative impact on water quality, and more members than in the other two classes considered cattle to be a cause of erosion. Nevertheless, in line with class 2, they considered that water quality was good, while less of this class' members considered there were no problems with erosion on their land. This class may thus consists of farmers with some knowledge of problems and what to do about them, with an additional motivation to maintain the currently good water quality.

It can, however, still be considered surprising that higher payments did not increase the likelihood of farmers in class 3 to choose contracts. However, some farmers may value conservation on its own, above the level of payment (e.g. Juutinen et al., 2013; Siebert et al., 2010; Putten et al., 2011). Bremer et al. (2014) found that the majority of Socio Bosque participants they interviewed described incentives as a conservation 'reward'. In general, many studies found that non-financial motivations, such as positive attitudes towards conservation influence participation in conservation-oriented programmes (Cranford and Mourato, 2011; Kosoy et al., 2007; Lienhoop and Brouwer, 2015).

That class 3 members showed a preference for conservation may also be due to their awareness of the Socio Bosque Programme (which was only focused on conservation at the time of field research). For example, Zanella et al. (2014) in their study on PES in Brazil found that access to information on PES programmes was the most important variable explaining higher likelihood of participation.

As was already mentioned, farmers in class 3 on average had larger farm areas, not only forest, but also pasture land. Larger farm sizes have been related to a more pronounced participation in conservation PES programmes (e.g. Zbinden and Lee, 2005).

Class 3 also contained more farm-oriented households, whereas class 2 was the least farm-oriented. Our results are thus in contrast to those of Nordén (2014), who found that landowners who did not participate in the Costa Rican PES programme were more dependent on farm income. Besides the larger land area and thus higher possibility of diversifying land uses, the choice of class 3 farmers for PES contracts could be due to the fact that households who depend on farm income for a larger share of their income could have a strong interest in additional ways to diversify farm-related income sources. PES contracts can have a risk reduction effect, as payments can be a more secure income source for households who want to diversify their income (Siebert et al., 2006; Wünscher et al., 2008).

Overall, the analysis did reveal some problems with the implementation of the proposed PES contracts. Specifically for classes 2 and 3, (higher) payment were not a significant contract attribute. This complicates the implementation of conditionality based on withholding payment. As such, the differentiation between PES and ICDPs based on conditionality becomes weaker. In addition, according to Kauffman (2013), in Ecuador ICDPs are used by to overcome concerns about privatization and the commoditization of nature associated with strict PES schemes. It is argued that "individual cash payments fail to instil a sense of value in conservation among farmers living in watersheds" (Kauffman, 2013, p. 9). The inclusion of ICDP-type attributes in contracts has also been shown to positively impact participation (e.g. Kaczan et al., 2013) or decrease the importance of cash payments (e.g. Balderas Torres et al., 2013).

Conversely, for class 1 conditionality could be implemented, but then the problem may be that the implementation of SPS and additional actions stops, once the contract stops. Hayes (2012) in her analysis of a silvopastoral PES programme in Colombia, found that without payments less than half of the programme participants would continue with the implementation of the SPS. She adds that "for many farmers, the decision to participate in the silvopastoral programme was influenced by perceived immediate gains, not the illusive long-term benefits of the new management system".

According to Siebert et al. (2006), (financial) incentives are a necessary, although clearly not sufficient condition to support practices oriented towards biodiversity protection. As such, to increase support and sustainability of the proposed programme for the adoption of SPS, a hybridized scheme in which PES concepts are combined with ICDPs may be better suited (e.g. Petheram and Campbell, 2010).

The Socio Bosque programme is now providing a range of different modules within its programme, and our analysis showed an interest for specific parts of the programme. However, trust and sustainability beyond conditionality should still be tackled. According to Clements et al. (2010), strengthening local resource management rules and building local management organisations, in addition to payments or up-front delivery of materials, may increase the sustainability of a PES programme, even beyond the duration of the programme. Cranford and Mourato (2011) state that community conservation can improve aspects such as environmental awareness and a social context encouraging conservation in a first stage of PES implementation. Afterwards, an incentive mechanism can be implemented to strengthen the new conservationist behaviour.

Furthermore, our CE was still based on a top-down proposal of what experts considered the most-suited actions for the improvement and conservation of watershed services and biodiversity. According to Hayes (2012), local resource users should be engaged from the start in the design and application of programmes. As such, even those who showed aversion to the contracts we proposed, could have suggested contract attributes that would have increased their choices for PES contracts. Engagement of local stakeholders in a hybridized ICDP-PES scheme may thus maximize participation, facilitate conditionality and increase the sustainability of a PES programme in our research area.

6.5 Conclusions

This chapter used a CE to elucidate farmers' preferences for alternative types of contracts for the adoption of SPS in the buffer zone of Podocarpus National Park. A three-class LCM was found to provide a good representation of observed choices, and a clear and relevant segmentation of farmers' choice behaviour. Farmers were found to have heterogeneous preferences for different SPS contracts with and without additional environmental requirements.

Results confirmed that there is potential for the implementation of a PES-type scheme for the adoption of SPS, given the strong preferences of a small majority of farmers (53%) for some type of contract for the establishment of SPS. In addition, one group of farmers (30%) did not show positive nor negative preference for the BAU. Thus, a large group of farmers could be encouraged to adopt SPS and thereby enhance delivery of certain watershed services. However, those farmers that were least dependent on farm income, showed a strong preference for the BAU. Risk, uncertainty and trust may be reasons as to why some farmers would prefer not to change production systems.

Results suggested that the best way forward for a maximum outreach is the provision of different types of PES contracts within the same incentive programme and to let farmers' chose - as the Socio Bosque programme is increasingly doing. However, while most farmers could be motivated to participate by providing different contract types, some appear to be reluctant to participate in any type of PES scheme. Building of trust, allowing direct participation of farmers in contract and programme design, and the inclusion of ICDPs within the PES programme may be able to address these challenges.

Chapter 7: Conclusions

7.1 Summary of the empirical conclusions

A PES contract is a coordination mechanism between those who provide ES and those who benefit from ES provision, or other actors, normally governments who represent ES users directly or indirectly. Following Van Huylenbroeck et al. (2009), these mechanisms are better understood as contractual arrangements, which is a hybrid mode of governance in contrast to other mechanisms such as markets (coordination through trade) or the strict use of environmental regulations (coordination through authority).

In the research area of this PhD thesis, the land that lies in the buffer zone of Podocarpus National Park is partially used by households for productive purposes. The area hosts globally significant biodiversity and includes many watersheds that provide ES to downstream beneficiaries who mainly reside in the city of Loja. These beneficiaries are organised through a municipal water company. Through water infrastructure it is possible to identify and charge beneficiaries for the provision of watershed services. Furthermore, these services have characteristics of a common-pool-resource (creating the need to coordinate resource use). Biodiversity has public good characteristics and often needs public intervention to assure its conservation. Biodiversity as an ES is frequently linked or bundled with the provision of watershed services. In areas where ES provision is degraded due to farm households' choices for specific land uses, where property rights over land are clear, and where funding mechanisms are available - as is the case in the research area - PES can be a mechanism to conserve and/or improve the provision of ES.

This research provided causal evidence of the effectiveness and impacts of PES and compares different mechanisms aimed at maintaining and/or securing ES, while simultaneously contributing to household income. The study focused on the trade-offs and synergies between nature conservation and the maintenance and improvement of ES, and the provision of rural income in the buffer zone of Podocarpus National Park in Ecuador. Both environmental and social outcomes of PES were evaluated, and compared to other mechanisms in order to increase understanding of the impact of PES relative to other mechanisms.

The first analytical block of this PhD (Chapter 3) introduced PES schemes' design. In this chapter the design characteristics of different Andean and Mesoamerican PES and PES-type schemes were analysed to assess if these schemes, as hybrid forms of governance, over time incorporated more market characteristics, more characteristics of a hierarchy or remained unchanged.

Findings show that the schemes analysed cover a range of hybrid arrangements; some resemble small markets and others are complete hierarchical organisations. Over time, more schemes have incorporated command-based methods to organise users, while most schemes are based on contracts with voluntary providers. Through the provision of positive incentives to providers that aim to increase or maintain the supply of ES there is a certain market orientation in PES. From the users' side there is more diversification, with schemes having incorporated a wide range of participation and payment setting mechanisms. Intermediaries remain the dominant actors in most schemes, and are predominantly national or local governments. Overall the organisation of PES (and PES-like) schemes studied became more complex over time.

Currently, in the common conceptualization of PES, PES are mainly seen as contractual arrangements between ES users, or an entity representing them, and individual and/or communal landholders for the provision of ES. To study the effectiveness of this mode of governance was our main focus in the two chapters devoted to assessing the impact of the PES contracts studied in the second analytical block of this thesis.

In Chapter 4 the cost-effectiveness in terms of ES provision, conservation and rural income generation of three incentive mechanisms were compared: A PES contract, Socio Bosque, was compared to buying-selling land through the FORAGUA programme, which is a hierarchical mode of governance, and the use of an organic coffee label as a case of the use of a market for an environmentally friendly product.

It was found that purchasing land should strongly increase the provision of ES, and thus scores best in terms of additionality. However, taking land out of production is expensive and may have negative effects on local development. Paying private landowners through conservation contracts increased total (rural) income. However, poorer households were often excluded from these contracts, and when they do participate, payments were low due to small land areas and the limited amount of payments they received per hectare. Moreover, mainly due to their lower access to land and higher opportunity costs, it costs more to include smallholders in a PES scheme. Receiving income through price premiums for organic production has, in the area studied, the most positive impact on rural incomes. PES and PES-like schemes were however cheaper to implement than a programme that aims to increase agricultural production. Hence, we found a trade-off between increasing conservation and ES provision, and the generation of rural (household) income.

Next, to gain more insights in the trade-offs between providing ES through the adoption or maintenance of specific land-use practices and income generation, a portfolio analysis was carried out to study the impact and combination of several programmes. The programmes studied were the Ecuadorian incentive programme for forest conservation and restoration on abandoned pastures (Socio Bosque), and an incentive programme for timber plantations. The current land use in the research area was milk production on existing pastures. Additionally one group of farmers had forests on their lands. Results suggest that most farmers would increase the area under conservation and/or restoration in a strategy that would decrease income risks the most in absolute terms (minimum variance portfolio), beyond what would be expected from decisions considering expected return only. In a risk-reduction strategy that aims at maximizing the returns per unit of risk (optimal portfolio), milk producers without forest would also allocate most of their land to restoration, although a small group would continue milk production on most of their land. Milk producers with forest would shift milk production on existing pastures to milk production on pastures established after deforestation under both portfolio strategies, which evidenced a potential leakage effect. Moreover, in any of the portfolio land use allocations analysed that included inventive programmes, only a small area was dedicated to the establishment of tree plantations.

None of the portfolios would increase all households' income to above the poverty line due to small farm sizes and low payments. Moreover, a negative impact on household income was observed from adopting the land use portfolios for milk producers without forest, whereas for forest holders all land use combinations studied would have a positive impact. For those producers without forest, there seems to be a trade-off between maximizing household income on the one hand, and risk reduction through a combination of restoration payments and tree plantations, on the other.

Conservation, and in certain situation restoration payments, can thus have a positive impact on household income when the payments are an extra source of income and/or where payments are higher than income from milk production. Yet, the effect on poverty reduction is limited. The size of Socio Bosque payments is not based on household income nor on the number of people in a household that received the payment, but uses only farm size as an (insufficient) proxy for poverty. So although it may be an important mechanism for conservation, and in the future maybe for restoration, there does exist a trade-off between the dual goals of protecting and restoring the largest area possible versus reducing poverty. This is mainly due to the limited area of land that poor landholders can conserve and/or restore, which results in limited income effects. For tree plantations the biggest issue is the long time span before income becomes available. As such it may be an interesting saving option, but it will not decrease poverty in the short term.

The third analytical block of this PhD research focused on understanding the (potential) impacts of PES contracts by analysing the design and preference for different contracts. Chapter 6 studied the preferences of milk producers in terms of PES contracts for silvopastoral systems with and without additional environmental requirements, and assessed the determinants of these preferences.

Results confirmed that there is potential for the implementation of a PES-type scheme for the adoption of silvopastoral systems in the research area, yet with high heterogeneity in these preferences. A large group of farmers could be encouraged to voluntarily adopt silvopastoral systems and thereby enhance delivery of watershed services and biodiversity. The results suggest that farmland area, income from agriculture - specifically from milk production, the percentage of farm income in total income, and landowners' perception of environmental problems provide a partial explanation for the considerable heterogeneity observed in the preferences for specific contracts. Although offering contracts with varying additional management requirements within the same scheme may contribute to enhance participation, and thus ES provision, enforcing contracts based on conditionality may be problematic as for one group of farmers payment levels did not significantly influence the choice of specific contracts. In addition, one small group of farmers is reluctant to change the way they farm and participate in PES contracts. Building of trust and combining PES with ICDPs may be better to target these farmers.

7.2. Overall conclusions

The main objective of this research was to analyse the (potential) impacts and trade-offs of PES contracts for the conservation and improvement of ES and the provision of rural income; and to compare the use of PES contracts with other incentive mechanisms and land uses implemented in the research area.

Designing an optimal strategy that maximizes both nature conservation and the services it provides, as well as income of poorer households is very difficult. Indeed, the analyses showed that it seems hard to reconcile both policy aims in a single PES scheme, as there is a trade-off between increasing ES provision and improving rural livelihoods.

Socio Bosque's PES contracts are more cost-effective for ES provision. Yet, more production oriented incentives such as the organic coffee label analysed or the potential of increased milk production, seem to be better tools to improve household income (Chapter 4 and 5). Increased milk production will, however, not have the same positive impact on ES provision as organic coffee unless there is a shift towards silvopastoral production systems, which could be supported through a PES mechanism (Chapter 6). Nonetheless, Socio Bosque's contracts do have a strong risk-reduction effect compared to income from more productive land uses, such as milk production and the establishment of tree plantations (Chapter 5). The lower impact of the PES contracts analysed on household income for the poorer households is related to the double effect of low payments and the limited land resources available to many households (Chapter 4 and 5). The and area available also influences the preferences for specific PES contracts, with those with more land being more likely to prefer contracts that include restoration/conservation measures, instead of only productive land uses (Chapter 6).

Overall, the research area is particularly adapted and receptive to the possibility of extending the PES programme from what is currently in place. In order to achieve this, PES should not stand alone and be considered as part of several programmes and/or actions, to promote sustainable ecosystem management. The combination of several programmes operating within the research area could consist of acquiring land in the most severely degraded watersheds or key biodiversity areas (Chapter 3 and 4), paying landowners for forest conservation and restoration of natural vegetation on pastures (Chapter 4 and 5), establishing additional contracts for silvopastoral systems (Chapter 6), and implementing projects to support improvement and/or adoption of coffee or other agroforestry production systems for smallholders (Chapter 4 and 6). This would facilitate balancing increasing conservation, ES provision, and rural livelihoods and landholders' preferences. To enhance the impacts of PES, the broader environment, the allocation of remaining resources such as through the intensification potentially occurring on the remaining land under production, as well as the combined effects when implementing several programmes need to be considered, as these may have a negative effect on the provision of particular ES.

7.3. Theoretical conclusions

The research defined PES as contractual arrangements. They are defined as a coordination mechanism or institutional arrangement that can provide a solution for governing resources with common-pool-resource and public good characteristics.

This theoretical framework provided a tool to connect the actors with the land uses that are part of the resource system, i.e. the buffer zone. The framework was suitable for the analysis of outcomes of PES and other modes of governance used in ES governance, by distinguishing within an action arena the interactions of the different actors from the measured outcomes. The focus was on the operational situation, i.e. the day-to-day impacts of incentives for actors on land uses.

The emphasis on individual households and contracts as a solution to problems of collective action is relevant in the research area, where over 50,000 households benefit from watershed services. In addition, land-use choice in terms of allocating land to different agricultural purposes is based partially on the income generating potential of these land uses. Comparing this potential with alternative income generating PES contracts makes is sensible from a households land allocation point of view.

However, the choice to focus on individual households does not imply that there is no room for community norms and regulation of behaviour in the research area. Community regulations may be implemented together with PES contracts and are not necessarily in conflict with PES. PES are not meant to replace social conventions for nature conservation by money. PES contracts are an additional tool to reward good stewardship that can go beyond the standards set in environmental regulations such as municipal ordinances, and as was previously discussed are closely related to the improved implementation of these regulations. In addition, for joint-PES contracts or for an organic certification to obtain a price premium, collective action through the formation of a producer association is necessary.

PES are not markets. However, theoretical market and hierarchy characteristics provide a tool to understand and better characterize different types of PES schemes. Hereby ES characteristics, user and provider group size and the perception of PES by policy implementers provide explanations of when and how contractual arrangements are used. Further research into the diversity of PES schemes could focus further on transaction costs economics theory (e.g. Williamson, 1985), specifically nature related transactions (Hagedorn, 2008), and understanding hybrid modes of governance not only as combinations of modes of governance, but as specific governance structures with their own defining characteristics.

The resistance amongst certain environmental pressure groups to putting a price on nature is often based on doubts about using efficiency criteria to evaluate the suitability of policies or the 'fear' of the commoditisation of nature. We argue for more cost-effectiveness studies instead of focusing on efficiency because, as proven in Chapter 4, cost-effectiveness analysis enables comparing the impact of different conservation and income generating programmes without having to put a monetary value on ES. Given limited budgets for implementing conservation actions, such analyses provide the necessary information to make informed choices. This does not mean that this PhD research advocates a utilitarian vision on nature, but only takes into account that in places where humans interact with the environment, the productive function of land use is an important driver for human behaviour and hence needs to be taken into account when trying to stop degradation or to reverse the situation. However, in this PhD we focused on the costs of reaching a predefined ecological outcome.

To improve the analysis the optimization of the outcome should also be included. Additionally, the cost-effectiveness analysis could further be improved by using quantitative data on measured ES provision of the different land uses analysed, instead of having to rely on a proxy such as the ESI used. Although the indicator provided a useful tool in Chapter 4 as actual data were not available, the indicator suffers from short-comings. The ESI does not take into account the potential trade-offs or synergies between different ES (Cordingley et al., 2015; Howe et al., 2014) and could raise problems of double counting of ES (Boyd and Banzhaf, 2007; Fu et al., 2010). An additional problem that was not considered when measuring ES provision with the ESI was the potential impact of scale. Scale can have a positive impact on the provision of ES, such as the positive impact of an increased forest area on water flow regulation (Farley et al., 2005; Thomas and Nisbet, 2007).

Portfolio theory was used to analyse income generation and risk reduction when adopting diversification strategies, while a choice experiment was used to analyse landholders' preferences for PES contracts for the adoption of silvopastoral systems. Such modelling and ex ante simulation exercises can be useful to predict behaviour of the local actors and to optimize the impact of proposed PES schemes as they operationalize the trade-offs between improving ES provision and income generation/risk reduction, and thus predict expected behaviour of potential stakeholders.

In this research poverty was understood from a monetary point of view and not from the broader vision of capabilities, livelihoods and well-being. This choice was inspired by the nature of PES, being a monetary transfer to ES providers impacting directly on income. The impact on well-being could be an area of future research.

A weakness of many PES is that they are externally designed and proposed and often do not take the opinion of the local stakeholders into account, while these stakeholders may have other possible solutions in mind. We therefore think that the analytical approaches proposed in this PhD may also be useful from this point of view as they can help to fill the gap between external and internal objectives and viewpoints. An additional step could be to link our research methodology to participative policy or planning tools. This could increase the fairness feeling about proposed solutions and thus also the acceptance and approval rate by local inhabitants.

A last theoretical remark is that, although this research started from a broad perspective on the provision of environmental goods and of ES, our analysis was limited to land uses that provide ES. The PES analysed intervene in the way land is used and do not concentrate on the commoditisation of environmental goods produced or on ES that stand apart from complex ecosystems.

7.4. Conclusions and recommendations

7.4.1. PES as a mechanism for nature conservation and the improvement of ES provision

It is widely acknowledged that PES are not a (or the) silver bullet that will halt or even reverse environmental degradation in the tropics. However, in areas where there is a clear link between land use and the ES provided, and between these ES and the benefits for their users, PES can improve ES delivery and provide a clear framework to justify the collection and spending of PES resources. These resources can be taxes or tariffs (as in the research area), but can also be provided through private financing in addition to government resources.

Some criticize PES because they are a way to create markets and put a price on nature, while they do not necessarily put a halt on ecosystem degradation. The focus on nature conservation/restoration and land use is in our view a way to go beyond this debate and not fall in the pitfall of commoditization of ES as such. Markets for ES and commoditized ES are actually an exception (Chapter 3), whereas putting a price on nature is not a prerequisite for PES (e.g. Wunder, 2013).

Land use decisions are often based on production choices, which impact on ES provision. If PES can alter this choice, they can make a positive contribution to nature conservation and ES provision. From an expected revenue perspective, the payments currently offered by the Socio Bosque programme are for most farmers lower than the income gained from alternative land uses; as such, the contracts do not fully compensate farmers for land use changes. This could decrease the impact of these contracts on improvement in ES provision. However, the (potential) impact on opportunity costs of improved application of land use regulations, and the risk reduction effect of contracts makes PES an interesting option for diversification strategies (Chapter 5). Furthermore, in the research area we found that households in general did express preferences for such contracts. Although households revealed these preferences in different ways, for different management requirements with diverse levels of ES provision (Chapter 6).

Payments included in PES contracts can however still be too low to improve ES provision in areas where higher value commodities are produced, or where there is high potential for land use intensification.

In both cases payments would have to be higher. Therefore PES are mainly a tool in areas where the trade-offs between ES provision and exploitive land uses are limited, but important, as is the case for most milk farms analysed in the research area (Chapter 4 and 5). PES can thus help to achieve conservation outcomes, but not everywhere, and not with every individual landholder. Their impact on nature conservation is much lower than the impact international commodities' markets and the implementation of protected areas have on land use decisions.

Protected area implementation is linked to land use regulations, and PES contracts are not independent from these regulations (Chapter 3 and 5). As good water quality is a necessity, and as a sufficient number of landholders need to participate in a scheme to make PES effective, the implementation of PES contracts is strongly related to the implementation of regulative approaches. Within the research area this relates specifically to the implementation of municipal ordinances to govern land uses (Chapter 5 and 6). As was mentioned earlier, regulations can decrease the opportunity costs of land uses with a (more) negative impact on ES provision, and as such make PES more attractive. Additionally, the combined use of regulations and PES can assure that a sufficient number of landholders participate to make the scheme effective.

This observation may go against the perception that PES participation should be completely voluntary (see Chapter 3). However, as was also observed in Chapter 3, legislation exists, also in Ecuador, that regulates land uses such as forest conversion (see also Chapter 2). The improved implementation of the law can go in hand with PES initiatives, even when this may decrease the efficiency of PES schemes. This joint implementation has been observed in Costa Rica's national scheme, where both the scheme and new forestry legislation were introduced simultaneously (Arriagada et al., 2012; Calvo-Alvarado et al., 2009; Sanchez-Azofeifa et al., 2007). Furthermore, compulsory participation in PES schemes exists (see for example China's sloping land programme, (Kelly and Huo, 2013)). Mandatory participation has been linked to improved positive environmental effects of PES schemes (Brouwer et al., 2011). PES are not a tool that is additional to the law, but a part of it. As such PES contracts should be seen as a set of institutional arrangements that have the potential to reinforce the law in its implementation and impact.

However, whether it is 'fair' to pay landholders to abide the existing laws is an ongoing discussion. In areas such as the research area it may be considered that it is fair to pay landholders through PES for income losses due to investment in conservation/restoration activities, as the majority of households are not part of the richest segments of society (see Chapters 4, 5 and 6). Moreover, including landholders as ES stewards may improve the protection status of the buffer zone, and thus also of the protected area, by increasing the number of people involved in conservation.

Besides PES for conservation and restoration of natural ecosystems, the research also showed the potential of including productive land uses in PES (Chapter 6). In the analysis productive land uses were mainly considered from the perspective of household income generation (Chapter 4 and 5), and less as a system to provide ES (Chapter 4 and 6). Yet, when no natural ecosystems have to be degraded for their establishment, certain productive systems can have a positive impact on ES provision. There is increasing evidence that biologically diversified farming systems can provide a much wider range of ES additional to food or fibre (Gomiero et al., 2011; Kremen and Miles, 2012). Diversified farming systems can achieve this provision of ES producing less externalities than conventional farming systems (Gomiero et al., 2011; Tilman et al., 2002). These systems can additionally take advantage of specific ES such as pollination or pest control (Hajjar et al., 2008; Letourneau et al., 2009; Morandin and Winston, 2006; Vandermeer et al., 2010), although they are not always as productive as conventional farming systems (de Ponti et al., 2012; Seufert et al., 2012).

Finally, the potential of PES impacting positively on ES provision and biodiversity conservation is not only linked to the use of positive incentives to influence landholders land use decisions, but also to the sanctions for non-compliance with the agreed contractual obligations (conditionality, as discussed in Chapter 3). This conditionality is important to ensure compliance for the duration of the PES contract. In order to further strengthen the conditionality linked to PES, an additional important aspect that should be addressed is the lack of sanctioning mechanisms for non-compliance for the entity that signs the contracts with ES providers (such as the Socio Bosque Secretariat). Providers can be sanctioned in most PES schemes, for example through withholding or stopping payments (as for example proposed in Chapter 6). PES contracts, however, normally do not stipulate sanctioning mechanisms in case a government or other actors involved in PES implementation decide to no longer pay providers, in a situation where providers are complying with the contract. There is often little accountability and external control for implementers. This asymmetry can be a source of distrust of landholders towards PES, hence potentially complicating PES participation (see e.g. preferences of some milk producers for the BAU scenario in Chapter 6). The lack of conditionality for users (and/or buyers) can furthermore decrease environmental effectiveness and PES sustainability. There is thus a need to develop PES with contractual obligations from both sides.

7.4.2. PES as a mechanism to reduce poverty and empower rural households

Despite some uncertainty linked with PES conditionality, it is much more complicated to use PES as a way to reduce poverty than it is to decrease environmental degradation. This was observed in Chapters 4 and 5. A significant positive impact on household income is unlikely in the current Socio Bosque Programme. Under the existing system poorer households will not be lifted out of poverty unless they own larger tracks of forests. Hence, to contribute more to poverty alleviation, payment levels should either increase or poorer households could be targeted by programmes focusing more on environmentally-friendly production systems that increase land productivity and/or agricultural income (Chapters 4 and 6). The latter is especially relevant if we consider that increasing PES payments could decrease environmental effectiveness, if increased payments are not covered through an increase in the budget available. With increased payments and a fixed budget, less households and thus a lower land area would be targeted. PES, as a contract for ES provision, can only do as much as to pay a correct remuneration for the delivery of ES or for the implementation of specific land use systems. However, PES are not the right policy option to address issues such as land inequality, real poverty reduction or income generation. Other actions such as land reform, improvement of infrastructure, education and off-farm job opportunities will probably have a much larger impact than any land-based policy, because poorer households are restricted in the land they own. PES can thus only be considered an additional tool for poverty reduction. Yet, it can be a choice to only pay poorer landholders for conservation even if that would decrease environmental effectiveness. This may however conflict with the initial objective of PES, which is to reduce environmental degradation. Moreover, only paying poorer households for conservation and not the relatively richer households may create conflicts in the programme area a and within communities. PES as a tool for poverty alleviation is thus probably more effective in areas with large numbers of poorer households and/or poor communities, where conflict between those who receive PES (poorer households) and those who don't (richer households) is less likely, as all or most households will benefit.

One aspect that may not directly impact household income, but that can increase social capital and decision-making power for poorer households, is taking into account preferences of households for specific land uses within a PES scheme, as in Chapter 6. This implies that ideally landholders should be included in PES design and implementation from the start.

FORAGUA (or its constituent, the municipality of Loja) does not implement PES within the research area, because all power is with the municipal water company that does not want to pay landholders for ES provision out of fear of being blackmailed by them (Chapter 3). As such, all decision-making power lies with one actor, who does not seem to trust the local landholders and who assumes to have the best knowledge on how to manage watersheds. PES implementation can spread the decision-making power as also landholders can then decide which land uses they prefer to implement. However, this will only work if the implementation of a PES scheme goes together with actively building trust between landholders and the watershed management, as was discussed in Chapter 6. Implementing a PES scheme goes hand in hand with strengthening local institutions, even if this would require more time to implement. PES schemes may then be more sustainable over time (e.g. Clements et al., 2010).

The absence of a shared decision-making process was not only observed when analysing FORAGUA and landholders within the watershed, but also occurs on a national level (Chapter 3). Making the state the sole 'owner' of ES, as in Ecuador, basically implies that a single actor can decide when, where and how PES should be implemented. However, to improve targeting it would be better if states coordinate certain actions, regional governments others, and local businesses and households even others, i.e. if there are different levels of governance.

7.4.3. PES and multiple levels of governance

Payments to improve or conserve the provision of specific ES can be part of a rural development scheme, which is designed on a local scale as this facilitates targeting. Although the Socio Bosque programme is more differentiated than it was originally, it remains a nationally designed and implemented programme. It is still a rather homogeneous solution to a diversity of contexts found within Ecuador. In a PES scheme linked to a rural development scheme, water funds and municipal water managers could be used to connect the nationally coordinated and financed programmes with the locally implemented and financed provision of watershed services (see e.g. Chapter 4). A broader diversity of mechanisms could be used, based on their importance for ES provision (e.g. acquiring key areas, offering higher payments for conservation actions, and implementing PES with projects for sustainable production systems – Chapters 4, 5 and 6), and based on giving households the choice among different land-use options to improve ES provision on their land (Chapter 6). Ideally, PES should thus be operating at different levels.

From a financing perspective additional levels of governance can include local actors and authorities responsible for funding the provision of services with local benefits (typically watershed services), supplemented by national and international resources for biodiversity, which provides global benefits. Furthermore, local actors (landholders and communities and local PES implementers) should then be responsible for the direct implementation of the agreed upon land uses. This local orientation could assure that land uses included in the PES scheme not only assure ES provision, but are also of interest to landholders in a specific area and adapted to local circumstances. At this level voluntary and compulsory payments (monetary or even in-kind) for local ES can be collected and managed. However, to assure that money is correctly collected and used, and to assure that not all resources are absorbed by one actor (for example a powerful landholder or a municipal agency that does not trust local landholders) on a second level a national (government) agency could be responsible, not for PES implementation, but for the control of how resources are spent and how the local PES scheme is managed. As such PES implementation would be more closely linked with how municipalities and the national government relate to one another. On a national level resources could be collected and redistributed to ensure the provision of ES such as biodiversity, but these could also be used to ensure that also poorer municipalities have access to PES resources (similar to the inter-municipal solidarity mechanism of FORAGUA as explained in Annex 2). Finally international users (buyers) of ES can additionally sign contracts with local or national PES implementers to finance a PES scheme. These multiple levels would also assure that schemes are not only financed locally. As Van Hecken and Bastiaensen (2010b) pointed out, it could be unfair to make only local users in poorer countries pay for ES that provide global benefits. PES, and specifically PES financing, should operate in multiple governance levels. The international payments should however not be charitable donations, but payment levels and duration should be based on specific goals (hectares of forest replanted, area conserved, etc.). This adds an additional layer of control, while also assuring extra income.

Finally, PES contracts can address the immediate cause of resource degradation, but not its broader causes, such as population growth and the increasing demand for resources, climate change due to the use of certain resources, etc. PES is not a tool to address these global issues and they are not capable of coordinating such complex relations.

PES should be understood as one tool, to be applied in specific circumstances among all different mechanisms and policies to address the complexities of achieving sustainable development.

References

Adhikari, B., Agrawal, A., 2013. Understanding the social and ecological outcomes of PES projects: A review and an analysis. Conserv. Soc. 11, 359. doi:10.4103/0972-4923.125748

Agresti, A., 2007. An Introduction to Categorical Data Analysis, 2nd ed, Wiley Series in Probability and Statistics. John Wiley & Sons, Inc., New Jersey, USA.

Aguirre, N., Palomeque, X., Weber, M., Stimm, B., Günter, S., 2011. Reforestation and Natural Succession as Tools for Restoration on Abandoned Pastures in the Andes of South Ecuador, in: Günter, S., Weber, M., Stimm, B., Mosandl, R. (Eds.), Silviculture in the Tropics, Tropical Forestry. Springer Berlin Heidelberg, pp. 513–524.

Ahn, J., Khandelwal, A.K., Wei, S.-J., 2011. The role of intermediaries in facilitating trade. J. Int. Econ. 84, 73–85. doi:10.1016/j.jinteco.2010.12.003

Aimin, H., 2010. Uncertainty, Risk Aversion and Risk Management in Agriculture. Agric. Agric. Sci. Procedia 1, 152–156. doi:10.1016/j.aaspro.2010.09.018

Aizaki, H., 2012. Basic functions for supporting an implementation of choice experiments in R. J. Stat. Softw. Code Snippets 50, 1–24.

Alavalapati, J.R.R., Shrestha, R.K., Stainback, G.A., Matta, J.R., 2004. Agroforestry development: An environmental economic perspective. Agrofor. Syst. 61-62, 299–310. doi:10.1023/B:AGFO.0000029006.64395.72

Albán, M., Argüello, M., 2004. Un análisis de los impactos sociales y económicos de los proyectos de fijación de carbono en el Ecuador. El caso de PROFAFOR-FACE (No. 07), Markets for Environmental Services. Internal Institute for Environment and Development, London, UK.

Alvarado-Quesada, I., Hein, L., Weikard, H.-P., 2014. Market-based mechanisms for biodiversity conservation: a review of existing schemes and an outline for a global mechanism. Biodivers. Conserv. 23, 1–21. doi:10.1007/s10531-013-0598-x

Angelsen, A., Wunder, S., 2003. Exploring the forest--poverty link: key concepts, issues and research implications (No. 40), CIFOR Occasional Paper. CIFOR, Bogor, Indonesia.

Antle, J.M., 1987. Econometric Estimation of Producers' Risk Attitudes. Am. J. Agric. Econ. 69, 509–522. doi:10.2307/1241687

Arriagada, R., 2009. Payments for environmental services and their impact on forest transition in Costa Rica (Working Paper No. 200907). Latin American and Caribbean Environmental Economics Program.

Arriagada, R.A., Ferraro, P.J., Sills, E.O., Pattanayak, S.K., Cordero-Sancho, S., 2012. Do payments for environmental services affect forest cover? A farm-level evaluation from Costa Rica. Land Econ. 88, 382–399.

Arriagada, R., Perrings, C., 2009. Making Payments for Ecosystem Services Work (Working Paper), Ecosystem Services Economics. UNEP, Nairobi, Kenya.

Ascough, J.C., Maier, H.R., Ravalico, J.K., Strudley, M.W., 2008. Future research challenges for incorporation of uncertainty in environmental and ecological decision-making. Ecol. Model. 219, 383–399. doi:10.1016/j.ecolmodel.2008.07.015

Aspers, P., 2007. The Practice of Defining Markets: A Comment on Charles W. Smith. Can. J. Sociol. 32, 477–486.

Asquith, N., 2011. Pagos por servicios ambientales hidricos (PSA-H) en Bolivia.

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

Asquith, N., Vargas, M.T., 2007. Fair deals for watershed services in Bolivia. Natural Resource Issues No. 7. International Institute for Environment and Development, London, UK.

Ataroff, M., Monasterio, M., 1997. Soil erosion under different management of coffee plantations in the Venezuelan Andes. Soil Technol., Soil Erosion Processes on Steep Lands 11, 95–108. doi:10.1016/S0933-3630(96)00118-3

Ataroff, M., Rada, F., 2000. Deforestation Impact on Water Dynamics in a Venezuelan Andean Cloud Forest. AMBIO J. Hum. Environ. 29, 440–444. doi:10.1579/0044-7447-29.7.440

Ávalos-Sartorio, B., Blackman, A., 2010. Agroforestry price supports as a conservation tool: Mexican shade coffee. Agrofor. Syst. 78, 169–183. doi:10.1007/s10457-009-9248-4

Balderas Torres, A., MacMillan, D.C., Skutsch, M., Lovett, J.C., 2013. Payments for ecosystem services and rural development: Landowners' preferences and potential participation in western Mexico. Ecosyst. Serv., Payments for Ecosystem Services and Their Institutional Dimensions: Analyzing the Diversity of Existing PES Approaches in Developing and Industrialized Countries 6, 72–81. doi:10.1016/j.ecoser.2013.03.002

Balmford, A., Whitten, T., 2003. Who should pay for tropical conservation, and how could the costs be met? Oryx 37, 238–250. doi:10.1017/S0030605303000413

Balvanera, P., Kremen, C., Martínez-Ramos, M., 2005. Applying Community Structure Analysis to Ecosystem Function: Examples from Pollination and Carbon Storage. Ecol. Appl. 15, 360–375.

Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J.-S., Nakashizuka, T., Raffaelli, D., Schmid, B., 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecol. Lett. 9, 1146–1156. doi:10.1111/j.1461-0248.2006.00963.x

Balvanera, P., Uriarte, M., Almeida-Leñero, L., Altesor, A., DeClerck, F., Gardner, T., Hall, J., Lara, A., Laterra, P., Peña-Claros, M., Silva Matos, D.M., Vogl, A.L., Romero-Duque, L.P., Arreola, L.F., Caro-Borrero, Á.P., Gallego, F., Jain, M., Little, C., de Oliveira Xavier, R., Paruelo, J.M., Peinado, J.E., Poorter, L., Ascarrunz, N., Correa, F., Cunha-Santino, M.B., Hernández-Sánchez, A.P., Vallejos, M., 2012. Ecosystem services research in Latin America: The state of the art. Ecosyst. Serv. 2, 56–70. doi:10.1016/j.ecoser.2012.09.006

Barham, B.L., Callenes, M., Gitter, S., Lewis, J., Weber, J., 2011. Fair Trade/Organic Coffee, Rural Livelihoods, and the "Agrarian Question": Southern Mexican Coffee Families in Transition. World Dev. 39, 134–145. doi:10.1016/j.worlddev.2010.08.005

Barrera, V.H., Alwang, J., Cruz Collaguazo, E. del P., 2010. Análisis de la viabilidad socio-económica y ambiental del sistema de producción papa-leche en la microcuenca del río Illangama-Ecuador. Arch. Latinoam. Prod. Anim. 18, 57–67.

Bateman, I.J., Carson, R.T., Day, B., Hanemann, M., Hanleys, N., Hett, T., Jones-Lee, M., Loomes, G., Mourato, S., Ozdemiroglu, E., Pearce, D., Sugden, R., Swanson, J., 2002. Economic valuation with stated preference techniques: a manual. Edward Elgar, Cheltenham, UK.

Bathurst, J.C., Bovolo, C.I., Cisneros, F., 2010. Modelling the effect of forest cover on shallow landslides at the river basin scale. Ecol. Eng., Special Issue: Vegetation and Slope Stability 36, 317–327. doi:10.1016/j.ecoleng.2009.05.001

Beck, E., Richter, M., 2008. Ecological aspects of a biodiversity hotspot in the Andes of southern Ecuador, in: Gradstein, S.R., Homeier, J., Gansert, D. (Eds.), The Tropical Mountain Forest - Patterns and Processes in a Biodiversity Hotspot, Biodiversity and Ecology Series. Göttingen Centre for Biodiversity and Ecology, Univ. Verlag Göttingen, Göttingen, Germany, pp. 195–217.

Begg, D.K.H., Fischer, S., Dornbusch, R., 2003. Economics, 7th ed. McGraw-Hill, London, UK.

Beharry-Borg, N., Smart, J.C.R., Termansen, M., Hubacek, K., 2013. Evaluating farmers' likely participation in a payment programme for water quality protection in the UK uplands. Reg. Environ. Change 13, 633–647. doi:10.1007/s10113-012-0282-9

Ben-Akiva, M., Lerman, S.R., 1985. Discrete Choice Analysis. Theory and Application to Travel Demand, MIT Press Series in Transportation Studies. MIT Press, Cambridge, MA, USA.

Benitez, P.C., Kuosmanen, T., Olschewski, R., van Kooten, G.C., 2006. Conservation payments under risk: A stochastic dominance approach. Am. J. Agric. Econ. 88, 1–15. doi:10.1111/j.1467-8276.2006.00835.x

Bennett, A.F., Radford, J.Q., Haslem, A., 2006. Properties of land mosaics: Implications for nature conservation in agricultural environments. Biol. Conserv. 133, 250–264. doi:10.1016/j.biocon.2006.06.008

Beuchelt, T.D., Zeller, M., 2011. Profits and poverty: Certification's troubled link for Nicaragua's organic and fairtrade coffee producers. Ecol. Econ. 70, 1316–1324. doi:10.1016/j.ecolecon.2011.01.005

Biernacki, P., Waldorf, D., 1981. Snowball Sampling: Problems and Techniques of Chain Referral Sampling. Sociol. Methods Res. 10, 141–163. doi:10.1177/004912418101000205

Blackman, A., Albers, H.J., Ávalos-Sartorio, B., Crooks, L., 2003. Land cover in a managed forest ecosystem: Mexican shade coffee (Discussion Paper No. 03-60). Resources for the Future, Washington DC, USA.

Blackman, A., Ávalos-Sartorio, B., Chow, J., 2007. Shade Coffee & Tree Cover Loss: Lessons from El Salvador. Environ. Sci. Policy Sustain. Dev. 49, 22–33. doi:10.3200/ENVT.49.7.22-33

Blanche, R., Cunningham, S.A., 2005. Rain Forest Provides Pollinating Beetles for Atemoya Crops. J. Econ. Entomol. 98, 1193–1201. doi:10.1603/0022-0493-98.4.1193

Bocquého, G., Jacquet, F., Reynaud, A., 2014. Expected utility or prospect theory maximisers? Assessing farmers' risk behaviour from field-experiment data. Eur. Rev. Agric. Econ. 41, 135–172. doi:10.1093/erae/jbt006

Bode, R., 2007. Knowledge management, learning and communication in value chains. A case analysis of the speciality coffee value chain of FAPECAFES, Ecuador. CIAT, Cali, Colombia.

Bodie, Z., Kane, A., Marcus, A., Perrakis, S., Ryan, P., 2008. Investments, 6th Cdn edition, Canadian edition. ed. McGraw-Hill Ryerson Higher Education, Toronto.

Bøgh, A., 1992. Composition and distribution of the vascular epiphyte flora of an Ecuadorian montane rain forest. Selbyana 13, 25–34.

Bond, G., Wonder, B., 1980. Risk attitudes amongst Australian farmers. Aust. J. Agric. Econ. 24, 16–34.

Borkhataria, R.R., Collazo, J.A., Groom, M.J., 2006. Additive Effects Of Vertebrate Predators On Insects In A Puerto Rican Coffee Plantation. Ecol. Appl. 16, 696–703. doi:10.1890/1051-0761(2006)016[0696:AEOVPO]2.0.CO;2

Börner, J., Wunder, S., 2012. The Scope for Reducing Emissions from Forestry and Agriculture in the Brazilian Amazon. Forests 3, 546–572. doi:10.3390/f3030546

Börner, J., Wunder, S., Wertz-Kanounnikoff, S., Tito, M.R., Pereira, L., Nascimento, N., 2010. Direct conservation payments in the Brazilian Amazon: Scope and equity implications. Ecol. Econ. 69, 1272–1282. doi:10.1016/j.ecolecon.2009.11.003

Boström, M., Klintman, M., 2006. State-centered versus Nonstate-driven Organic Food Standardization: A Comparison of the US and Sweden. Agric. Hum. Values 23, 163–180. doi:10.1007/s10460-005-6099-0

Boxall, P.C., Adamowicz, W.L., 2002. Understanding Heterogeneous Preferences in Random Utility Models: A Latent Class Approach. Environ. Resour. Econ. 23, 421–446. doi:10.1023/A:1021351721619

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. Ecol. Econ., Ecological Economics of Coastal DisastersCoastal Disasters Special Section 63, 616–626. doi:10.1016/j.ecolecon.2007.01.002

Brauman, K.A., Daily, G.C., Duarte, T.K., Mooney, H.A., 2007. The Nature and Value of Ecosystem Services: An Overview Highlighting Hydrologic Services. Annu. Rev. Environ. Resour. 32, 67–98. doi:10.1146/annurev.energy.32.031306.102758

Brealey, R.A., Myers, S.C., Allen, F., 2011. Principles of Corporate Finance, 10th ed. McGraw-Hill Education, Columbus, USA.

Bremer, L.L., Farley, K.A., Lopez-Carr, D., 2014a. What factors influence participation in payment for ecosystem services programs? An evaluation of Ecuador's SocioPáramo program. Land Use Policy 36, 122–133. doi:10.1016/j.landusepol.2013.08.002

Bremer, L.L., Farley, K.A., Lopez-Carr, D., Romero, J., 2014b. Conservation and livelihood outcomes of payment for ecosystem services in the Ecuadorian Andes: What is the potential for “win–win”? Ecosyst. Serv. doi:10.1016/j.ecoser.2014.03.007

Broch, S.W., Vedel, S.E., 2012. Using Choice Experiments to Investigate the Policy Relevance of Heterogeneity in Farmer Agri-environmental Contract Preferences. Environ. Resour. Econ. 51, 561–581. doi:http://dx.doi.org/10.1007/s10640-011-9512-8

Bromiley, P., Cummings, L.L., 1995. Transactions Costs in Organizations with Trust, in: Bies, R.J., Lewicki, R.J., Sheppard, B.H. (Eds.), Research on Negotiation in Organizations. JAI Press Inc., Greenwich CT, USA, pp. 219–247.

Bromley, D., 1989. Economic Interests and Institutions: The Conceptual Foundations of Public Policy. Basil Blackwell, New York and Oxford.

Bromley, D.W., 1997. Rethinking markets. Am. J. Agric. Econ. 79, 1383–1393.

Brouwer, R., Tesfaye, A., Pauw, P., 2011. Meta-analysis of institutional-economic factors explaining the environmental performance of payments for watershed services. Environ. Conserv. 38, 380–392. doi:10.1017/S0376892911000543

Bruijnzeel, L.A., 2004. Hydrological functions of tropical forests: not seeing the soil for the trees? Agric. Ecosyst. Environ. 104, 185–228. doi:10.1016/j.agee.2004.01.015

Bryner, N., Fierst, S., Ludwig, J., Stackhouse, T., 2012. COMSERBO: A Case Study of Community-Based Forest Conservation in the Bolivian Amazon, Latin American and Hemispheric Studies Program. Master of Arts Capstone Research Project. Elliott School of International Affairs. The George Washington University.

Buytaert, W., Iñiguez, V., Bièvre, B.D., 2007. The effects of afforestation and cultivation on water yield in the Andean páramo. For. Ecol. Manag., Planted Forests and Water Including selected papers from a special session of the World Congress of the International Union of Forest Research Organisations, Brisbane 8-13 August 2005 251, 22–30. doi:10.1016/j.foreco.2007.06.035

Calle, A., Montagnini, F., Felipe Zuluaga, A., 2009. Farmer’s perceptions of silvopastoral system promotion in Quindio, Colombia. Bois Forets Trop. 79–94.

Calvo-Alvarado, J., McLennan, B., Sánchez-Azofeifa, A., Garvin, T., 2009. Deforestation and forest restoration in Guanacaste, Costa Rica: Putting conservation policies in context. For. Ecol. Manag., Ecology and regeneration of tropical dry forests in the Americas: Implications for management 258, 931–940. doi:10.1016/j.foreco.2008.10.035

Camacho, D.C., 2008. Payment schemes for environmental services in watersheds in Ecuador. Investig. Agrar.-Sist. Recur. For. 17, 54–66.

Campbell, P.J.Y., Viceira, P.L.M., 2002. Strategic Asset Allocation: Portfolio Choice for Long-Term Investors, Clarendon Lectures in Economics. Oxford University Press, Oxford, UK.

Canty, A., Ripley, B., 2015. boot: Bootstrap R (S-Plus) Functions. R package version 1.3-15.

Cao, S., Zhong, B., Yue, H., Zeng, H., Zeng, J., 2009. Development and testing of a sustainable environmental restoration policy on eradicating the poverty trap in China’s Changting County. Proc. Natl. Acad. Sci. U. S. A. 106, 10712–10716. doi:10.1073/pnas.0900197106

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S., Naeem, S., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59–67. doi:10.1038/nature11148

Carpenter, S.R., Mooney, H.A., Agard, J., Capistrano, D., DeFries, R.S., Díaz, S., Dietz, T., Duraiappah, A.K., Oteng-Yeboah, A., Pereira, H.M., Perrings, C., Reid, W.V., Sarukhan, J., Scholes, R.J., Whyte, A., 2009. Science for managing ecosystem services: Beyond the Millennium Ecosystem Assessment. Proc. Natl. Acad. Sci. 106, 1305–1312. doi:10.1073/pnas.0808772106

Carpentier, C.L., Vosti, S.A., Witcover, J., 2000. Intensified production systems on western Brazilian Amazon settlement farms: could they save the forest? Agric. Ecosyst. Environ. 82, 73–88. doi:10.1016/S0167-8809(00)00217-6

Carret, J.C., Loyer, D., 2003. Madagascar protected area network sustainable financing economic analysis perspective. Presented at the 5th World Parks Conference, Durban, South Africa.

Carwardine, J., Wilson, K.A., Ceballos, G., Ehrlich, P.R., Naidoo, R., Iwamura, T., Hajkowicz, S.A., Possingham, H.P., 2008. Cost-effective priorities for global mammal conservation. Proc. Natl. Acad. Sci. 105, 11446–11450. doi:10.1073/pnas.0707157105

Casas, A., 2008. Marcos legales para el pago por servicios ambientales en America Latina y el Caribe: Análisis de Ocho Países. Organization of American States, Washington DC, USA.

Castaño-Villa, G.J., Estevez, J.V., Fontúrbel, F.E., 2014. The role of native forest plantations in the conservation of Neotropical birds: the case of the Andean alder. J. Nat. Conserv. 22, 547–551. doi:10.1016/j.jnc.2014.08.010

Castelle, A.J., Johnson, A.W., 2000. Riparian Vegetation Effectiveness (No. 799), Tech. Bull. National Council for Air and Stream Improvement, Research Triangle Park, NC, USA.

Castro, L.M., Calvas, B., Hildebrandt, P., Knoke, T., 2013. Avoiding the loss of shade coffee plantations: how to derive conservation payments for risk-averse land-users. Agrofor. Syst. 87, 331–347. doi:10.1007/s10457-012-9554-0

Castro, L.M., Calvas, B., Knoke, T., 2015. Ecuadorian Banana Farms Should Consider Organic Banana with Low Price Risks in Their Land-Use Portfolios. PLoS ONE 10, e0120384. doi:10.1371/journal.pone.0120384

Célleri, R., Feyen, J., 2009. The Hydrology of Tropical Andean Ecosystems: Importance, Knowledge Status, and Perspectives. Mt. Res. Dev. 29, 350–355. doi:10.1659/mrd.00007

Celleri, R., Willems, P., Buytaert, W., Feyen, J., 2007. Space–time rainfall variability in the Paute basin, Ecuadorian Andes. Hydrol. Process. 21, 3316–3327. doi:10.1002/hyp.6575

CFN, 2015. Programa de Agroforestería [WWW Document]. Corp. Financ. Nac. URL http://www.cfn.fin.ec/index.php?option=com_content&view=article&id=1245&Itemid=785 (accessed 6.29.15).

Chacón, G., Gagnon, D., Paré, D., 2009. Comparison of soil properties of native forests, Pinus patula plantations and adjacent pastures in the Andean highlands of southern Ecuador: land use history or recent vegetation effects? Soil Use Manag. 25, 427–433. doi:10.1111/j.1475-2743.2009.00233.x

Chaddad, F., 2009. Both Market and Hierarchy: Understanding the Hybrid Nature of Cooperatives. Presented at the International Workshop “Rural Cooperation in the 21st Century: Lessons from the Past, Pathways to the Future,” Rehovot, Israel.

Chandler, D.G., 2006. Reversibility of forest conversion impacts on water budgets in tropical karst terrain. For. Ecol. Manag., Catchment Processes in Southeast Asia 224, 95–103. doi:10.1016/j.foreco.2005.12.010

Chang, H.-H., Boisvert, R.N., 2009. Are farmers’ decisions to work off the farm related to their decisions to participate in the conservation reserve program? Appl. Econ. 41, 71–85. doi:10.1080/00036840701720861

Chang, T.-J., Yang, S.-C., Chang, K.-J., 2009. Portfolio optimization problems in different risk measures using genetic algorithm. Expert Syst. Appl. 36, 10529–10537. doi:10.1016/j.eswa.2009.02.062

Chará, J., Murgueitio, E., 2005. The role of silvopastoral systems in the rehabilitation of Andean stream habitats. Livest. Res. Rural Dev. 17, Article 20.

Chavas, J.-P., 2009. On the Productive Value of Biodiversity. Environ. Resour. Econ. 42, 109–131. doi:10.1007/s10640-008-9206-z

CI, 1997. The Cordillera del Cóndor Region of Ecuador and Peru: A biological assessment. (Working Paper No. 7), RAP. Conservation International, Arlington, VA, USA.

Clarke, R., Silva, H. de, Thorley, S., 2013. Risk Parity, Maximum Diversification,and Minimum Variance: An Analytic Perspective. J. Portf. Manag. 39, 39–53. doi:10.3905/jpm.2013.39.3.039

Clark, S.G., Cherney, D.N., Angulo, I., Bernardi De Leon, R., Moran-Cahusac, C., 2009a. A Problem-Oriented Overview of Management Policy for Podocarpus National Park, Ecuador. J. Sustain. For. 28, 663–679. doi:10.1080/10549810902936243

Clark, S.G., Cherney, D.N., Ashton, M.S., 2009b. Development and Environmental Challenges, Podocarpus National Park, Ecuador. J. Sustain. For. 28, 597–613. doi:10.1080/10549810902922268
Clements, T., John, A., Nielsen, K., An, D., Tan, S., Milner-Gulland, E.J., 2010. Payments for biodiversity conservation in the context of weak institutions: Comparison of three programs from Cambodia. Ecol. Econ. 69, 1283–1291. doi:10.1016/j.ecolecon.2009.11.010
Clements, T., Rainey, H., An, D., Rours, V., Tan, S., Thong, S., Sutherland, W.J., Milner-Gulland, E.J., 2013. An evaluation of the effectiveness of a direct payment for biodiversity conservation: The Bird Nest Protection Program in the Northern Plains of Cambodia. Biol. Conserv. 157, 50–59. doi:10.1016/j.biocon.2012.07.020
Coase, R.H., 1960. Problem of Social Cost, The. JL Econ 3, 1.
CONAFOR, 2013. Reglas de Operación del Programa Nacional Forestal 2013 [WWW Document]. URL http://www.conafor.gob.mx/apoyos/index.php/inicio/todos/2013 (accessed 11.24.13).
Conti, G., Díaz, S., 2013. Plant functional diversity and carbon storage – an empirical test in semi-arid forest ecosystems. J. Ecol. 101, 18–28. doi:10.1111/1365-2745.12012
Corbera, E., Brown, K., 2008. Building Institutions to Trade Ecosystem Services: Marketing Forest Carbon in Mexico. World Dev. 36, 1956–1979. doi:10.1016/j.worlddev.2007.09.010
Corbera, E., Kosoy, N., Tuna, M.M., 2007. Equity implications of marketing ecosystem services in protected areas and rural communities: Case studies from Meso-America. Glob. Environ. Change-Hum. Policy Dimens. 17, 365–380. doi:10.1016/j.gloenvcha.2006.12.005
Corbera, E., Soberanis, C.G., Brown, K., 2009. Institutional dimensions of Payments for Ecosystem Services: An analysis of Mexico's carbon forestry programme. Ecol. Econ. 68, 743–761. doi:10.1016/j.ecolecon.2008.06.008
Cordingley, J.E., Newton, A.C., Rose, R.J., Clarke, R.T., Bullock, J.M., 2015. Habitat Fragmentation Intensifies Trade-Offs between Biodiversity and Ecosystem Services in a Heathland Ecosystem in Southern England. PLoS ONE 10, e0130004. doi:10.1371/journal.pone.0130004
Cranford, M., Mourato, S., 2011. Community conservation and a two-stage approach to payments for ecosystem services. Ecol. Econ. 71, 89–98. doi:10.1016/j.ecolecon.2011.08.007
Crawford, S.E.S., Ostrom, E., 1995. A Grammar of Institutions. Am. Polit. Sci. Rev. 89, 582–600. doi:10.2307/2082975
Dagang, A.B.K., Nair, P.K.R., 2003. Silvopastoral research and adoption in Central America: recent findings and recommendations for future directions. Agrofor. Syst. 59, 149–155. doi:10.1023/A:1026394019808
Davies, T.E., Fazey, I.R.A., Cresswell, W., Pettorelli, N., 2014. Missing the trees for the wood: Why we are failing to see success in pro-poor conservation. Anim. Conserv. 17, 303–312. doi:10.1111/acv.12094
Davis, L.E., North, D.C., 1971. Institutional Change and American Economic Growth. Cambridge University Press, Cambridge.
Davison, A.C., Hinkley, D.V., 1997. Bootstrap Methods and Their Applications, Cambridge Series in Statistical and Probabilistic Mathematics. Cambridge University Press, Cambridge, UK.
DeFries, R., Karanth, K.K., Pareeth, S., 2010. Interactions between protected areas and their surroundings in human-dominated tropical landscapes. Biol. Conserv., Conservation and management in human-dominated landscapes: Case studies from India 143, 2870–2880. doi:10.1016/j.biocon.2010.02.010
de Koning, F., Aguiñaga, M., Bravo, M., Chiu, M., Lascano, M., Lozada, T., Suarez, L., 2011. Bridging the gap between forest conservation and poverty alleviation: the Ecuadorian Socio Bosque program. Environ. Sci. Policy 14, 531–542. doi:10.1016/j.envsci.2011.04.007
DeMay, S.M., Roon, D.A., Rachlow, J.L., Cisneros, R., 2014. Selective foraging on bromeliads by Andean bears in the Ecuadorian páramo. Ursus 25, 139–147. doi:10.2192/URSUS-D-14-00022.1
de Ponti, T., Rijk, B., van Ittersum, M.K., 2012. The crop yield gap between organic and conventional agriculture. Agric. Syst. 108, 1–9. doi:10.1016/j.agsy.2011.12.004

Derissen, S., Latacz-Lohmann, U., 2013. What are PES? A review of definitions and an extension. Ecosyst. Serv., Payments for Ecosystem Services and Their Institutional Dimensions: Analyzing the Diversity of Existing PES Approaches in Developing and Industrialized Countries 6, 12–15. doi:10.1016/j.ecoser.2013.02.002
Díaz, S., Fargione, J., Chapin, F.S., III, Tilman, D., 2006. Biodiversity Loss Threatens Human Well-Being. PLoS Biol 4, e277. doi:10.1371/journal.pbio.0040277
Dıaz, S., Tilman, D., Fargione, J., Chapin III, F.S., Dirzo, R., Ktzberber, T., others, 2005. Biodiversity regulation of ecosystem services, in: Hassan, R., Scholes, R., Ash, N. (Eds.), Ecosystems and Human Well-Being: Current State and Trends, Volume 1, The Millennium Ecosystem Assessment Series. Island Press, Washington DC, USA, pp. 297–329.
Duke, E.A., Goldstein, J.H., Teel, T.L., Finchum, R., Huber-Stearns, H., Pitty, J., Rodríguez P., G.B., Rodríguez, S., Sánchez, L.O., 2014. Payments for ecosystem services and landowner interest: Informing program design trade-offs in Western Panama. Ecol. Econ. 103, 44–55. doi:10.1016/j.ecolecon.2014.04.013
Dunn, W.W., Lynch, A.M., Morgan, P., 1990. Benefit-cost analysis of fuelwood management using native alder in Ecuador. Agrofor. Syst. 11, 125–139. doi:10.1007/BF00838725
Echavarria, M., Vogel, J., Albán, M., Meneses, F., 2004. The impacts of payments for watershed services in Ecuador. Emerging lessons from Pimampiro and Cuenca (Working Paper), Environmental Economics Programme. International Institute for Environment and Development, London, UK.
Engel, S., Pagiola, S., Wunder, S., 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecol. Econ. 65, 663–674. doi:10.1016/j.ecolecon.2008.03.011
Engle Warnick, J.C., Escobal, J., Laszlo, S.C., 2011. Ambiguity Aversion and Portfolio Choice in Small-Scale Peruvian Farming. BE J. Econ. Anal. Policy 11, 1–56.
Epstein, G., Vogt, J.M., Mincey, S.K., Cox, M., Fischer, B., 2013. Missing ecology: integrating ecological perspectives with the social-ecological system framework. Int. J. Commons 7, 432–453.
Estrada, A.G., Damián, M.A.M., Cano, M.A., 2006. Análisis de riesgo, portafolios óptimos y diversificación en la agricultura. Agrociencia 40, 409–417.
FAO, 2014. Payment for ecosystem services for forests (PES) and forest financing. Twenty-second session of the FAO Commitee on Forestry.
FAO, 2012. Payments for environmental services (PES). PES: One tool in the box. [WWW Document]. URL http://www.fao.org/es/esa/PESAL/aboutPES5.html (accessed 4.26.12).
FAO, 2011. Payments for ecosystem services and food security. Food and Agriculture Organization of the United Nations, Rome, Italy.
FAO, 2009a. Pago por servicios ambientales en áreas protegidas en América Latina: programa FAO/OAPN; fortalecimiento del manejo sostenible de los recursos naturales en las áreas protegidas de América Latina. Food and Agriculture Organization of the United Nations, Rome, Italy.
FAO, 2009b. The market for organic and fair-trade coffee. Study prepared in the framework of FAO project GCP/RAF/404/GER. Food and Agriculture Organization of the United Nations, Rome, Italy.
FAO, 2004. Global forest resources assessment update 2005. Terms and definitions (Final version) (Working Paper No. 83/E), Forest Resources Assessment Programme. FAO, Forestestry Department, Rome, Italy.
FAO, 2003. Forests and the forestry sector: Ecuador [WWW Document]. URL http://www.fao.org/forestry/country/57478/en/ecu/ (accessed 7.6.13).
FAOSTAT, 2014. Trade data [WWW Document]. URL http://faostat.fao.org/desktopdefault.aspx?pageid=342&lang=en&country=58 (accessed 5.11.14).
Farley, J., 2008. The role of prices in conserving critical natural capital. Conserv. Biol. J. Soc. Conserv. Biol. 22, 1399–1408. doi:10.1111/j.1523-1739.2008.01090.x

Farley, J., Costanza, R., 2010. Payments for ecosystem services: From local to global. Ecol. Econ. 69, 2060–2068. doi:10.1016/j.ecolecon.2010.06.010
Farley, K.A., Anderson, W.G., Bremer, L.L., Harden, C.P., 2011. Compensation for ecosystem services: an evaluation of efforts to achieve conservation and development in Ecuadorian páramo grasslands. Environ. Conserv. 38, 393–405. doi:10.1017/S037689291100049X
Farley, K.A., Jobbágy, E.G., Jackson, R.B., 2005. Effects of afforestation on water yield: a global synthesis with implications for policy. Glob. Change Biol. 11, 1565–1576. doi:10.1111/j.1365-2486.2005.01011.x
Farley, K.A., Kelly, E.F., Hofstede, R.G., 2004. Soil Organic Carbon and Water Retention after Conversion of Grasslands to Pine Plantations in the Ecuadorian Andes. Ecosystems 7, 729–739. doi:http://dx.doi.org/10.1007/s10021-004-0047-5
Fauzi, A., Anna, Z., n.d. The complexity of the institution of payment for environmental services: A case study of two Indonesian PES schemes. Ecosyst. Serv. doi:10.1016/j.ecoser.2013.07.003
Fehse, J., Hofstede, R., Aguirre, N., Paladines, C., Kooijman, A., Sevink, J., 2002. High altitude tropical secondary forests: a competitive carbon sink? For. Ecol. Manag. 163, 9–25. doi:10.1016/S0378-1127(01)00535-7
Ferraro, P.J., 2011. The Future of Payments for Environmental Services. Conserv. Biol. 25, 1134–1138. doi:10.1111/j.1523-1739.2011.01791.x
Ferraro, P.J., 2008. Asymmetric information and contract design for payments for environmental services. Ecol. Econ. 65, 810–821. doi:10.1016/j.ecolecon.2007.07.029
Ferraro, P.J., Kiss, A., 2002. Direct Payments to Conserve Biodiversity. Science 298, 1718–1719. doi:10.1126/science.1078104
Ferraro, P.J., Pattanayak, S.K., 2006. Money for Nothing? A Call for Empirical Evaluation of Biodiversity Conservation Investments. PLoS Biol 4, e105. doi:10.1371/journal.pbio.0040105
Ferraro, P.J., Simpson, R.D., 2002. The Cost-Effectiveness of Conservation Payments. Land Econ. 78, 339–353. doi:10.2307/3146894
Ferraro, P.J., Uchida, T., Conrad, J.M., 2005. Price Premiums for Eco-friendly Commodities: Are "Green" Markets the Best Way to Protect Endangered Ecosystems? Environ. Resour. Econ. 32, 419–438. doi:10.1007/s10640-005-7962-6
Fisher, B., Kulindwa, K., Mwanyoka, I., Turner, R.K., Burgess, N.D., 2010. Common pool resource management and PES: Lessons and constraints for water PES in Tanzania. Ecol. Econ., Special Section - Payments for Environmental Services: Reconciling Theory and Practice 69, 1253–1261. doi:10.1016/j.ecolecon.2009.11.008
Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., Balzer, C., Bennett, E.M., Carpenter, S.R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., Zaks, D.P.M., 2011. Solutions for a cultivated planet. Nature 478, 337–342. doi:10.1038/nature10452
FONAFIFO, 2015. Modalidades Pago de Servicios Ambientales [WWW Document]. FONAFIFO. URL http://www.fonafifo.go.cr/psa/modalidades_psa.html (accessed 6.29.15).
FONAG, 2014. FONAG [WWW Document]. URL http://www.fonag.org.ec/ (accessed 10.24.14).
FORAGUA, 2014. G.A.D. Municipal Loja [WWW Document]. FORAGUA. URL http://www.foragua.org/?q=node/81 (accessed 11.14.14).
Frazee, S.R., Cowling, R.M., Pressey, R.L., Turpie, J.K., Lindenberg, N., 2003. Estimating the costs of conserving a biodiversity hotspot: a case-study of the Cape Floristic Region, South Africa. Biol. Conserv., Conservation Planning in the Cape Floristic Region 112, 275–290. doi:10.1016/S0006-3207(02)00400-7
Fu, B.-J., Su, C.-H., Wei, Y.-P., Willett, I.R., Lü, Y.-H., Liu, G.-H., 2010. Double counting in ecosystem services valuation: causes and countermeasures. Ecol. Res. 26, 1–14. doi:10.1007/s11284-010-0766-3

Fu, B., Liu, Y., Lü, Y., He, C., Zeng, Y., Wu, B., 2011. Assessing the soil erosion control service of ecosystems change in the Loess Plateau of China. Ecol. Complex., Special Section: Complexity of Coupled Human and Natural Systems 8, 284–293. doi:10.1016/j.ecocom.2011.07.003
Gabre-Madhin, E.Z., 2001. The role of intermediaries in enhancing market efficiency in the Ethiopian grain market. Agric. Econ. 25, 311–320. doi:10.1111/j.1574-0862.2001.tb00211.x
Gadd, M.E., 2005. Conservation outside of parks: attitudes of local people in Laikipia, Kenya. Environ. Conserv. null, 50–63. doi:10.1017/S0376892905001918
Gardner, T.A., Barlow, J., Chazdon, R., Ewers, R.M., Harvey, C.A., Peres, C.A., Sodhi, N.S., 2009. Prospects for tropical forest biodiversity in a human-modified world. Ecol. Lett. 12, 561–582. doi:10.1111/j.1461-0248.2009.01294.x
Gasson, R., 1973. Goals and Values of Farmers. J. Agric. Econ. 24, 521–542. doi:10.1111/j.1477-9552.1973.tb00952.x
Gauvin, C., Uchida, E., Rozelle, S., Xu, J., Zhan, J., 2010. Cost-Effectiveness of Payments for Ecosystem Services with Dual Goals of Environment and Poverty Alleviation. Environ. Manage. 45, 488–501. doi:10.1007/s00267-009-9321-9
GEF, 2010. Payment for Ecosystem Services. Global Environmental Facility, Washington DC, USA.
GEF, 2006. The role of local benefits in global environmental programs (No. 30), Evaluation Report. Global Environment Facility Evaluation Office, Washington DC, USA.
Ghimire, K.B., Ghimire, K.B., Pimbert, M.P., 1997. Social Change and Conservation: Environmental Politics and Impacts of National Parks and Protected Areas. Earthscan, London, UK.
Global Ecolabelling Network, 2012. What is ecolabelling? [WWW Document]. URL http://www.globalecolabelling.net/what_is_ecolabelling/index.htm (accessed 4.15.12).
Goldman-Benner, R.L., Benitez, S., Boucher, T., Calvache, A., Daily, G., Kareiva, P., Kroeger, T., Ramos, A., 2012. Water funds and payments for ecosystem services: practice learns from theory and theory can learn from practice. Oryx 46, 55–63. doi:10.1017/S0030605311001050
Goldman, R.L., Benitez, S., Calvache, A., Davidson, S., Ennaanay, D., McKenzie, E., Tallis, H., 2010. Linking People and Nature through Watershed Conservation in the East Cauca Valley, Colombia. TEEB case study. Available at: TEEBweb.org.
Gomez-Baggethun, E., Ruiz Perez, M., 2011. Economic valuation and the commodification of ecosystem services. Prog. Phys. Geogr. 35, 613–628. doi:10.1177/0309133311421708
Gómez-Delgado, F., Roupsard, O., le Maire, G., Taugourdeau, S., Pérez, A., van Oijen, M., Vaast, P., Rapidel, B., Harmand, J.M., Voltz, M., Bonnefond, J.M., Imbach, P., Moussa, R., 2011. Modelling the hydrological behaviour of a coffee agroforestry basin in Costa Rica. Hydrol Earth Syst Sci 15, 369–392. doi:10.5194/hess-15-369-2011
Gómez-Limón, J.A., Arriaza, M., Riesgo, L., 2003. An MCDM analysis of agricultural risk aversion. Eur. J. Oper. Res. 151, 569–585. doi:10.1016/S0377-2217(02)00625-2
Gomiero, T., Pimentel, D., Paoletti, M.G., 2011. Environmental Impact of Different Agricultural Management Practices: Conventional vs. Organic Agriculture. Crit. Rev. Plant Sci. 30, 95–124. doi:10.1080/07352689.2011.554355
Gordillo, F., 2013. Mecanismo financiero de los GAD´s Municipales de la Región 7 del Sur del Ecuador para la conservación de las fuentes de agua.
Greene, W.H., 2007. LIMDEP Version 9. Student Reference Guide. Econometric Software, Inc., New York, USA.
Greiner, R., Stanley, O., 2013. More than money for conservation: Exploring social co-benefits from PES schemes. Land Use Policy 31, 4–10. doi:10.1016/j.landusepol.2011.11.012
Grieg-Gran, M., Porras, I., Wunder, S., 2005. How can market mechanisms for forest environmental services help the poor? Preliminary lessons from Latin America. World Dev. 33, 1511–1527. doi:10.1016/j.worlddev.2005.05.002
Günter, S., Gonzalez, P., Álvarez, G., Aguirre, N., Palomeque, X., Haubrich, F., Weber, M., 2009. Determinants for successful reforestation of abandoned pastures in the Andes: Soil conditions and vegetation cover. For. Ecol. Manag. 258, 81–91. doi:10.1016/j.foreco.2009.03.042

Hagedorn, K., 2008. Particular requirements for institutional analysis in nature-related sectors. Eur. Rev. Agric. Econ. 35, 357–384. doi:10.1093/erae/jbn019
Hajjar, R., Jarvis, D.I., Gemmill-Herren, B., 2008. The utility of crop genetic diversity in maintaining ecosystem services. Agric. Ecosyst. Environ. 123, 261–270. doi:10.1016/j.agee.2007.08.003
Hansen, A.J., Davis, C.R., Piekielek, N., Gross, J., Theobald, D.M., Goetz, S., Melton, F., DeFries, R., 2011. Delineating the Ecosystems Containing Protected Areas for Monitoring and Management. BioScience 61, 363–373. doi:10.1525/bio.2011.61.5.5
Harden, C.P., Hartsig, J., Farley, K.A., Lee, J., Bremer, L.L., 2013. Effects of Land-Use Change on Water in Andean Páramo Grassland Soils. Ann. Assoc. Am. Geogr. 103, 375–384. doi:10.1080/00045608.2013.754655
Harrison, P.A., Berry, P.M., Simpson, G., Haslett, J.R., Blicharska, M., Bucur, M., Dunford, R., Egoh, B., Garcia-Llorente, M., Geamănă, N., Geertsema, W., Lommelen, E., Meiresonne, L., Turkelboom, F., 2014. Linkages between biodiversity attributes and ecosystem services: A systematic review. Ecosyst. Serv. 9, 191–203. doi:10.1016/j.ecoser.2014.05.006
Harvey, C.A., Haber, W.A., 1998. Remnant trees and the conservation of biodiversity in Costa Rican pastures. Agrofor. Syst. 44, 37–68. doi:10.1023/A:1006122211692
Harvey, C.A., Komar, O., Chazdon, R., Ferguson, B.G., Finegan, B., Griffith, D.M., Martínez-Ramos, M., Morales, H., Nigh, R., Soto-Pinto, L., Van Breugel, M., Wishnie, M., 2008. Integrating Agricultural Landscapes with Biodiversity Conservation in the Mesoamerican Hotspot. Conserv. Biol. 22, 8–15. doi:10.1111/j.1523-1739.2007.00863.x
Harvey, C.A., Medina, A., Sánchez, D.M., Vílchez, S., Hernández, B., Saenz, J.C., Maes, J.M., Casanoves, F., Sinclair, F.L., 2006. Patterns of Animal Diversity in Different Forms of Tree Cover in Agricultural Landscapes. Ecol. Appl. 16, 1986–1999. doi:10.2307/40061768
Harvey, C.A., Villalobos, J.A.G., 2007. Agroforestry systems conserve species-rich but modified assemblages of tropical birds and bats. Biodivers. Conserv. 16, 2257–2292. doi:10.1007/s10531-007-9194-2
Harvey, C.A., Villanueva, C., Villacís, J., Chacón, M., Muñoz, D., López, M., Ibrahim, M., Gómez, R., Taylor, R., Martinez, J., Navas, A., Saenz, J., Sánchez, D., Medina, A., Vilchez, S., Hernández, B., Perez, A., Ruiz, F., López, F., Lang, I., Sinclair, F.L., 2005. Contribution of live fences to the ecological integrity of agricultural landscapes. Agric. Ecosyst. Environ. 111, 200–230. doi:10.1016/j.agee.2005.06.011
Hatanaka, N., Wright, W., Loyn, R.H., Mac Nally, R., 2011. "Ecologically complex carbon"– linking biodiversity values, carbon storage and habitat structure in some austral temperate forests. Glob. Ecol. Biogeogr. 20, 260–271. doi:10.1111/j.1466-8238.2010.00591.x
Haug, I., Wubet, T., Weiß, M., Aguirre, N., Weber, M., Günter, S., Kottke, I., 2010. Species-rich but distinct arbuscular mycorrhizal communities in reforestation plots on degraded pastures and in neighboring pristine tropical mountain rain forest. Trop. Ecol. 51, 125–148.
Hausman, J., McFadden, D., 1984. Specification tests for the multinomial logit model. Econometrica 52, 1219–1240.
Hayes, T.M., 2012. Payment for ecosystem services, sustained behavioural change, and adaptive management: peasant perspectives in the Colombian Andes. Environ. Conserv. 39, 144–153. doi:http://dx.doi.org/10.1017/S0376892912000045
Hefting, M.M., Bobbink, R., de Caluwe, H., 2003. Nitrous oxide emission and denitrification in chronically nitrate-loaded riparian buffer zones. J. Environ. Qual. 32, 1194–1203.
Heidkamp, P., Hanink, D.M., Cromley, R.G., 2008. A land use model of the effects of eco-labeling in coffee markets. Ann. Reg. Sci. 42, 725–746. doi:10.1007/s00168-007-0176-9
Hein, L., van Koppen, K., de Groot, R.S., van Ierland, E.C., 2006. Spatial scales, stakeholders and the valuation of ecosystem services. Ecol. Econ. 57, 209–228. doi:10.1016/j.ecolecon.2005.04.005
Heithaus, M.R., Frid, A., Wirsing, A.J., Worm, B., 2008. Predicting ecological consequences of marine top predator declines. Trends Ecol. Evol. 23, 202–210. doi:10.1016/j.tree.2008.01.003

Hejnowicz, A.P., Raffaelli, D.G., Rudd, M.A., White, P.C.L., 2014. Evaluating the outcomes of payments for ecosystem services programmes using a capital asset framework. Ecosyst. Serv. doi:10.1016/j.ecoser.2014.05.001

Hensher, D.A., Rose, J.M., Greene, W.H., 2005. Applied Choice Analysis. A Primer. Cambridge University Press, Cambridge, UK.

Hernández-Hernández, R.M., Ramírez, E., Castro, I., Cano, S., 2008. Cambios en indicadores de calidad de suelos de ladera reforestados con pinos (Pinus caribaea) y eucaliptos (Eucalyptus robusta). Agrociencia 42, 253–266.

Hildebrandt, P., Knoke, T., 2011. Investment decisions under uncertainty—A methodological review on forest science studies. For. Policy Econ. 13, 1–15. doi:10.1016/j.forpol.2010.09.001

Hinkel, J., Bots, P.W.G., Schlüter, M., 2014. Enhancing the Ostrom social-ecological system framework through formalization. Ecol. Soc. 19. doi:10.5751/ES-06475-190351

Hoehn, P., Tscharntke, T., Tylianakis, J.M., Steffan-Dewenter, I., 2008. Functional group diversity of bee pollinators increases crop yield. Proc. R. Soc. Lond. B Biol. Sci. 275, 2283–2291. doi:10.1098/rspb.2008.0405

Hofstede, R.G.M., Groenendijk, J.P., Coppus, R., Fehse, J.C., Sevink, J., 2002. Impact of Pine Plantations on Soils and Vegetation in the Ecuadorian High Andes. Mt. Res. Dev. 22, 159–167. doi:10.1659/0276-4741(2002)022[0159:IOPPOS]2.0.CO;2

Holloway, G., Nicholson, C., Delgado, C., Staal, S., Ehui, S., 2000. Agroindustrialization through institutional innovation Transaction costs, cooperatives and milk-market development in the east-African highlands. Agric. Econ. 23, 279–288. doi:10.1111/j.1574-0862.2000.tb00279.x

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setälä, H., Symstad, A.J., Vandermeer, J., Wardle, D.A., 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecol. Monogr. 75, 3–35. doi:10.1890/04-0922

Horne, P., 2006. Forest owners’ acceptance of incentive based policy instruments in forest biodiversity conservation-A choice experiment based approach. Silva Fenn. 40, 169.

Howe, C., Suich, H., Vira, B., Mace, G.M., 2014. Creating win-wins from trade-offs? Ecosystem services for human well-being: A meta-analysis of ecosystem service trade-offs and synergies in the real world. Glob. Environ. Change 28, 263–275. doi:10.1016/j.gloenvcha.2014.07.005

Ibelings, B.W., Portielje, R., Lammens, E.H.R.R., Noordhuis, R., Berg, M.S. van den, Joosse, W., Meijer, M.L., 2007. Resilience of Alternative Stable States during the Recovery of Shallow Lakes from Eutrophication: Lake Veluwe as a Case Study. Ecosystems 10, 4–16. doi:10.1007/s10021-006-9009-4

Ibrahim, M., Villanueva, C., Casasola, F., Rojas, J., 2006. Sistemas silvopastoriles como una herramienta para el mejoramiento de la productividad y restauración de la integridad ecológica de paisajes ganaderos [WWW Document]. Pastos Forrajes. URL http://www.redalyc.org/resumen.oa?id=269121676004 (accessed 7.16.14).

ICRAF, 1993. Report of the First External Programme and Management Review of the International Centre for Research in Agroforestry (ICRAF). TAC SECRETARIAT and FAO.

IMF, 2011. World Economic Outlook September 2011. Slowing Growth, Rising Risks, World Economic and Financial Surveys. International Monetary Fund, Washington DC, USA.

INEC, 2014. Visualizador de Estadísticas Agropecuarias del Ecuador [WWW Document]. Inst. Nac. Estad. URL http://www.inec.gob.ec/estadisticas/?option=com_content&view=article&id=103&Itemid=75 (accessed 2.11.14).

INEC, 2012a. Provincias por Cantones (PDF) Actualizada al 31-12-2012 [WWW Document]. Inst. Nac. Estad. URL http://www.inec.gob.ec/estadisticas/?option=com_content&view=article&id=358 (accessed 11.13.14).

INEC, 2012b. Encuesta de Superficie y Producción Agropecuaria Continua ESPAC-2012.

INEC, 2012c. Encuesta Nacional de Ingresos y Gastos de los Hogares Urbanos y Rurales [WWW Document]. Inst. Nac. Estad. Censos. URL http://www.ecuadorencifras.gob.ec/encuesta-nacional-de-ingresos-y-gastos-de-los-hogares-urbanos-y-rurales/ (accessed 4.6.15).
INEC, 2012d. Ecuador - Índice de Precios al Consumidor 2012, Base [WWW Document]. URL http://anda.inec.gob.ec/anda/index.php/catalog/152/reports (accessed 5.10.14).
INEC, 2011a. Ecuador en Cifras [WWW Document]. Inst. Nac. Estad. URL http://www.ecuadorencifras.gob.ec/resultados/ (accessed 10.28.14).
INEC, 2011b. Últimos resultados de pobreza, desigualdad y mercado laboral en el Ecuador.
INEC, 2011c. Datos Estadísticos Agropecuarios. Resumen Ejecutivo.
Ingram, J.C., Wilkie, D., Clements, T., McNab, R.B., Nelson, F., Baur, E.H., Sachedina, H.T., Peterson, D.D., Foley, C.A.H., 2014. Evidence of Payments for Ecosystem Services as a mechanism for supporting biodiversity conservation and rural livelihoods. Ecosyst. Serv. 7, 10–21. doi:10.1016/j.ecoser.2013.12.003
Iñiguez–Armijos, C., Leiva, A., Frede, H., Hampel, H., Breuer, L., 2014. Deforestation and Benthic Indicators: How Much Vegetation Cover Is Needed to Sustain Healthy Andean Streams? PLoS ONE 9, e105869. doi:10.1371/journal.pone.0105869
Jack, B.K., Kousky, C., Sims, K.R.E., 2008. Designing payments for ecosystem services: Lessons from previous experience with incentive-based mechanisms. Proc. Natl. Acad. Sci. U. S. A. 105, 9465–9470. doi:10.1073/pnas.0705503104
Jack, B.K., Leimona, B., Ferraro, P.J., 2009. A Revealed Preference Approach to Estimating Supply Curves for Ecosystem Services: Use of Auctions to Set Payments for Soil Erosion Control in Indonesia. Conserv. Biol. 23, 359–367. doi:10.1111/j.1523-1739.2008.01086.x
Jaeck, M., Lifran, R., 2014. Farmers' Preferences for Production Practices: A Choice Experiment Study in the Rhone River Delta. J. Agric. Econ. 65, 112–130. doi:10.1111/1477-9552.12018
Jakoby, O., Quaas, M.F., Müller, B., Baumgärtner, S., Frank, K., 2014. How do individual farmers' objectives influence the evaluation of rangeland management strategies under a variable climate? J. Appl. Ecol. 51, 483–493. doi:10.1111/1365-2664.12216
Jordan, A., Wurzel, R.K.W., Zito, A., 2005. The Rise of "New" Policy Instruments in Comparative Perspective: Has Governance Eclipsed Government? Polit. Stud. 53, 477–496. doi:10.1111/j.1467-9248.2005.00540.x
Josse, C., 2001. La biodiversidad del Ecuador : informe 2000. Ministry of the Environment of Ecuador; EcoCiencia and UICN, Quito, Ecuador.
Just, R.E., Pope, R.D. (Eds.), 2002. A Comprehensive Assessment of the Role of Risk in U.S. Agriculture. Springer US, Boston, MA.
Juutinen, A., Mantymaa, E., Ollikainen, M., 2013. Landowners' conservation motives and the size of information rents in environmental bidding systems. J. For. Econ. 19, 128–148. doi:10.1016/j.jfe.2012.12.001
Kaczan, D., Swallow, B.M., Adamowicz, W.L. (Vic), 2013. Designing a payments for ecosystem services (PES) program to reduce deforestation in Tanzania: An assessment of payment approaches. Ecol. Econ. 95, 20–30. doi:10.1016/j.ecolecon.2013.07.011
Kauffman, C.M., 2013. Financing watershed conservation: Lessons from Ecuador's evolving water trust funds. Agric. Water Manag. doi:10.1016/j.agwat.2013.09.013
Keating, P.L., 2008. The Floristic Composition and Biogeographical Significance of a Megadiverse Páramo Site in the Southern Ecuadorian Andes. J. Torrey Bot. Soc. 135, 554–570.
Keating, P.L., 2000. Chronically Disturbed Páramo Vegetation at a Site in Southern Ecuador. J. Torrey Bot. Soc. 127, 162–171. doi:10.2307/3088693
Keese, J., Mastin, T., Yun, D., 2007. Identifying and Assessing Tropical Montane Forests on the Eastern Flank of the Ecuadorian Andes. J. Lat. Am. Geogr. 6, 63–84.
Kelly, P., Huo, X., 2013. Do farmers or governments make better land conservation choices? Evidence from China's Sloping Land Conversion Program. J. For. Econ. 19, 32–60. doi:10.1016/j.jfe.2012.08.002

Kessler, M., Kluge, J., 2008. Diversity and endemism in tropical montane forests—from patterns to processes, in: Gradstein, S.R., Homeier, J., Gansert, D. (Eds.), The Tropical Mountain Forest - Patterns and Processes in a Biodiversity Hotspot, Biodiversity and Ecology Series. Göttingen Centre for Biodiversity and Ecology, Univ. Verlag Göttingen, Göttingen, Germany, pp. 35–50.
Kinzig, A.P., Perrings, C., Chapin, F.S., Polasky, S., Smith, V.K., Tilman, D., Turner, B.L., 2011. Paying for Ecosystem Services-Promise and Peril. Science 334, 603–604. doi:10.1126/science.1210297
Klein, A.-M., Vaissière, B.E., Cane, J.H., Steffan-Dewenter, I., Cunningham, S.A., Kremen, C., Tscharntke, T., 2007. Importance of pollinators in changing landscapes for world crops. Proc. R. Soc. Lond. B Biol. Sci. 274, 303–313. doi:10.1098/rspb.2006.3721
Knoke, T., 2008. Mixed forests and finance — Methodological approaches. Ecol. Econ. 65, 590–601. doi:10.1016/j.ecolecon.2007.08.009
Knoke, T., Bendix, J., Pohle, P., Hamer, U., Hildebrandt, P., Roos, K., Gerique, A., Sandoval, M.L., Breuer, L., Tischer, A., Silva, B., Calvas, B., Aguirre, N., Castro, L.M., Windhorst, D., Weber, M., Stimm, B., Günter, S., Palomeque, X., Mora, J., Mosandl, R., Beck, E., 2014. Afforestation or intense pasturing improve the ecological and economic value of abandoned tropical farmlands. Nat. Commun. 5, 5612. doi:10.1038/ncomms6612
Knoke, T., Calvas, B., Aguirre, N., Román-Cuesta, R.M., Günter, S., Stimm, B., Weber, M., Mosandl, R., 2009a. Can tropical farmers reconcile subsistence needs with forest conservation? Front. Ecol. Environ. 7, 548–554. doi:10.1890/080131
Knoke, T., Hildebrandt, P., Klein, D., Mujica, R., Moog, M., Mosandl, R., 2008. Financial compensation and uncertainty: using mean-variance rule and stochastic dominance to derive conservation payments for secondary forests. Can. J. For. Res. 38, 3033–3046. doi:10.1139/X08-137
Knoke, T., Steinbeis, O.-E., Bösch, M., Román-Cuesta, R.M., Burkhardt, T., 2011. Cost-effective compensation to avoid carbon emissions from forest loss: An approach to consider price–quantity effects and risk-aversion. Ecol. Econ. 70, 1139–1153. doi:10.1016/j.ecolecon.2011.01.007
Knoke, T., Stimm, B., Ammer, C., Moog, M., 2005. Mixed forests reconsidered: A forest economics contribution on an ecological concept. For. Ecol. Manag. 213, 102–116. doi:10.1016/j.foreco.2005.03.043
Knoke, T., Weber, M., Barkmann, J., Pohle, P., Calvas, B., Medina, C., Aguirre, N., Günter, S., Stimm, B., Mosandl, R., von Walter, F., Maza, B., Gerique, A., 2009b. Effectiveness and distributional impacts of payments for reduced carbon emissions from. ERDKUNDE 63, 365–384. doi:10.3112/erdkunde.2009.04.06
Kolinjivadi, V., Adamowski, J., Kosoy, N., 2014. Recasting payments for ecosystem services (PES) in water resource management: A novel institutional approach. Ecosyst. Serv. 10, 144–154. doi:10.1016/j.ecoser.2014.08.008
Kollmair, M., Rasul, G., 2010. Addressing equity and poverty concerns in payments for environmental services. Mt. Forum Bull. 10, 12–14.
Korff, G.D., 2008. Reviving the Forgotten American Dream. Penn State Law Rev. 113, 417–460.
Kosoy, N., Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. Ecol. Econ. 69, 1228–1236. doi:10.1016/j.ecolecon.2009.11.002
Kosoy, N., Martinez-Tuna, M., Muradian, R., Martinez-Alier, J., 2007. Payments for environmental services in watersheds: Insights from a comparative study of three cases in Central America. Ecol. Econ. 61, 446–455. doi:10.1016/j.ecolecon.2006.03.016
Krause, T., Collen, W., Nicholas, K.A., 2013. Evaluating Safeguards in a Conservation Incentive Program: Participation, Consent, and Benefit Sharing in Indigenous Communities of the Ecuadorian Amazon. Ecol. Soc. 18. doi:10.5751/ES-05733-180401
Krause, T., Loft, L., 2013. Benefit Distribution and Equity in Ecuador's Socio Bosque Program. Soc. Nat. Resour. 26, 1170–1184. doi:10.1080/08941920.2013.797529
Krause, T., Nielsen, T.D., 2014. The legitimacy of incentive-based conservation and a critical account of social safeguards. Environ. Sci. Policy 41, 44–51. doi:10.1016/j.envsci.2014.04.015

Krause, T., Zambonino, H., 2013. More than just trees – animal species diversity and participatory forest monitoring in the Ecuadorian Amazon. Int. J. Biodivers. Sci. Ecosyst. Serv. Manag. 9, 225–238. doi:10.1080/21513732.2013.822930

Kremen, C., Miles, A., 2012. Ecosystem Services in Biologically Diversified versus Conventional Farming Systems: Benefits, Externalities, and Trade-Offs. Ecol. Soc. 17. doi:10.5751/ES-05035-170440

Kremen, C., Niles, J.O., Dalton, M.G., Daily, G.C., Ehrlich, P.R., Fay, J.P., Grewal, D., Guillery, R.P., 2000. Economic Incentives for Rain Forest Conservation Across Scales. Science 288, 1828–1832. doi:10.1126/science.288.5472.1828

Kroeger, T., Casey, F., 2007a. An assessment of market-based approaches to providing ecosystem services on agricultural lands. Ecol. Econ., Special Section - Ecosystem Services and Agriculture Ecosystem Services and Agriculture 64, 321–332. doi:10.1016/j.ecolecon.2007.07.021

Kroeger, T., Casey, F., 2007b. An assessment of market-based approaches to providing ecosystem services on agricultural lands. Ecol. Econ. 64, 321–332. doi:10.1016/j.ecolecon.2007.07.021

Kurttila, M., Pykäläinen, J., Leskinen, P., 2006. Defining the forest landowner's utility-loss compensative subsidy level for a biodiversity object. Eur. J. For. Res. 125, 67–78. doi:http://dx.doi.org/10.1007/s10342-005-0079-1

Lambin, E.F., Turner, B.L., Geist, H.J., Agbola, S.B., Angelsen, A., Bruce, J.W., Coomes, O.T., Dirzo, R., Fischer, G., Folke, C., George, P.S., Homewwod, K., Imbernon, J., Leemans, R., Li, X., Moran, E.F., Mortimore, M., Ramakrishnan, P.S., Richards, J.F., Skanes, H., Steffen, W., Stone, G.D., Svedin, U., Veldkamp, T.A., Vogel, C., Xu, J., 2001. The causes of land-use and land-cover change : Moving beyond the myths. Glob. Environ. Change 11, 261–269.

Lancaster, K.J., 1966. A new approach to consumer theory. J. Polit. Econ. 74, 132–157.

Landell-Mills, N., Porras, I.T., 2002. Silver bullet or fools' gold? A global review of markets for forest environmental services and their impact on the poor. International Institute for Environment and Development, London, UK.

Lant, C.L., Ruhl, J.B., Kraft, S.E., 2008. The Tragedy of Ecosystem Services. BioScience 58, 969–974. doi:10.1641/B581010

Laurans, Y., Lemenager, T., Aoubid, S., 2012. Payments for Ecosystem Services. From Theory to Practice – What are the Prospects for Developping Countries? AFD, Paris, France.

Lavorel, S., Grigulis, K., 2012. How fundamental plant functional trait relationships scale-up to trade-offs and synergies in ecosystem services. J. Ecol. 100, 128–140. doi:10.1111/j.1365-2745.2011.01914.x

Lavrakas, P., 2008. Encyclopedia of Survey Research Methods. SAGE Publications, Inc., 2455 Teller Road, Thousand Oaks California 91320 United States.

Layton, D.F., Siikamäki, J., 2009. Payments for Ecosystem Services Programs: Predicting Landowner Enrollment and Opportunity Cost Using a Beta-Binomial Model. Environ. Resour. Econ. 44, 415–439. doi:10.1007/s10640-009-9293-5

Le Coq, J.-F., Soto, G., González Hernández, C., 2011. PES and Eco-Label. A comparative analysis of their limits and opportunities to foster environmental services provision, in: Rapidel, B., Le Coq, J.F., Beer, J., DeClerck, F. (Eds.), Ecosystem Services from Agriculture and Agroforestry : Measurement and Payment. Earthscan Publications, London, UK, pp. 237–264.

León, F., Moncayo, M.C., Renner, I., Prem, I., 2010. Compensación por servicios ecosistémicos: Guía de monitoreo de impactos. Las microcuencas Mishiquiyacu, Rumiyacu y Almendra de San Martín, Perú.

Le, Q.B., Park, S.J., Vlek, P.L.G., Cremers, A.B., 2008. Land-Use Dynamic Simulator (LUDAS): A multi-agent system model for simulating spatio-temporal dynamics of coupled human–landscape system. I. Structure and theoretical specification. Ecol. Inform. 3, 135–153. doi:10.1016/j.ecoinf.2008.04.003

Letourneau, D.K., Jedlicka, J.A., Bothwell, S.G., Moreno, C.R., 2009. Effects of Natural Enemy Biodiversity on the Suppression of Arthropod Herbivores in Terrestrial Ecosystems. Annu. Rev. Ecol. Evol. Syst. 40, 573–592. doi:10.1146/annurev.ecolsys.110308.120320

Lewandrowski, J., Darwin, R.., Tsigas, M., Raneses, A., 1999. Estimating costs of protecting global ecosystem diversity. Ecol. Econ. 29, 111–125. doi:10.1016/S0921-8009(98)00058-5

Lewin, B., Giovannucci, D., Varangis, P., 2004. Coffee markets : New paradigms in global supply and demand (Working Paper No. 3), Agriculture and Rural Development discussion paper. World Bank, Washington DC, USA.

Libecap, G.D., 2005. State Regulation of Open-Access, Common-Pool Resources, in: Ménard, C., Shirley, M.M. (Eds.), Handbook of New Institutional Economics. Springer US, pp. 545–572.

Lienhoop, N., Brouwer, R., 2015. Agri-environmental policy valuation: Farmers' contract design preferences for afforestation schemes. Land Use Policy 42, 568–577. doi:10.1016/j.landusepol.2014.09.017

Lindblom, C.E., 2001. The Market System. What It Is, How It Works, and What To Make of It. Yale University Press, New Haven, Connecticut, USA.

Loreau, M., Hector, A., 2001. Partitioning selection and complementarity in biodiversity experiments. Nature 412, 72–76. doi:10.1038/35083573

Louviere, J.J., Hensher, D.A., Swait, J.D., 2000. Stated Choice Methods. Analysis and Applications. Cambridge University Press, Cambridge, UK.

Luce, R.D., 1959. Individual Choice Behavior: A Theoretical Analysis. John Wiley & Sons, New York, USA.

Lurie, S., Bennett, D.E., Duncan, S., Gosnell, H., Hunter, M.L., Morzillo, A.T., Moseley, C., Nielsen-Pincus, M., Parker, R., White, E.M., 2013. PES marketplace development at the local scale: The Eugene Water and Electric Board as a local watershed services marketplace driver. Ecosyst. Serv., Payments for Ecosystem Services and Their Institutional Dimensions: Analyzing the Diversity of Existing PES Approaches in Developing and Industrialized Countries 6, 93–103. doi:10.1016/j.ecoser.2013.09.005

MA, 2005a. Millennium Ecosystem Assessment: Ecosystems and Human Well-Being, Biodiversity Synthesis. World Resources Institute, Washington DC, USA.

MA, 2005b. Millennium Ecosystem Assessment. Ecosystems and Human Well-being: Synthesis. Island Press, Washington DC, USA.

MA, 2005c. Ecosystems and human well-being. Health Synthesis. A Report of the Millennium Ecosystem Assessment. World Health Organization, France.

Madsen, J.E., Øllgaard, B., 1994. Floristic composition, structure, and dynamics of an upper montane rain forest in Southern Ecuador. Nord. J. Bot. 14, 403–423. doi:10.1111/j.1756-1051.1994.tb00626.x

Mahanty, S., Suich, H., Tacconi, L., 2013. Access and benefits in payments for environmental services and implications for REDD+: Lessons from seven PES schemes. Land Use Policy, Themed Issue 1-Guest Editor Romy GreinerThemed Issue 2- Guest Editor Davide Viaggi 31, 38–47. doi:10.1016/j.landusepol.2011.10.009

Makarieva, A.M., Gorshkov, V.G., Li, B.-L., 2006. Conservation of water cycle on land via restoration of natural closed-canopy forests: implications for regional landscape planning. Ecol. Res. 21, 897–906. doi:10.1007/s11284-006-0036-6

Malawska, A., Topping, C.J., Nielsen, H.Ø., 2014. Why do we need to integrate farmer decision making and wildlife models for policy evaluation? Land Use Policy 38, 732–740. doi:10.1016/j.landusepol.2013.10.025

Manzano Díaz, I., 2010. Pagos por Servicios Ambientales. Caso Ecuador-REDD 74. Rev. Derecho Económico 2, 255–264.

Marenya, P., Smith, V.H., Nkonya, E., 2014. Relative Preferences for Soil Conservation Incentives among Smallholder Farmers: Evidence from Malawi. Am. J. Agric. Econ. 96, 690–710. doi:10.1093/ajae/aat117

Markandya, A., Perelet, R., Mason, P., Taylor, T., 2001. Dictionary of Environmental Economics. Taylor & Francis, London, UK.
Markowitz, H., 1952. Portfolio Selection*. J. Finance 7, 77–91. doi:10.1111/j.1540-6261.1952.tb01525.x
Martinez, R., Green, K.M., DeWan, A., 2013. Establishing reciprocal agreements for water and biodiversity conservation through a social marketing campaign in Quanda Watershed, Peru. Conserv. Evid. 10, 42–47.
Martino, D., 2001. Buffer Zones Around Protected Areas: A Brief Literature Review. Electron. Green J. 1.
Martin-Ortega, J., Ojea, E., Roux, C., 2013. Payments for Water Ecosystem Services in Latin America: A literature review and conceptual model. Ecosyst. Serv., Payments for Ecosystem Services and Their Institutional Dimensions: Analyzing the Diversity of Existing PES Approaches in Developing and Industrialized Countries 6, 122–132. doi:10.1016/j.ecoser.2013.09.008
Matta, J., Alavalapati, J., Tanner, G., 2007. A framework for developing marked-based policies to further biodiversity on non-industrial private forests (NIPF). For. Policy Econ., Economic perspectives and analyses of multiple forest values and sustainable forest management 9, 779–788. doi:10.1016/j.forpol.2006.03.008
Matta, J.R., Alavalapati, J.R.R., Mercer, D.E., 2009. Incentives for Biodiversity Conservation Beyond the Best Management Practices: Are Forestland Owners Interested? Land Econ. 85, 132–143. doi:10.3368/le.85.1.132
Matzdorf, B., Sattler, C., Engel, S., 2013. Institutional frameworks and governance structures of PES schemes. For. Policy Econ., 1. International developments in the administration of publicly-funded forest research: Challenges and opportunities 2. Payments for ecosystem services and their institutional dimensions: Institutional frameworks and governance structures of PES schemes 37, 57–64. doi:10.1016/j.forpol.2013.10.002
McDonnell, P., 2008. Optimal Portfolio Modeling: Models to Maximize Returns and Control Risk in Excel and R. John Wiley & Sons.
McFadden, D., 1974. Conditional logit analysis of qualitative choice behavior, in: Zarembka, P. (Ed.), Frontiers in Econometrics. Academic Press, pp. 105–142.
McGinnis, M.D., Ostrom, E., 2014. Social-ecological system framework: initial changes and continuing challenges. Ecol. Soc. 19. doi:10.5751/ES-06387-190230
McGinnis, M.D., Ostrom, E., 2012. SES framework: initial changes and continuing challenges (Working Paper No. W11-6).
McGinnis, M.D., Ostrom, E., 2010. IAD and SES Dynamic Flows: Introducing the Program in Institutional Analysis of Social-Ecological Systems (PIASES) Framework. Presented at the 13th Economics of Infrastructures Conference, Delft, Netherlands.
Ménard, C., 2012. Hybrid Modes of Organization. Alliances, Joint Ventures, Networks, and Other "Strange" Animals, in: Gibbons, R., Roberts, J. (Eds.), The Handbook of Organizational Economics. Princeton University Press, New Jersey, USA, pp. 1066–1108.
Ménard, C., 2005. New institutions for governing the agri-food industry. Eur. Rev. Agric. Econ. 32, 421–440. doi:10.1093/eurrag/jbi013
Ménard, C., 2004. The economics of hybrid organizations. J. Institutional Theor. Econ. JITE 160, 345–376.
Ménard, C., 1996a. Why organizations matter: A journey away from the fairy tale. Atl. Econ. J. 24, 281–300. doi:10.1007/BF02298432
Ménard, C., 1996b. Inside the Black Box: The Variety of Hierarchical Forms, in: Groenewegen, J. (Ed.), Transaction Cost Economics and Beyond, Recent Economic Thought Series. Springer Netherlands, The Netherlands, pp. 149–170.
Milder, J.C., Scherr, S.J., Bracer, C., 2010. Trends and Future Potential of Payment for Ecosystem Services to Alleviate Rural Poverty in Developing Countries. Ecol. Soc. 15.

Miles, L., Kapos, V., 2008. Reducing Greenhouse Gas Emissions from Deforestation and Forest Degradation: Global Land-Use Implications. Science 320, 1454–1455. doi:10.1126/science.1155358
Milne, S., Adams, B., 2012. Market Masquerades: Uncovering the Politics of Community-level Payments for Environmental Services in Cambodia. Dev. Change 43, 133–158. doi:10.1111/j.1467-7660.2011.01748.x
Milne, S., Niesten, E., 2009. Direct payments for biodiversity conservation in developing countries: practical insights for design and implementation. Oryx 43, 530–541. doi:http://dx.doi.org/10.1017/S0030605309990330
Ministry of Agriculture, Ecuador, 2013a. Resolución No. 002.
Ministry of Agriculture, Ecuador, 2013b. Acuerdo No. 502.
Ministry of Agriculture, Ecuador, 2012. III CENSO NACIONAL AGROPECUARIO [WWW Document]. SINAGAP. URL http://sinagap.agricultura.gob.ec/censo-nacional-agropecuario (accessed 5.7.14).
Ministry of the Environment, Ecuador, 2014a. Plan Nacional de Restauración Forestal 2014-2017.
Ministry of the Environment, Ecuador, 2014b. Acuerdo N. 211.
Ministry of the Environment, Ecuador, 2012a. Sistematización de Socio Bosque.
Ministry of the Environment, Ecuador, 2012b. Línea Base de Deforestación del Ecuador Continental.
Ministry of the Environment, Ecuador, 2012c. Manual Operativo Unificado Proyecto Socio Bosque.
Ministry of the Environment, Ecuador, 2012d. Acuerdo N. 130 Manual Operativo.
Ministry of the Environment, Ecuador, 2010. Socio Bosque: Conceptualización, Implementación y Avances al Segundo Año de Implementación.
Miranda, M., Dieperink, C., Glasbergen, P., 2007. Voluntary agreements in watershed protection. Experiences from Costa Rica. Environ. Dev. Sustain. 9, 1–19. doi:10.1007/s10668-005-9000-9
Molnar, A., Scherr, S.J., Khare, A., 2004. Who conserves the world's forests? Community-driven strategies to protect forests and respect rights, Policy Brief. Forest Trends, Ecoagriculture Partners, Washington DC, USA.
Molua, E.L., 2005. The economics of tropical agroforestry systems: the case of agroforestry farms in Cameroon. For. Policy Econ. 7, 199–211. doi:10.1016/S1389-9341(03)00032-7
Mönkkönen, M., Reunanen, P., Kotiaho, J.S., Juutinen, A., Tikkanen, O., Kouki, J., 2011. Cost-effective strategies to conserve boreal forest biodiversity and long-term landscape-level maintenance of habitats. Eur. J. For. Res. 130, 717–727. doi:http://dx.doi.org/10.1007/s10342-010-0461-5
Montagnini, F., Nair, P.K.R., 2004. Carbon sequestration: An underexploited environmental benefit of. Agrofor. Syst. 61-62, 281–295. doi:10.1023/B:AGFO.0000029005.92691.79
Morandin, L.A., Winston, M.L., 2006. Pollinators provide economic incentive to preserve natural land in agroecosystems. Agric. Ecosyst. Environ. 116, 289–292. doi:10.1016/j.agee.2006.02.012
Morse, W.C., McLaughlin, W.J., Wulfhorst, J.D., Harvey, C., 2013. Social ecological complex adaptive systems: a framework for research on payments for ecosystem services. Urban Ecosyst. 16, 53–77. doi:10.1007/s11252-011-0178-3
Mosandl, R., Günter, S., Stimm, B., Weber, M., 2008. Ecuador Suffers the Highest Deforestation Rate in South America, in: Beck, E., Bendix, J., Kottke, I., Makeschin, F., Mosandl, R. (Eds.), Gradients in a Tropical Mountain Ecosystem of Ecuador, Ecological Studies. Springer Berlin Heidelberg, pp. 37–40.
Moscardi, E., Janvry, A. de, 1977. Attitudes toward Risk among Peasants: An Econometric Approach. Am. J. Agric. Econ. 59, 710–716. doi:10.2307/1239398
Municipality of Loja, 2007. Protección Ambiental de las Micro Cuencas en el Cantón Loja. Ordenanza Municipal 3.
Municipality of Pimampiro, 2013. La primera reforma a la ordenanza que establece la creación del fondo para el pago por servicios ambientales para la conservación de bosques y páramos con fines de regulación de agua.

Municipality of San Pedro del Norte, 2011. Ordenanza Municipal: Pagos por Servicios Ambientales Hidricos PSAH, en la Microcuenca “Paso de los Caballos”.
Muñoz Escobar, M., Hollaender, R., Pineda Weffer, C., 2013. Institutional durability of payments for watershed ecosystem services: Lessons from two case studies from Colombia and Germany. Ecosyst. Serv., Payments for Ecosystem Services and Their Institutional Dimensions: Analyzing the Diversity of Existing PES Approaches in Developing and Industrialized Countries 6, 46–53. doi:10.1016/j.ecoser.2013.04.004
Muñoz-Piña, C., Guevara, A., Torres, J.M., Braña, J., 2008. Paying for the hydrological services of Mexico’s forests: Analysis, negotiations and results. Ecol. Econ. 65, 725–736. doi:10.1016/j.ecolecon.2007.07.031
Muradian, R., 2013. Payments for Ecosystem Services as Incentives for Collective Action. Soc. Nat. Resour. 26, 1155–1169. doi:10.1080/08941920.2013.820816
Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. Ecol. Econ. 69, 1202–1208. doi:10.1016/j.ecolecon.2009.11.006
Muradian, R., Gómez-Baggethun, E., 2013. The Institutional Dimension of “Market-Based Instruments” for Governing Ecosystem Services: Introduction to the Special Issue. Soc. Nat. Resour. 26, 1113–1121. doi:10.1080/08941920.2013.829380
Muradian, R., Rival, L., 2012. Between markets and hierarchies: The challenge of governing ecosystem services. Ecosyst. Serv. 1, 93–100.
Murcia, C., 1997. Evaluation of Andean alder as a catalyst for the recovery of tropical cloud forests in Colombia. For. Ecol. Manag. 99, 163–170. doi:10.1016/S0378-1127(97)00202-8
Murgueitio, E., Cuellar, P., Ibrahim, M., Gobbi, J., Cuartas, C.A., Naranjo, J.F., Zapata, A., Mejía, C.E., Zuluaga, A.F., Cassasola, F., 2006. Adopción de sistemas agroforestales pecuarios. Adoption of agroforestry systems for animal production. Pastos Forrajes 29, 365–379.
Mutke, J., Barthlott, W., 2005. Patterns of vascular plant diversity at continental to global scales. Biol. Skr. 55, 521–531.
Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858. doi:10.1038/35002501
Naeem, S., Thompson, L.J., Lawler, S.P., Lawton, J.H., Woodfin, R.M., 1994. Declining biodiversity can alter the performance of ecosystems. Nature 368, 734–737. doi:10.1038/368734a0
Naughton-Treves, L., Grossberg, R., Treves, A., 2003. Paying for Tolerance: Rural Citizens’ Attitudes toward Wolf Depredation and Compensation. Conserv. Biol. 17, 1500–1511. doi:10.1111/j.1523-1739.2003.00060.x
Newmark, W.D., Leonard, N.L., Sariko, H.I., Gamassa, D.-G.M., 1993. Conservation attitudes of local people living adjacent to five protected areas in Tanzania. Biol. Conserv. 63, 177–183.
Ninan, K.N., Jyothis, S., Babu, P., Ramakrishnappa, V., 2007. The Economics of Biodiversity Conservation: Valuation in Tropical Forest Ecosystems. Earthscan, London, UK.
Nordén, A., 2014. Payment Types and Participation in Payment for Ecosystem Services Programs. Stated Preferences of Landowners (Working Paper No. 14-11), Discussion Paper Series. Environment for Development Center for Central America, Costa Rica.
Norgaard, R.B., 2010. Ecosystem services: From eye-opening metaphor to complexity blinder. Ecol. Econ. 69, 1219–1227. doi:10.1016/j.ecolecon.2009.11.009
North, D.C., 1991. Institutions. J. Econ. Perspect. 5, 97–112.
North, D.C., 1990. Institutions, Institutional Change and Economic Performance. Cambridge University Press, Cambridge, UK.
OECD, 2004. Handbook of Market Creation for Biodiversity. Organisation for Economic Co-operation and Development, Paris, France.
Ojea, E., Martin-Ortega, J., 2015. Understanding the economic value of water ecosystem services from tropical forests: A systematic review for South and Central America. J. For. Econ. 21, 97–106. doi:10.1016/j.jfe.2015.02.001

Olschewski, R., Benítez, P.C., 2005. Secondary forests as temporary carbon sinks? The economic impact of accounting methods on reforestation projects in the tropics. Ecol. Econ. 55, 380–394. doi:10.1016/j.ecolecon.2004.09.021
Olsen, N., Bishop, J., Anstee, S., 2011. Exploring ecosystem valuation to move towards net positive impact on biodiversity in the mining sector, IUCN and Rio Tinto Technical Series. International Union for Conservation of Nature, Gland, Switzerland.
Ormerod, S.J., 2003. Restoration in applied ecology: editor's introduction. J. Appl. Ecol. 40, 44–50. doi:10.1046/j.1365-2664.2003.00799.x
Ostrom, E., 2010. Institutional Analysis and Development: Elements of the Framework in historical Perspective, in: Crothers, C. (Ed.), Historical Developments and Theoretical Approaches in Sociology. UNESCO Encyclopedia of Life Support Systems.
Ostrom, E., 2009. A General Framework for Analyzing Sustainability of Social-Ecological Systems. Science 325, 419–422. doi:10.1126/science.1172133
Ostrom, E., 2007. A diagnostic approach for going beyond panaceas. Proc. Natl. Acad. Sci. 104, 15181–15187.
Ostrom, E., Cox, M., 2010. Moving beyond panaceas: a multi-tiered diagnostic approach for social-ecological analysis. Environ. Conserv. 37, 451–463. doi:10.1017/S0376892910000834
Pagiola, S., 2008. Payments for environmental services in Costa Rica. Ecol. Econ. 65, 712–724. doi:10.1016/j.ecolecon.2007.07.033
Pagiola, S., Agostini, P., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., Ramirez, E., Rosales, M., Ruiz, J.P., 2005a. Paying for Biodiversity conservation services - Experience in Colombia, Costa Rica, and Nicaragua. Mt. Res. Dev. 25, 206–211. doi:10.1659/0276-4741(2005)025[0206:PFBCS]2.0.CO;2
Pagiola, S., Arcenas, A., Platais, G., 2005b. Can Payments for Environmental Services Help Reduce Poverty? An Exploration of the Issues and the Evidence to Date from Latin America. World Dev. 33, 237–253. doi:10.1016/j.worlddev.2004.07.011
Pagiola, S., Platais, G., 2002. Payments for Environmental Services (Discussion Paper No. 3), Environmental Strategy Notes. World Bank, Washington DC, USA.
Pagiola, S., Ramirez, E., Gobbi, J., de Haan, C., Ibrahim, M., Murgueitio, E., Ruiz, J.P., 2007. Paying for the environmental services of silvopastoral practices in Nicaragua. Ecol. Econ. 64, 374–385. doi:10.1016/j.ecolecon.2007.04.014
Pagiola, S., Rios, A.R., Arcenas, A., 2010. Poor Household Participation in Payments for Environmental Services: Lessons from the Silvopastoral Project in Quindío, Colombia. Environ. Resour. Econ. 47, 371–394. doi:10.1007/s10640-010-9383-4
Pagiola, S., Rios, A.R., Arcenas, A., 2008. Can the poor participate in payments for environmental services? Lessons from the Silvopastoral Project in Nicaragua. Environ. Dev. Econ. 13, 299–325. doi:10.1017/S1355770X08004270
Palumbi, S.R., McLeod, K.L., Grünbaum, D., 2008. Ecosystems in Action: Lessons from Marine Ecology about Recovery, Resistance, and Reversibility. BioScience 58, 33–42. doi:10.1641/B580108
Pan, Y., Birdsey, R.A., Fang, J., Houghton, R., Kauppi, P.E., Kurz, W.A., Phillips, O.L., Shvidenko, A., Lewis, S.L., Canadell, J.G., Ciais, P., Jackson, R.B., Pacala, S.W., McGuire, A.D., Piao, S., Rautiainen, A., Sitch, S., Hayes, D., 2011. A Large and Persistent Carbon Sink in the World's Forests. Science 333, 988–993. doi:10.1126/science.1201609
Parker, C., Cranford, M., Oakes, N., Leggett, M., 2012. The Little Biodiversity Finance Book. A guide to proactive investment in natural capital (PINC)., 3rd ed. Global Canopy Programme, Oxford, UK.
Parkin, M., Powell, M., Matthews, K., 2011. Economics, 8th ed. Pearson Education, Oxford, UK.
Pattanayak, S.K., Mercer, D.E., Sills, E., Yang, J.-C., 2003. Taking stock of agroforestry adoption studies. Agrofor. Syst. 57, 173–186. doi:10.1023/A:1024809108210
Paul, S., Veldkamp, E., Flessa, H., 2008. Soil organic carbon in density fractions of tropical soils under forest – pasture – secondary forest land use changes. Eur. J. Soil Sci. 59, 359–371. doi:10.1111/j.1365-2389.2007.01010.x

Pearce, D., 2007. Do we really care about Biodiversity? Environ. Resour. Econ. 37, 313–333. doi:10.1007/s10640-007-9118-3
Pereira, S.N.C., 2010. Payment for Environmental Services in the Amazon Forest: How Can Conservation and Development Be Reconciled? J. Environ. Dev. 19, 171–190. doi:10.1177/1070496510368047
Persson, U.M., Alpízar, F., 2013. Conditional Cash Transfers and Payments for Environmental Services—A Conceptual Framework for Explaining and Judging Differences in Outcomes. World Dev. 43, 124–137. doi:10.1016/j.worlddev.2012.10.006
Peters, T., Diertl, K.-H., Gawlik, J., Rankl, M., Richter, M., 2010. Vascular Plant Diversity in Natural and Anthropogenic Ecosystems in the Andes of Southern Ecuador. Mt. Res. Dev. 30, 344–352. doi:10.1659/MRD-JOURNAL-D-10-00029.1
Petheram, L., Campbell, B.M., 2010. Listening to locals on payments for environmental services. J. Environ. Manage. 91, 1139–1149. doi:10.1016/j.jenvman.2010.01.002
Philpott, S.M., Arendt, W.J., Armbrecht, I., Bichier, P., Diestch, T.V., Gordon, C., Greenberg, R., Perfecto, I., Reynoso-Santos, R., Soto-Pinto, L., Tejeda-Cruz, C., Williams-Linera, G., Valenzuela, J., Zolotoff, J.M., 2008. Biodiversity Loss in Latin American Coffee Landscapes: Review of the Evidence on Ants, Birds, and Trees. Conserv. Biol. 22, 1093–1105. doi:10.1111/j.1523-1739.2008.01029.x
Philpott, S.M., Bichier, P., 2012. Effects of shade tree removal on birds in coffee agroecosystems in Chiapas, Mexico. Agric. Ecosyst. Environ. 149, 171–180. doi:10.1016/j.agee.2011.02.015
Philpott, S.M., Soong, O., Lowenstein, J.H., Pulido, A.L., Lopez, D.T., Flynn, D.F.B., DeClerck, F., 2009. Functional richness and ecosystem services: bird predation on arthropods in tropical agroecosystems. Ecol. Appl. 19, 1858–1867. doi:10.1890/08-1928.1
Pirard, R., 2012a. Market-based instruments for biodiversity and ecosystem services: A lexicon. Environ. Sci. Policy 19-20, 59–68. doi:10.1016/j.envsci.2012.02.001
Pirard, R., 2012b. Payments for Environmental Services (PES) in the public policy landscape: “Mandatory” spices in the Indonesian recipe. For. Policy Econ. 18, 23–29. doi:10.1016/j.forpol.2011.09.002
Pirard, R., Billé, R., Sembrés, T., 2010. Questioning the theory of Payments for Ecosystem Services (PES) in light of emerging experience and plausible developments. (Working Paper No. N°04/2010), SERIES. IDDRI, Paris, France.
Pirard, R., Lapeyre, R., 2014. Classifying market-based instruments for ecosystem services: A guide to the literature jungle. Ecosyst. Serv. doi:10.1016/j.ecoser.2014.06.005
Plurinational State of Bolivia, 2012. Ley N° 300. Ley Marco de la Madre Tierra y Desarrollo Integral para Vivir Bien.
Polak, G.G., Rogers, D.F., Sweeney, D.J., 2010. Risk management strategies via minimax portfolio optimization. Eur. J. Oper. Res. 207, 409–419. doi:10.1016/j.ejor.2010.04.025
Ponette-González, A.G., Weathers, K.C., Curran, L.M., 2010. Water inputs across a tropical montane landscape in Veracruz, Mexico: synergistic effects of land cover, rain and fog seasonality, and interannual precipitation variability. Glob. Change Biol. 16, 946–963. doi:10.1111/j.1365-2486.2009.01985.x
Porras, I., 2010. Fair and green? Social impacts of payments for environmental services in Costa Rica. International Institute for Environment and Development, London, UK.
Porras, I., Grieg-Gran, M., Neves, N., 2008. All that glitters. A review of payments for watershed services in developing countries. International Institute for Environment and Development, London, UK.
Power, A.G., 2010. Ecosystem services and agriculture: tradeoffs and synergies. Philos. Trans. R. Soc. B Biol. Sci. 365, 2959–2971. doi:10.1098/rstb.2010.0143
Preece, N.D., Crowley, G.M., Lawes, M.J., van Oosterzee, P., 2012. Comparing above-ground biomass among forest types in the Wet Tropics: Small stems and plantation types matter in carbon accounting. For. Ecol. Manag. 264, 228–237. doi:10.1016/j.foreco.2011.10.016
Prigent, J.-L., 2007. Portfolio Optimization and Performance Analysis. CRC Press.

Prins, H.H.T., Wind, J., 1993. Research for nature conservation in south-east Asia. Biol. Conserv. 63, 43–46. doi:10.1016/0006-3207(93)90071-8
Prokofieva, I., Gorriz, E., 2013. Institutional analysis of incentives for the provision of forest goods and services: An assessment of incentive schemes in Catalonia (north-east Spain). For. Policy Econ. 37, 104–114. doi:10.1016/j.forpol.2013.09.005
Putten, van I.E., Jennings, S.M., Louviere, J.J., Burgess, L.B., 2011. Tasmanian landowner preferences for conservation incentive programs: A latent class approach. J. Environ. Manage. 92, 2647–2656. doi:10.1016/j.jenvman.2011.06.002
Quintero, M., Wunder, S., Estrada, R.D., 2009. For services rendered? Modeling hydrology and livelihoods in Andean payments for environmental services schemes. For. Ecol. Manag. 258, 1871–1880. doi:10.1016/j.foreco.2009.04.032
Rabotyagov, S.S., Lin, S., 2013. Small forest landowner preferences for working forest conservation contract attributes: A case of Washington State, USA. J. For. Econ. 19, 307–330. doi:10.1016/j.jfe.2013.06.002
Rahbek, C., Bloch, H., Poulsen, M.K., Rasmussen, J.F., 1995. The avifauna of the Podocarpus National Park - the “Andean jewel in the crown” of Ecuador’s protected areas. Ornitol. Neotropical 6, 113–120.
Rametsteiner, E., 2002. The role of governments in forest certification—a normative analysis based on new institutional economics theories. For. Policy Econ. 4, 163–173. doi:10.1016/S1389-9341(02)00004-7
Rappole, J.H., King, D.I., Rivera, J.H.V., 2003. Coffee and Conservation. Conserv. Biol. 17, 334–336. doi:10.1046/j.1523-1739.2003.01548.x
Raudsepp-Hearne, C., Peterson, G.D., Tengö, M., Bennett, E.M., Holland, T., Benessaiah, K., MacDonald, G.K., Pfeifer, L., 2010. Untangling the Environmentalist’s Paradox: Why Is Human Well-being Increasing as Ecosystem Services Degrade? Bioscience 60, 576–589.
R Core Team, 2014. R: A language and environment for statistical computing.
Redondo-Brenes, A., 2009. An Assessment of a Fund for Watershed Conservation in the Southern Ecuadorian Andes. J. Sustain. For. 28, 755–776. doi:10.1080/10549810902922375
Redondo-Brenes, A., Welsh, K., 2010. Procuencas Project, Costa Rica, available at: TEEBweb.org.
Reeves, L.H., Lilieholm, R.J., 1993. Reducing financial risk in agroforestry planning: a case study in Costa Rica. Agrofor. Syst. 21, 169–175. doi:10.1007/BF00705228
Republic of Ecuador, 2008. Constitución Política del Ecuador.
Restrepo, C., Alvarez, N., 2006. Landslides and Their Contribution to Land-cover Change in the Mountains of Mexico and Central America1. Biotropica 38, 446–457. doi:10.1111/j.1744-7429.2006.00178.x
Rex, K., Kelm, D.H., Wiesner, K., Kunz, T.H., Voigt, C.C., 2008. Species richness and structure of three Neotropical bat assemblages. Biol. J. Linn. Soc. 94, 617–629. doi:10.1111/j.1095-8312.2008.01014.x
Rhoades, C.C., Eckert, G.E., Coleman, D.C., 2000. Soil carbon differences among forest, agriculture, and secondary vegetation in lower montane Ecuador. Ecol. Appl. 10, 497–505. doi:10.1890/1051-0761(2000)010[0497:SCDAFA]2.0.CO;2
Rice, J., 2007. Mathematical Statistics and Data Analysis, 3rd ed. Cengage Learning, USA.
Ricketts, T.H., Daily, G.C., Ehrlich, P.R., Michener, C.D., 2004. Economic value of tropical forest to coffee production. Proc. Natl. Acad. Sci. U. S. A. 101, 12579–12582. doi:10.1073/pnas.0405147101
Roa-García, M.C., Brown, S., Schreier, H., Lavkulich, L.M., 2011. The role of land use and soils in regulating water flow in small headwater catchments of the Andes. Water Resour. Res. 47, 1–12. doi:10.1029/2010WR009582
Robert, N., Stenger, A., 2013. Can payments solve the problem of undersupply of ecosystem services? For. Policy Econ., New Frontiers of Forest Economics 35, 83–91. doi:10.1016/j.forpol.2013.06.012

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J., Nykvist, B., de Wit, C.A., Hughes, T., van der Leeuw, S., Rodhe, H., Sörlin, S., Snyder, P.K., Costanza, R., Svedin, U., Falkenmark, M., Karlberg, L., Corell, R.W., Fabry, V.J., Hansen, J., Walker, B., Liverman, D., Richardson, K., Crutzen, P., Foley, J.A., 2009. A safe operating space for humanity. Nature 461, 472–475. doi:10.1038/461472a

Rodricks, S., 2013. Bundling of ecosystem services in agroforestry, Mexico. mainly based on Corbera, E., et al 2009.

Rodríguez de Francisco, J.C., Budds, J., Boelens, R., 2013. Payment for Environmental Services and Unequal Resource Control in Pimampiro, Ecuador. Soc. Nat. Resour. 26, 1217–1233. doi:10.1080/08941920.2013.825037

Rodríguez, L.C., Pascual, U., Muradian, R., Pazmino, N., Whitten, S., 2011. Towards a unified scheme for environmental and social protection: Learning from PES and CCT experiences in developing countries. Ecol. Econ. 70, 2163–2174. doi:10.1016/j.ecolecon.2011.06.019

Rojas, M., Aylward, B., 2003. What are we learning from experiences with markets for environmental services in Costa Rica? A review and critique of the literature. (Working Paper), Environmental Economics Programme. International Institute for Environment and Development, London, UK.

Rojas, M., Aylward, B., 2002. Cooperation between a small private hydropower producer and a conservation NGO for forest protection: The case of La Esperanza, Costa Rica, Land-Water Linkages in Rural Watersheds Case Study Series. FAO, Rome, Italy.

Rosa, H., Barry, D., Dimas, L., Kandel, S., 2004. Compensation for Environmental Services and Rural Communities: Lessons from the Americas (Working Paper No. wp96). Political Economy Research Institute, University of Massachusetts at Amherst.

Rosenbaum, E.F., 2000. What is a market? On the methodology of a contested concept. Rev. Soc. Econ. 58, 455–482.

Rosenberg, A., Bigford, T.E., Leathery, S., Hill, R.L., Bickers, K., 2000. Ecosystem approaches to fishery management through essential fish habitat. Bull. Mar. Sci. 66, 535–542.

Rudd, M.A., 2004. An institutional framework for designing and monitoring ecosystem-based fisheries management policy experiments. Ecol. Econ. 48, 109–124. doi:10.1016/j.ecolecon.2003.10.002

Ruto, E., Garrod, G., 2009. Investigating farmers' preferences for the design of agri-environment schemes: a choice experiment approach. J. Environ. Plan. Manag. 52, 631–647. doi:10.1080/09640560902958172

Sanchez-Azofeifa, G.A., Pfaff, A., Robalino, J.A., Boomhower, J.P., 2007. Costa Rica's payment for environmental services program: Intention, implementation, and impact. Conserv. Biol. 21, 1165–1173. doi:10.1111/j.1523-1739.2007-00751.x

Sattler, C., Trampnau, S., Schomers, S., Meyer, C., Matzdorf, B., 2013. Multi-classification of payments for ecosystem services: How do classification characteristics relate to overall PES success? Ecosyst. Serv., Payments for Ecosystem Services and Their Institutional Dimensions: Analyzing the Diversity of Existing PES Approaches in Developing and Industrialized Countries 6, 31–45. doi:10.1016/j.ecoser.2013.09.007

Schaafsma, M., Morse-Jones, S., Posen, P., Swetnam, R.D., Balmford, A., Bateman, I.J., Burgess, N.D., Chamshama, S.A.O., Fisher, B., Freeman, T., Geofrey, V., Green, R.E., Hepelwa, A.S., Hernández-Sirvent, A., Hess, S., Kajembe, G.C., Kayharara, G., Kilonzo, M., Kulindwa, K., Lund, J.F., Madoffe, S.S., Mbwambo, L., Meilby, H., Ngaga, Y.M., Theilade, I., Treue, T., van Beukering, P., Vyamana, V.G., Turner, R.K., 2011. The importance of local forest benefits: Valuation of non-timber forest products in the Eastern Arc Mountains in Tanzania (Working Paper No. 2011-05), CSERGE working paper. CSERGE, University of East Anglia, UK.

Scheidel, A., 2013. Flows, funds and the complexity of deprivation: Using concepts from ecological economics for the study of poverty. Ecol. Econ., Sustainable Urbanisation: A resilient future 86, 28–36. doi:10.1016/j.ecolecon.2012.10.019

Scherr, S.J., Bennett, M.T., Loughney, M., Canby, K., 2006. Developing future ecosystem service payments in China: Lessons learned from international experience. A report prepared for the China Council for International Cooperation on Environment and Development (CCICED) Taskforce on Ecocompensation. Forest Trends, Washington DC, USA.
Scherr, S.J., White, A., Khare, A., International Tropical Timber Organization, 2004. For services rendered: the current status and future potential of markets for the ecosystem services provided by tropical forests. International Tropical Timber Organization, Yokohama, Japan.
Scheufele, G., Bennett, J., 2013. Effective Implementation of Payments for Environmental Services in Lao PDR (No. 1), Research Report. Crawford School of Public Policy, Australian National University, Canberra, Australia.
Schlüter, A., Madrigal, R., 2012. The SES Framework in a Marine Setting: Methodological Lessons. Ration. Mark. Morals J. 148–167.
Schomers, S., Matzdorf, B., 2013. Payments for ecosystem services: A review and comparison of developing and industrialized countries. Ecosyst. Serv. 6, 16–30. doi:10.1016/j.ecoser.2013.01.002
Secchi, S., Gassman, P.W., Williams, J.R., Babcock, B.A., 2009. Corn-Based Ethanol Production and Environmental Quality: A Case of Iowa and the Conservation Reserve Program. Environ. Manage. 44, 732–744. doi:10.1007/s00267-009-9365-x
Seufert, V., Ramankutty, N., Foley, J.A., 2012. Comparing the yields of organic and conventional agriculture. Nature 485, 229–232. doi:10.1038/nature11069
Shapiro-Garza, E., 2013. Contesting the market-based nature of Mexico's national payments for ecosystem services programs: Four sites of articulation and hybridization. Geoforum 46, 5–15. doi:10.1016/j.geoforum.2012.11.018
Sharpe, W.F., 1994. The Sharpe Ratio. J. Portf. Manag. 21, 49–58. doi:10.3905/jpm.1994.409501
Shelley, B.G., 2011. What should we call instruments commonly known as payments for environmental services? A review of the literature and a proposal. Ann. N. Y. Acad. Sci. 1219, 209–225. doi:10.1111/j.1749-6632.2010.05941.x
Siebert, R., Berger, G., Lorenz, J., Pfeffer, H., 2010. Assessing German farmers' attitudes regarding nature conservation set-aside in regions dominated by arable farming. J. Nat. Conserv. 18, 327–337. doi:10.1016/j.jnc.2010.01.006
Siebert, R., Toogood, M., Knierim, A., 2006. Factors Affecting European Farmers' Participation in Biodiversity Policies. Sociol. Rural. 46, 318–340. doi:10.1111/j.1467-9523.2006.00420.x
Siles, P., Harmand, J.-M., Vaast, P., 2010. Effects of Inga densiflora on the microclimate of coffee (Coffea arabica L.) and overall biomass under optimal growing conditions in Costa Rica. Agrofor. Syst. 78, 269–286. doi:10.1007/s10457-009-9241-y
Smith, J., Scherr, S.J., 2002. Forest carbon and local livelihoods: assessment of opportunities and policy recommendations (No. 37), CIFOR Occasional Paper. Centre for International Forestry Research, Bogor Barat Indonesia.
Sommerville, M.M., Jones, J.P., Milner-Gulland, E.J., 2009. A revised conceptual framework for payments for environmental services. Ecol. Soc. 14, 34.
Sommerville, M.M., Milner-Gulland, E.J., Jones, J.P.G., 2011. The challenge of monitoring biodiversity in payment for environmental service interventions. Biol. Conserv. 144, 2832–2841. doi:10.1016/j.biocon.2011.07.036
Soto-Pinto, L., Anzueto, M., Mendoza, J., Ferrer, G.J., Jong, B. de, 2010. Carbon sequestration through agroforestry in indigenous communities of Chiapas, Mexico. Agrofor. Syst. 78, 39–51. doi:10.1007/s10457-009-9247-5
Southgate, D., Wunder, S., 2009. Paying for Watershed Services in Latin America: A Review of Current Initiatives. J. Sustain. For. 28, 497–524. doi:10.1080/10549810902794493
Swait, J., 1994. A structural equation model of latent segmentation and product choice for cross-sectional revealed preference choice data. J. Retail. Consum. Serv. 1, 77–89. doi:10.1016/0969-6989(94)90002-7

Swallow, B., Meinzen-Dick, R., 2009. Payment for Environmental Services: Interactions with Property Rights and Collective Action, in: Dr, V.B., Dr, M.P. (Eds.), Institutions and Sustainability. Springer Netherlands, pp. 243–265.

Swallow, B.M., Kallesoe, M.F., Iftikhar, U.A., van Noordwijk, M., Bracer, C., Scherr, S.J., Raju, K.V., Poats, S.V., Duraiappah, A.K., Ochieng, B.O., Mallee, H., Rumley, R., 2009. Compensation and Rewards for Environmental Services in the Developing World: Framing Pan-Tropical Analysis and Comparison. Ecol. Soc. 14.

Tacconi, L., 2012. Redefining payments for environmental services. Ecol. Econ. 73, 29–36. doi:10.1016/j.ecolecon.2011.09.028

Tacconi, L., 2000. Biodiversity & Ecological Economics: Participation, Values and Resource Management. Earthscan Publications, London, UK.

Tanaka, Y., Munro, A., 2014. Regional Variation in Risk and Time Preferences: Evidence from a Large-scale Field Experiment in Rural Uganda. J. Afr. Econ. 23, 151–187. doi:10.1093/jae/ejt020

TEEB, 2010a. The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations. Earthscan, London and Washington.

TEEB, 2010b. The economics of ecosystems and biodiversity for local and regional policy makers.

ten Brink, P., 2011. Rewarding benefits through payments and markets., in: Brink, P. ten (Ed.), The Economics of Ecosystems and Biodiversity in National and International Policy Making. Routledge, Oxford, UK, p. 528.

Tesfaye, A., Brouwer, R., 2012. Testing participation constraints in contract design for sustainable soil conservation in Ethiopia. Ecol. Econ. 73, 168–178. doi:10.1016/j.ecolecon.2011.10.017

Thomas, H., Nisbet, T.R., 2007. An assessment of the impact of floodplain woodland on flood flows. Water Environ. J. 21, 114–126. doi:10.1111/j.1747-6593.2006.00056.x

Tietenberg, T., 2006. Tradable Permits in Principle and Practice. Penn State Environ. Law Rev. 14, 251–281.

Tilman, D., Cassman, K.G., Matson, P.A., Naylor, R., Polasky, S., 2002. Agricultural sustainability and intensive production practices. Nature 418, 671–677. doi:10.1038/nature01014

To, P.X., Dressler, W.H., Mahanty, S., Pham, T.T., Zingerli, C., 2012. The Prospects for Payment for Ecosystem Services (PES) in Vietnam: A Look at Three Payment Schemes. Hum. Ecol. 40, 237–249. doi:10.1007/s10745-012-9480-9

Torres-Lezama, A., Vilanova, E., Ramírez-Angulo, H., Alciaturi, G., 2012. Socioeconomic and Environmental Basis for the Development of Small Scale Forestry in a Highly Degraded Watershed in the Venezuelan Andes. Small-Scale For. 11, 321–337. doi:10.1007/s11842-011-9186-7

Turlach, B.A., Weingessel, A., 2013. quadprog: Functions to solve Quadratic Programming Problems. R package version 1.5-5. http://CRAN.R-project.org/package=quadprog.

Turner, R.K., Daily, G.C., 2008. The Ecosystem Services Framework and Natural Capital Conservation. Environ. Resour. Econ. 39, 25–35. doi:10.1007/s10640-007-9176-6

Turner, R.K., Paavola, J., Cooper, P., Farber, S., Jessamy, V., Georgiou, S., 2003. Valuing nature: lessons learned and future research directions. Ecol. Econ. 46, 493–510. doi:10.1016/S0921-8009(03)00189-7

Turpie, J.K., Marais, C., Blignaut, J.N., 2008. The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in South Africa. Ecol. Econ., Payments for Environmental Services in Developing and Developed Countries 65, 788–798. doi:10.1016/j.ecolecon.2007.12.024

UN, 1992. Convention on Biological Diversity.

UNEP, 2008. Biodiversity and Ecosystem Services. Bloom or Bust? UNEP Finance Initiative, Biodiversity & Ecosystem Services Work Stream.

UNSTATS, 2014. United Nations Statistics Division. Households and families. [WWW Document]. Demogr. Soc. Stat. URL http://unstats.un.org/unsd/demographic/sconcerns/fam/fammethods.htm (accessed 12.16.14).

Valkila, J., 2009. Fair Trade organic coffee production in Nicaragua — Sustainable development or a poverty trap? Ecol. Econ. 68, 3018–3025. doi:10.1016/j.ecolecon.2009.07.002
van den Berg, A.E., Vlek, C.A.J., Coeterier, J.F., 1998. Group differences in the aesthetic evaluation of nature development plans: A multilevel approach. J. Environ. Psychol. 18, 141–157. doi:10.1006/jevp.1998.0080
Vandermeer, J., Perfecto, I., Philpott, S., 2010. Ecological Complexity and Pest Control in Organic Coffee Production: Uncovering an Autonomous Ecosystem Service. BioScience 60, 527–537. doi:10.1525/bio.2010.60.7.8
Van Hecken, G., Bastiaensen, J., 2010. Payments for ecosystem services: justified or not? A political view. Environ. Sci. Policy 13, 785–792. doi:10.1016/j.envsci.2010.09.006
Van Huylenbroeck, G., Vuylsteke, A., Verbeke, W., 2009. Public Good Markets: The Possible Role of Hybrid Governance Structures in Institutions for Sustainability, in: Beckmann, V., Padmanabhan, M. (Eds.), Institutions and Sustainability. Springer Netherlands, pp. 175–191.
van Noordwijk, M., Leimona, B., 2010. Principles for Fairness and Efficiency in Enhancing Environmental Services in Asia: Payments, Compensation, or Co-Investment? Ecol. Soc. 15.
van Noordwijk, M., Leimona, B., Emerton, L., Tomich, T.P., Velarde, S.J., Kallesoe, M., Sekher, M., Swallow, B., 2007. Criteria and indicators for environmental service compensation and reward mechanisms: realistic, voluntary, conditional and pro-poor: CES Scoping Study Issue Paper no. 2 (Working Paper No. 37), ICRAF Working Paper. World Agroforestry Centre (ICRAF), Nairobi, Kenya.
Vatn, A., 2015. Markets in environmental governance. From theory to practice. Ecol. Econ. 117, 225–233. doi:doi:10.1016/j.ecolecon.2014.07.017
Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecol. Econ. 69, 1245–1252. doi:10.1016/j.ecolecon.2009.11.018
Vedeld, P., Angelsen, A., Bojö, J., Sjaastad, E., Kobugabe Berg, G., 2007. Forest environmental incomes and the rural poor. For. Policy Econ. 9, 869–879. doi:10.1016/j.forpol.2006.05.008
Vignola, R., Koellner, T., Scholz, R.W., McDaniels, T.L., 2010. Decision-making by farmers regarding ecosystem services: Factors affecting soil conservation efforts in Costa Rica. Land Use Policy 27, 1132–1142. doi:10.1016/j.landusepol.2010.03.003
Villamor, G.B., van Noordwijk, M., Djanibekov, U., Chiong-Javier, M.E., Catacutan, D., 2014. Gender differences in land-use decisions: shaping multifunctional landscapes? Curr. Opin. Environ. Sustain., Sustainability challenges 6, 128–133. doi:10.1016/j.cosust.2013.11.015
Wallace, G.N., Barborak, J.R., MacFarland, C., 2003. Land-use planning and regulation in and around protected areas: A study of best practices and capacity building needs in Mexico and Central America. Presented at the 5th World Parks Conference, Durban, South Africa.
Wang, G., Fang, Q., Zhang, L., Chen, W., Chen, Z., Hong, H., 2010. Valuing the effects of hydropower development on watershed ecosystem services: Case studies in the Jiulong River Watershed, Fujian Province, China. Estuar. Coast. Shelf Sci. 86, 363–368. doi:10.1016/j.ecss.2009.03.022
WCMC, 2000. Global biodiversity: Earth's living resources in the 21st century. Cambridge Press, Cambridge, UK.
Webber, L., 2009. Diagnóstico y plan de monitoreo de la calidad del agua en las áreas de interés hídrico de los cantones Celica, Pindal, Puyango y Macará. Loja - Ecuador. Naturaleza y Cultura International, Loja, Ecuador.
Whaley, L., Weatherhead, E.K., 2014. An Integrated Approach to Analyzing (Adaptive) Comanagement Using the "Politicized" IAD Framework. Ecol. Soc. 19. doi:10.5751/ES-06177-190110
Williamson, O.E., 2010. Transaction Cost Economics: The Natural Progression. Am. Econ. Rev. 100, 673–90. doi:10.1257/aer.100.3.673
Williamson, O.E., 2002. The Theory of the Firm as Governance Structure: From Choice to Contract. J. Econ. Perspect. 16, 171–195. doi:10.1257/089533002760278776
Williamson, O.E., 2000. The new institutional economics: taking stock, looking ahead. J. Econ. Lit. 595–613.

Williamson, O.E., 1991. Comparative Economic Organization: The Analysis of Discrete Structural Alternatives. Adm. Sci. Q. 36, 269–296. doi:10.2307/2393356

Williamson, O.E., 1985. The Economic Institutions of Capitalism. Free Press, New York, USA.

Windhorst, D., Silva, B., Peters, T., Meyer, H., Thies, B., Bendix, J., Frede, H.-G., Breuer, L., 2013. Impacts of Local Land-Use Change on Climate and Hydrology, in: Bendix, J., Beck, E., Bräuning, A., Makeschin, F., Mosandl, R., Scheu, S., Wilcke, W. (Eds.), Ecosystem Services, Biodiversity and Environmental Change in a Tropical Mountain Ecosystem of South Ecuador, Ecological Studies. Springer Berlin Heidelberg, pp. 275–286.

Wunder, S., 2015. Revisiting the concept of payments for environmental services. Ecol. Econ. 117, 234–243. doi:10.1016/j.ecolecon.2014.08.016

Wunder, S., 2013. When payments for environmental services will work for conservation. Conserv. Lett. 6, 230–237. doi:10.1111/conl.12034

Wunder, S., 2012. Of PES and other animals. Oryx 46, 1–2. doi:http://dx.doi.org/10.1017/S0030605311002195

Wunder, S., 2008. Necessary Conditions for Ecosystem Service Payments. Presented at the Economics and Conservation in the Tropics: A Strategic Dialogue, Resources for the Future, San Francisco, CA, USA., p. 10.

Wunder, S., 2007. The Efficiency of Payments for Environmental Services in Tropical Conservation. Conserv. Biol. 21, 48–58. doi:10.1111/j.1523-1739.2006.00559.x

Wunder, S., 2006. Are direct payments for environmental services spelling doom for sustainable forest management in the tropics. Ecol. Soc. 11, 23.

Wunder, S., 2005. Payments for environmental services: Some nuts and bolts. CIFOR Occasional Paper No. 42.

Wunder, S., 2000. The economics of deforestation: the example of Ecuador. MacMillan and St. Martin Press in association with St. Anthony's College.

Wunder, S., Alban, M., 2008. Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. Ecol. Econ. 65, 685–698. doi:10.1016/j.ecolecon.2007.11.004

Wunder, S., Engel, S., Pagiola, S., 2008. Taking stock: A comparative analysis of payments for environmental services programs in developed and developing countries. Ecol. Econ. 65, 834–852. doi:10.1016/j.ecolecon.2008.03.010

Wünscher, T., Engel, S., Wunder, S., 2008. Spatial targeting of payments for environmental services: A tool for boosting conservation benefits. Ecol. Econ. 65, 822–833. doi:10.1016/j.ecolecon.2007.11.014

Wynne-Jones, S., 2013. Connecting payments for ecosystem services and agri-environment regulation: An analysis of the Welsh Glastir Scheme. J. Rural Stud. 31, 77–86. doi:10.1016/j.jrurstud.2013.01.004

Yerena, E., Padrón, J., Vera, R., Martínez, Z., Bigio, D., 2003. Building Consensus on Biological Corridors in the Venezuelan Andes. Mt. Res. Dev. 23, 215–218. doi:10.1659/0276-4741(2003)023[0215:BCOBCI]2.0.CO;2

Yin, R., Liu, T., Yao, S., Zhao, M., 2013. Designing and implementing payments for ecosystem services programs: Lessons learned from China's cropland restoration experience. For. Policy Econ. 35, 66–72. doi:10.1016/j.forpol.2013.06.010

Zanella, M.A., Schleyer, C., Speelman, S., 2014. Why do farmers join Payments for Ecosystem Services (PES) schemes? An Assessment of PES water scheme participation in Brazil. Ecol. Econ. 105, 166–176. doi:10.1016/j.ecolecon.2014.06.004

Zapata, S.D., Benavides, H.M., Carpio, C.E., Willis, D.B., 2012. The economic value of basin protection to improve the quality and reliability of potable water supply: the case of Loja, Ecuador. Water Policy 14, 1. doi:10.2166/wp.2011.007

Zbinden, S., Lee, D.R., 2005. Paying for Environmental Services: An Analysis of Participation in Costa Rica's PSA Program. World Dev. 33, 255–272. doi:10.1016/j.worlddev.2004.07.012

Annex 1: The Socio Bosque programme

Based on: TEEBcase: Raes, L. and Mohebalian, P. (2014). The Socio Bosque Program for rainforest and páramo conservation, Ecuador. Available at: www.teebweb.org.

1.1. What was the problem?

Ecuador has been identified as one of 17 most ecologically diverse countries in the world (WCMC 2000, Mittermeier et al., 2004). The country has a total surface of 283,560 km^2, of which between 113,076 to 122,620 km^2 is native forest (Ministry of the Environment, Ecuador (MAE), 2012a). These native forests include primary as well as regenerated secondary forests. About 68,000 km2 of these forests are privately and collectively owned. The deforestation rate is one of the highest in South America with an annual rate of 890 km2 between 1990-2000 and 776 km2 between 2000-2008 (Mosandl et al., 2008, MAE, 2012a).

The Ecuadorian national policy framework has a strong mandate to slow the rate of deforestation. The Ministry of Environment found it important to develop a national conservation programme that would have the double objective of forests conservation and poverty alleviation (MAE, 2012b). The Socio Bosque Programme (SBP) was developed as one part of a larger group of conservation measures.

The project also arose based on the success of some local experiences, such as the *Gran Reserva Chachi*, led by Conservation International and formal GTZ (German Technical Cooperation, now renamed to German International Cooperation - GIZ), located in the province of Esmeraldas, where conservation agreements have been implemented in exchange for financial incentives (GTZ 2010, de Koning et al., 2011).

Another example is the municipality of Pimampiro, where agreements were established between local authorities and landowners with rights to areas of importance for water resources, to ensure the conservation of these areas (Wunder and Alban, 2008).

1.2. Which ecosystem services were examined and how?

In 2008, with the objective of diminishing the national rate of deforestation, the Ecuadorian government set up a payment scheme to incentivize forest conservation among individual and communal forest landowners. Although in the past the scheme did not specifically target certain regions there does exist a model of geographic prioritization that can be implemented depending on the number of participants and the availability of funding. Prioritization was performed though spatial targeting based on a ranking system, using three main criteria: (1) deforestation threat; (2) importance for ecosystem services provision: carbon storage, water cycle regulation, and habitat for biodiversity; (3) poverty levels. The threat of deforestation was prioritized based on the areas proximity to roads and waterways. The metric for the threat of deforestation was adjusted by using a three-dimensional model which included topography as a limiting factor. Carbon storage was prioritized based on the adaptation of studies conducted by FAO and IPCC which geographically estimated the comparative sequestration of carbon among national ecosystems. To prioritize water regulation, catchment areas were classified according to their importance in providing water to lower basins. Importance for biodiversity was defined based on the relative percentage of ecosystems represented in national protected areas. Points are given to each ecosystem service and level of poverty. Areas with the highest number of points are ranked as having the highest priority. The SBP sees the provision of these ecosystem services as a secondary output of the incentives for the more general goal of nature conservation. In part to comply with Article 74 of the 2008 constitution, that provides only the state the authority to appropriate, produce, deliver and regulate ecosystem services (MAE 2012b).

Forests must provide at least two of three ecosystem services. The SBP defines "forests" as any plant formation consisting of native species, which result from the natural process of ecological succession. This definition excludes commercial forest plantations. Secondary forests are allowed to enter the programme if they have been in a state of regeneration for over 20 years and have not been actively managed (MAE, 2012c). Initially protected areas were excluded from the programme, but in a later phase it was decided that families or communities that have legal land titles from before the date of protected area creation can also participate (de Koning et al., 2011).

Originally, the SBP only focused on forest ecosystems. From 2009 onwards, it also included páramo grassland ecosystems (MAE, 2010). Páramo are native Andean high-altitude grasslands, crucial for regulation of freshwater flows (Hofstede 1997, Mena et al. 2001). In Ecuador, intact páramo ecosystems comprise nearly 5% of the national territory of which approximately 40% lies within protected areas (Ortiz and Mena 2002). Prioritization for inclusion of páramo in the SBP is based on the area's level of threat, the provision of ecosystem services and levels of poverty. The ecosystem services included are hydrological services (identified by parameters such as: seasonal distribution, total rainfall and water demand by users), carbon storage, biodiversity refuge and connectivity (MAE, 2012c).

The priority maps are rarely used as many areas in the SBP are in low priority regions. However, since 2010, the maps have been used to analyse the applications when there exist budget constraints to help with the selection of areas. If a property is not located in a priority area, the application has to wait for nomination in a subsequent period. The application gets analysed for inclusion in the initiative if there are resources available.

1.3. How does the mechanism work?

The SBP consists of the transfer of a direct monetary incentive per hectare to individual and communal landowners. They contractually commit to the conservation and protection of native forests and/or páramos for a period of 20 years (MAE, 2012b).

Participation in the SBP is voluntary. Participants must be identified as belonging to at least one of the following legal categories: natural persons, legally constituted communes, indigenous nationalities, cooperatives and associations (MAE, 2012b). An official property title is a prerequisite for participation. However, in the case of indigenous communities also an "ancestry certificate" or a management agreement between the Ministry of the Environment and the community may be used. Another requisite is a topographical survey of the property. For areas smaller than 50 ha the programme may finance the survey totally or partially based on the socio-economic conditions of the participant(s) (MAE, 2012c).

The beneficiaries are required to protect and conserve the area as outlined by the contract. These requirements include the prohibition of: (1) logging, (2) changing the existing land-use, (3) burning, (4) activities which disturb the natural behaviour or threaten the territories capacity to harbour biodiversity, alter hydrological conditions or reduce carbon storage, and (5) commercial or sport hunting and fishing activities in the SBP area; as well as reporting to the Ministry of the Environment title transfers of the land benefiting from the incentive, preventing fires in areas under conservation and reporting changes to the vegetation cover within five days to the Ministry of the Environment and other authorities (MAE, 2011).

From 2008 until October 2011, the incentive scale applied by the SBP was uniform and did not differentiate between individual and communal landowners. However, the scale was changed significantly and now differentiates between type of landownership and ecosystem under conservation (Krause and Loft, 2013). For the first 50 ha of conservation area enrolled, the incentive is US$ 30/ha/year for individuals who own more than 20 ha of land. From ha 51 to 100 ha, the incentive decreases to US$ 20/ha/year and decreases further for additional ha (Supplementary Table 1.1). For individuals with less than 20 ha of land the payment is US$ 60/ha. Each SBP participant can freely decide how many hectares of forest or páramo ecosystem to enter in the programme (MAE, 2012c).

Supplementary Table 1.1.: Incentive Scale Socio Bosque Programme

Individuals with more than 20 hectares in their overall land title			Individuals with less than 20 hectares in their overall land title			Communities and associations for forests			Communities and associations for páramo		
Range of ha		Amount (US$)	Range of ha		Amount (US$)	Range of ha		Amount (US$)	Range of ha		Amount (US$)
1	50	30.00	1	20	60.00	1	100	35.00	1	50	60.00
51	100	20.00				101	500	22.00	51	100	40.00
101	500	10.00				501	1,800	13.00	101	900	20.00
501	5,000	5.00				1,801	5,000	6.00	901	3,000	10.00
5,001	10,000	2.00				5,001	10,000	3.00	3,001	10,000	4.00
> 10,001		0.50				> 10,001		0.70	> 10,001		1.00

Source: Ministry of the Environment, 2012c

The majority of funding for the programme comes from Ecuadorian state resources. Additionally, as of 2012, the German Development Bank (KfW) provides funding within a framework of cooperation between Germany and Ecuador (MAE, 2012b). In addition NGOs such as Conservation International (CI) through its Conservation Stewards Programme have been supporting the programme (CDKN Global, 2012). Recently the company General Motors Omnibus BB signed a cooperation agreement with the SBP for the conservation of 10,000 ha through an annual payment of US$ 230,000 during five years (MAE, 2014).

One of the aims of the SBP is that it should have direct and verifiable benefits for poverty alleviation and local development. A specific instrument was designed to guide and follow this process, called social investment plans. Each SBP applicant is required to complete a form outlining how the applicant(s) are planning to use the monetary incentive. The applicants have the flexibility to use the incentive according to their needs and preferences but are guided among different categories of investment (de Koning et al., 2011).

The Ministry of Environment monitors compliance of the SBP conventions and has the right to make on-site inspections at any time. Through the SBP the Ministry of Environment has generated a geo-database of the conservation areas. It can check compliance through satellite imagery and aerial photography. At the same time the idea is that participants are actively involved in the continuous monitoring of the conservation areas. Workshops are being held to educate participants about forest monitoring techniques. In addition, every two years participants have to provide a legal declaration of compliance with the programme's requisites. To monitor the socioeconomic impact of the programme an analysis of the social investment plans is carried out. These are combined with field visits to evaluate the investments (MAE, 2011).

In case of fulfilment with the agreement, transfers are made twice a year, in May and October (MAE, 2012b). The incentives will be suspended in case of non-compliance. Moreover, the agreement can be terminated indefinitely when there is major noncompliance with the conservation agreement (MAE, 2012c). If the participant decides to exit the programme before the end of the agreement and without any breach of the obligations, the environmental authority can establish a (partial) reimbursement to the Ministry of Environment of the incentive transferred so far (MAE, 2012c).

1.4. What was achieved?

The programme has experienced substantial growth since its initiation in 2008. As of June 2013 1,123,410.96 ha have been conserved through 2,052 individual and 147 communal agreements (Supplementary Table 1.2).

Supplementary Table 1.2: Results of the SBP until June 2013

	Individual Agreements		Collective Agreements		Total	
year	Number of contracts	Hectares	Number of contracts	Hectares	Number of contracts	Hectares
2008	40	107.31	21	168,765.33	61	168,872.64
2009	325	3,555.8	21	196,446.79	346	200,002.60
2010	525	13,837.97	20	167,606.87	545	181,444.85
2011	544	23,502.14	26	199,734.76	570	223,236.90
2012	419	30,573.79	45	247,282.05	464	277,855.84
2013	199	71,998.11	10	802,148.25	209	71,998.11
Total	2,052	143,575.14	147	979,835.81	2185	1,123,410.96
Incentive in 2013:						
US$ 3,042,414.76				US$ 5,224,704.30		US$ 8,267,119.06
Total allocated incentive since 2008: US$ 22,922,602.16						

Source: http://sociobosque.ambiente.gob.ec/

1.5. Lessons learned

As of 2014, the Ecuadorian government has allocated the majority of the programmes financial resources. Currently, the government is working to obtain additional financing in order to ensure the long term support. The overall financial sustainability strategy of the programme includes issuing SBP Certificates, international cooperation, off-setting, and possible REDD mechanisms (MAE, 2012c).

In terms of equity, one of the aims of the programme was to allow poorer households to participate in the programme. In the beginning the payment system provided US$ 30/ha to landowners for up to 50 ha of forestland enrolled. To allow smaller farmers with forestland to participate, the incentives were increased to US$ 60/ha for private landholders with less than 20 ha of land overall, not just forest (MAE, 2012b). Incentives for participating communities were also increased. An additional US$ 5/ha was provided to communities who enrol less than 100 ha. Krause and Loft (2013) found that while the change in the structure of the incentives made substantial improvements in the equitability of the SBP, additional changes should be made to design contracts based on the number of beneficiaries per contract and poverty indexes.

The majority of the costs of participating in the programme are incurred by the participants (Chapter 4). One of these costs was an annual legal declaration, which participants had to obtain to comply with the programmes requirements. This declaration in front of a notary now has to be done only every two years, decreasing the costs for the participants (MAE, 2012c).

References:

Climate and Development Knowledge Network (CDKN) Global. (2012). Ecuador’s Socio Bosque Programme. URL: www.cepf.net/news/top_stories/Pages/Conservation-incentivesMaking-conservation-pay-off-for-Ecuadorians.aspx.

Deutsche Gesellschaft für Technische Zusammenarbeit - GTZ. (2010). Conservación y Desarrollo, La Gran Reserva Chachi. Germany. URL: www.unl.edu.ec/agropecuaria/wpcontent/uploads/2012/03/Conservacion-y-Desarrollo-La-Gran-Reserva-Chachi1.pdf.

de Koning, F., Aguinaga, M., Bravo, M., Chiu, M., Lascano, M., Lozada, T. and Suarez, L. (2011). Bridging the gap between forest conservation and poverty alleviation: the Ecuadorian Socio Bosque program. Environmental Science & Policy, 14(5), 531–542.

Hofstede, R. (1997). La importancia hídrica del páramo y aspectos de su manejo. EcoPar, Quito, Ecuador.

Krause, T. and Loft, L. (2013). Benefit Distribution and Equity in Ecuador’s Socio Bosque Program. Society & Natural Resources, 26, 15.

Mena Vásconez, P, G. Medina, and R. Hofstede. (2001). Los Páramos del Ecuador: Particularidades, problemas y perspectiva. Abya Yala/Proyecto Páramo. Quito, Euador.

MAE - Ministry of the Environment of Ecuador. (2010). Socio Bosque: Conceptualización Implementación socio Y Avances Al Segundo Año De Implementaciòn. URL: http://sociobosque.ambiente.gob.ec.
MAE. (2011). Metodología de monitoreo para las áreas bajo conservación de Socio Bosque. URL: http://sociobosque.ambiente.gob.ec/files/monitoreometodo2011.pdf.
MAE. (2012a). Línea Base de Deforestación del Ecuador Continental. URL:http://sociobosque.ambiente.gob.ec/files/Folleto%20mapa-parte1.pdf.
MAE. (2012b). Sistematización de Socio Bosque 2012. URL: http://sociobosque.ambiente.gob.ec/files/%20SISTEMATIZACION%20SOCIO%20BOSQUE.pdf.
MAE. (2012c). Manual Operativo Unificado. Proyecto Socio Bosque. URL: http://sociobosque.ambiente.gob.ec/files/MANUAL%20OPERATIVO%20SB%20UNIFICADO%202012.pdf.
MAE . (2014). MAE y GM OBB firman convenio de conservación de 10 200 hectáreas de bosque. Proyecto Socio Bosque. URL: www.ambiente.gob.ec/tag/socio-bosque.
Mosandl, R., S. Günter, B. Stimm, and M. Weber. (2008). Ecuador suffers the highest deforestation rate in South America. pp 37–40. In: Beck, E., Bendix, J., Kottke, I., Makeschin,F. and Mosandl, R. (eds). Gradients in a tropical mountain ecosystem of Ecuador, Ecological studies Volume 198. Springer, Berlin, Germany.
Ortiz, D. and Mena, P. (2002). Serie Páramo 10: Las áreas protegidas y los páramos. GTP/Abya Yala, Quito, Ecuador.
Plan Nacional para el Buen Vivir - SENPLADES. (2009). URL: www.planificacion.gob.ec/plan-nacional-para-el-buen-vivir-2009-2013/.
Plan Nacional para el Buen Vivir 2013-2017 - SENPLADES. (2013). URL: www.planificacion.gob.ec/plan-nacional-para-el-buen-vivir-2013-2017/.
Sierra, R., C. Cerón, W. Palacios and R. Valencia, (1999) Mapa de vegetación del Ecuador continental. Proyecto INEFAN/GEF-BIRF. Wildlife Conservation Society and EcoCiencia, Quito, Ecuador.
World Conservation Monitoring Center - WCMC. (2000). Global biodiversity: Earth's living resources in the 21st century. Cambridge Press, Cambridge, UK.
Wunder, S., and Alban, M. (2008). Decentralized payments for environmental services: The cases of Pinampiro and PROFAFOR in Ecuador. Ecological Economics, 65(4), 685–698.

Annex 2: The Regional Water Fund, FORAGUA

Besed on TEEBcase by Raes, L.; Rengel, E. and Romero, J. (2012). Intermunicipal cooperation in watershed conservation through the establishment of a regional water fund – FORAGUA – in Southern Ecuador. Available at www.teebweb.org

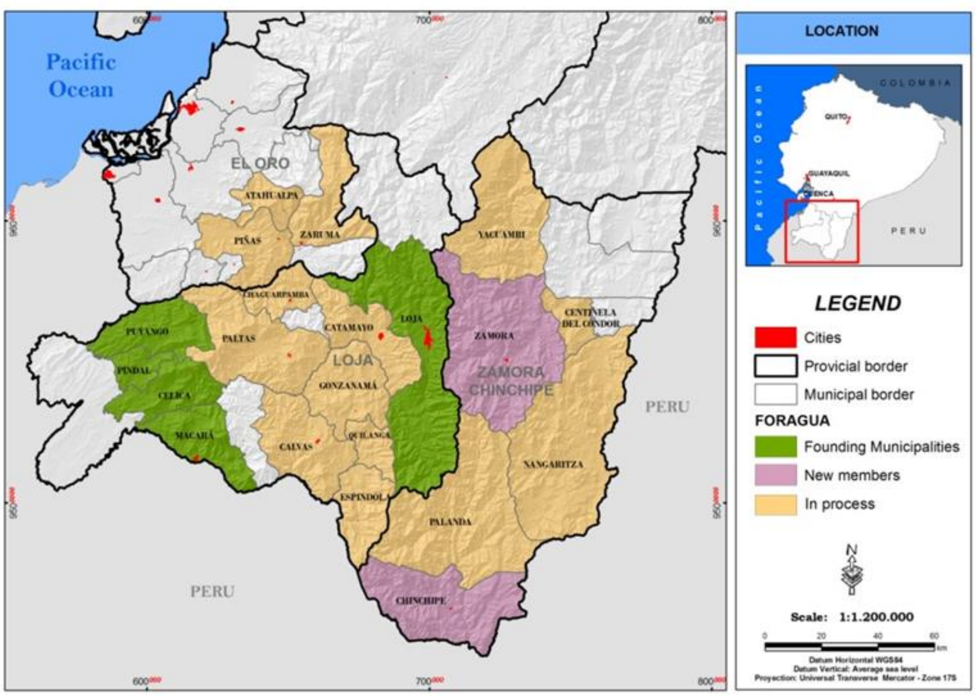
2.1. What was the problem?

In the Andean Region of Ecuador, mountain forests and Andean grasslands (páramos) provide key hydrological services. The most important hydrological services provided by these highland ecosystems are improved water quality through sediment retention (Brauman et al., 2007; Célleri and Feyen, 2009) and regulation of water flow (Bruijnzeel, 2004; Roa-Garcia et al., 2011). In addition to providing drinking water for people, these ecosystems provide habitat for an abundance of plant and animal species (José, 2001; Mutke and Barthlott, 2005).

The ability of these natural ecosystems to provide water services to people in up- and downstream areas of the watershed has been degraded by their conversion to agricultural land. This conversion has introduced all the problematic consequences of livestock grazing, periodic burning of pastures, and the use of pesticides, such as reduced water retention capacity and contaminated water due to sediment, manure and pesticide effluence. The growing population and the subsequent increasing demand for water adds to the problems of water provision. This is worsened by the drought part of the year, from which the Southern Andean region suffers. Recognizing the importance of protecting mountain ecosystems for their multiple ecosystem services, resource managers are pursuing innovative mechanisms to finance the conservation of these ecosystems, including the establishment of water funds which link upstream ecosystem service providers with downstream users.

2.2. Which ecosystem services were examined and how?

The Regional Water Fund (Fondo Regional del Agua – FORAGUA), has as its main objective the conservation of intact and restoration of degraded ecosystems through reforestation and natural regeneration in areas important for hydrological services in watersheds in Southern Ecuador (supplementary Figure 2.1). This is based on the assumed relationship between forests and hydrological service provision, not on actual measurements.



Supplementary Figure 2.1: Municipalities of FORAGUA (Source: NCI and Raes, 2013)

FORAGUA was established in 2009 with the participation of five municipalities (Celica, Pindal, Loja, Macara and Puyango) and an NGO (Nature and Culture International–NCI). Two municipalities joined the fund in 2011 (Chinchipe and Zamora) and work began to integrate two more (Piñas and Zaruma). In addition, 12 municipalities (Atahualpa, Centinela del Condor, Chaguarpamba Calvas, Catamayo, Espíndola, Gonzanamá, Nangartiza, Palanda, Paltas, Quilanga and Yacuambi), and a provincial government (Loja) have expressed interest in participating (supplementary Figure 1.1). The total endowment for the establishment of the fund was US$ 532,000 distributed among property and cash resources. Currently the capital fund is worth US$ 700,000. This includes the value of the properties that are incorporated as endowments.

By mandate, the municipalities that constitute the fund should devote the resources exclusively to water conservation. The trust-fund was established for a period of 80 years, which allows for undertaking long-term conservation programmes. Future additional benefits of the fund include increased biodiversity and carbon absorption, as forest is allowed to re-grow in the areas purchased by the municipalities, either through natural regeneration or reforestation. Biodiversity conservation is especially important in this region, as the Southern Andean region of Ecuador is a hot spot for biodiversity (Keese et al., 2007).

Watersheds in the region go from a height of 400 m.a.s.l. in the municipalities of Pindal and Macará, up to 3900 m.a.s.l. in the municipality of Loja. Key hydrological services targeted include water regulation and nutrient and sediment retention. FORAGUA has purchased land of hydrological interest from individual landholders in key watersheds.

In the Municipality of Loja most of the landholders that sold their land were living in the city and the land in the watersheds was not their main source of income. This made the effect of selling land on their livelihoods minimal. As watersheds where people are actively using the land gets incorporated, there is a growing need for alternative strategies for watershed conservation, so as not to displace the people involuntarily from their land. The aim is to conserve remaining ecosystems and regenerate forests, which serve to protect and enhance hydrological services as well as to promote biodiversity conservation (see also Goldman et al., 2010).

The importance of the hydrological services was measured primarily by the number of beneficiaries served by a particular watershed. Those watersheds that provide water to the largest number of users were targeted first. Also, the use of maps has been key. Especially aerial photographs and satellite images have allowed to identify the current land uses, and to have a much clearer idea of the state of the water catchments. Information such as soil types, slope, fertility, temperature and precipitation were also collected to determine if the current land use was the best within the range of potential uses of that land, where forest was assumed the best use. With all this information it was possible to determine which areas within the watershed are being over-exploited and which should be priority areas to be bought by FORAGUA.

2.3. How does the mechanism work?

The mechanism is based largely on the willingness of citizens to pay an additional amount on their water bill, known as the "environmental charge" for the conservation and restoration of water catchment areas. To do so, a survey examined the willingness to pay an additional charge for the protection of watersheds (Zapata el al., 2012). The total costs to implement protection and restoration measures as well as the costs of purchasing land were going to be high. In order not to increase the costs too much for individual households, especially those with limited resources, it was decided to make a classification of users using the same categories as were already used by the municipalities, i.e. residential, commercial and industrial, and official users. Finally the fee was set trying to average it with already existing ones (garbage collection, street lighting, etc.). The charge was formalized through the issuance of a municipal ordinance and is the responsibility of the Decentralized Autonomous Municipal Governments. Ordinances have been put in place in Loja, Macará, Puyango, Celica, Pindal and Chinchipe (supplementary Figure 2.1). The ordinances specify the size of the environmental charge. The fee ranges from the residential tariff of US$ 0.03/m^3 for a consumption of 0 to 50 m^3/month to US$ 0.07/m^3 for 101 m^3/month or more, a commercial tariff of US$ 0.07/m^3 and an official tariff of US$ 0.05/m^3. In addition to the funds obtained through the environmental fee, other sources of funding such as economic resources which are allocated by the municipality in its

budget or funds obtained on the basis of voluntary donation of the income tax can also be destined to the fund.
The ordinance also establishes the authority to declare municipal reserves designed to protect biodiversity and water sources. The declaration of land as a municipal reserve limits the use that can be made of the natural resources in the affected properties. Although the main focus is currently on purchasing land in the watersheds from individual landowners, private persons can maintain their land within the areas of hydrological importance, but with restrictions. In the case of private land, the owner or owners may retain ownership, while respecting the limitations established by the municipal ordinance and its regulations.
The ordinance only created the environmental charge and how to manage the collection of the money. The fund was created by deed, wherein the constituents established the mandates governing FORAGUA. One of the advantages of the fund is that it can ensure "fair use" of the financial resources collected.
FORAGUA is an endowment fund, where it is not the interest generated by the fund, but a portion of the fund itself that is used to finance conservation activities in the watershed (Laurans et al., 2012). The investment of the financial resources provided by the fund can only be done by implementing an individual investment plan for each municipality that is approved by the municipal council of the fund on a yearly basis. Of the total of funds raised with the environmental charge, 90% of the revenues are reinvested in the municipalities proportionally to the amount each municipality collected, and 10% is used for the functioning of the technical secretariat of the fund. Because the fund's financial resources are public, they are administered by the National Finance Corporation (Corporación Financiera Nacional – CFN). There exists a directive which empowers the CFN to invest the endowments in the stock market. The interests generated will complement the activities of the Secretariat. The annual amount for investment of the fund is US$ 400,000.
The mechanism is designed so that all municipalities provide their resources to the management of the fund's activities. Each municipality alone could not achieve this because in the case of small municipalities, the resources would not be sufficient to manage a technical secretariat or to implement broad conservation activities. Solidarity and collaboration are key to the proper functioning of the fund. Municipalities who generate more income support those who generate less.

2.4. Did the examination of ecosystem services generate impacts on decision-making or policies and, if so, how?

The Regional Water Fund now has a team consisting of three professionals, a furnished office equipped with computers, and a vehicle. Additionally, in March 2012, NCI established a support agreement through which four engineers are now working full time for the secretariat, assisting in management activities, technical reinforcement, and fundraising.
Following the enactment of the municipal ordinances, six municipalities collected about US$ 500,000 through environmental water charges. This money is being invested in watershed conservation programmes. More than 33,000 ha of municipal reserves have been created to protect and restore ecosystems that provide water to over 250,000 people. In addition to increasing protected areas by investing in “green infrastructure” such as forests, water utilities can save money by not having to invest in “grey infrastructure,” such as water filtration systems. Over the past two years, the municipality of Loja saved US$ 200,000 in chemicals for water treatment, due to the removal of livestock in the watershed following FORAGUA funded purchases of land.
As well as purchasing land of high hydrological value and declaring it municipal reserves, FORAGUA and the municipalities have initiated other projects in the watersheds, such as reforestation in the El Carmen watershed of Loja or the promotion of coffee agro-forestry instead of intensive maize production in Pindal.

2.5. Lessons learned

When small municipalities join together in a water fund, they can create economies of scale that make the working of the fund possible. In the case of FORAGUA this is currently not fully achieved. A portion of the budget needed for the functioning of the secretariat is provided through donors, mainly USAID and NCI. As more municipalities join the fund (supplementary Figure 2.1), the need for external donors to

finance the secretariat's operations decreases. Collaboration also facilitates the transfer of knowledge and good management practices; it makes solidarity between smaller and bigger municipalities possible; and, it strengthens the possibility of applying for national and international financial aid.
Currently the main focus of the fund has been the purchase of land of hydrological importance. The purchase of land can have an effect on the rural development of the region. More emphasis will go towards implementing production systems in the watersheds that improve hydrological services compared to current systems. This is for example already the case in Pindal were agroforestry systems are being implemented.

References

Brauman K.A, Daily G.C., Duarte T.K. and Mooney, H.A. 2007. The Nature and Value of Ecosystem Services: An Overview Highlighting Hydrologic Services. Annual Review of Environment and Resources 32 (1) 67-98

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

Célleri, R. and Feyen, J. 2009. The Hydrology of Tropical Andean Ecosystems: Importance, Knowledge Status, and Perspectives. Mountain Research and Development 29(4) 350-355

Goldman R.L, Benitez S., Calvache A., Davidson S., Ennaanay D., McKenzie E. and Tallis H. 2010. Water Funds for conservation of ecosystem services in watersheds, Colombia, available at: TEEBweb.org

José, C. 2001. La biodiversidad de Ecuador. Informe 2000. Ministerio de Ambiente, EcoCiencia y UICN, Quito, Ecuador.

Keese J., Mastin T. and Yun D. 2007. Identifying and Assessing Tropical Montane Forests on the Eastern Flank of the Ecuadorian Andes. Journal of Latin American Geography 6(1), 63-84

Laurans, Y., Leménager, T. and Aoubid, S. (2012). Payments for Ecosystem Services. From Theory to Practice – What are the Prospects for Developing Countries? STIN, France.

Mutke, J., Barthlott,W., (2005). Patterns of vascular plant diversity at continental to global scales. Biologiske Skrifter, 55, 521–531

Roa-García M.C., Brown S., Schreier H. and Lavkulich L.M. 2011. The role of land use and soils in regulating water flow in small headwater catchments of the Andes. Water Resources Research, 47 (5), W05510, doi:10.1029/2010WR009582

Zapata, S.D., Benavides, H.M., Carpio, C.E., Willis, D.B., 2012. The economic value of basin protection to improve the quality and reliability of potable water supply: the case of Loja, Ecuador. Water Policy 14, 1. doi:10.2166/wp.2011.007

Annex 3

The 16 cases where obtained through the following steps: (1) countries were identified with search words 'PES' and 'country name' (Supplementary Table A.1). This was followed by (2) a search with 'name of the scheme' (as identified through papers obtained with the first search). Information on schemes identified through the first search was additionally obtained through case studies published by The Economics of Ecosystems and Biodiversity (TEEB) network, and IIED country profiles[41] for PES. This was particularly important for those schemes for which no specific papers where identified in the second 'Web of Science' search. Additional information was further obtained through grey literature and official documents (see list below).

Supplementary Table A.1: Overview of the 16 study cases

Country	Case name	Hits ISI Web of Science		TEEB case	IIED case
		PES + country name	Scheme name		
Bolivia	NK-CAP	2	5		
	RWA Los Negros		2		X
Colombia	Water Fund for Life and Sustainability[a]	6	0	X	X
Costa Rica	ESPH-PROCUENCAS	22	0	X	X
	PES La Esperanza		0	X	X
	PSA		5	X	
Ecuador	FONAG	6	0	X	X
	FORAGUA		0	X	X
	PES Pimampiro		2		X
	PROFAFOR[a]		1		
	Socio Bosque Programme		4	X	
Mexico	Scolel Té	12	1	X	X
	PRONAFOR		1	X	X
Nicaragua	San Pedro del Norte	8	0		X
	Gil Gonzalez		0	X	X
Peru	Alto Mayo	1	0	X	X

[a] Additionally the search term for this scheme was 'water fund (or PES) Cauca Valley'
Note: search carried out on 2 and 3 February, 2014

Case study papers and reports used for preliminary analysis

Bolivia:
Noel Kempff Climate Action Project (NK-CAP):

Asquith, N.M., Ríos, M.T.V., Smith, J., 2002. Can Forest-protection carbon projects improve rural livelihoods? Analysis of the Noel Kempff Mercado climate action project, Bolivia. Mitig. Adapt. Strateg. Glob. Change 7, 323–337. doi:10.1023/A:1024712424319
Casas, A., 2008. Marcos legales para el pago por servicios ambientales en America Latina y el Caribe: Análisis de Ocho Países. Organization of American States, Washington DC, USA.
FAO, 2004. A review of carbon sequestration projects (No. AGL/MISC/37/2004). Food and Agriculture Organization of the United Nations, Rome, Italy.
Pereira, S.N.C., 2010. Payment for Environmental Services in the Amazon Forest: How Can Conservation and Development Be Reconciled? J. Environ. Dev. 19, 171–190. doi:10.1177/1070496510368047

[41] TEEB cases can be found at http://www.teebweb.org/resources/case-studies and the IIED cases at http://www.watershedmarkets.org/casestudies.html

RWA Los Negros:
Asquith, N., 2011. Pagos por servicios ambientales hidricos (PSA-H) en Bolivia.
Asquith, N.M., Vargas, M.T., Wunder, S., 2008. Selling two environmental services: In-kind payments for bird habitat and watershed protection in Los Negros, Bolivia. Ecol. Econ. 65, 675–684. doi:10.1016/j.ecolecon.2007.12.014
Asquith, N., Vargas, M.T., 2007. Fair deals for watershed services in Bolivia. Natural Resource Issues No. 7. International Institute for Environment and Development, London, UK.
Gomez Wichtendahl, C.C., 2009. Annex II: Bolivia Country Report, in: Greiber, T. (Ed.), Payments for Ecosystem Services. Legal and Institutional Frameworks. IUCN, Gland, Switzerland, pp. 125–186.
Porras, I., Neves, N., 2006. Los Negros, Bolivia. Compensation for Hydrological Environmental Services in Los Negros Cloud Forest. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.
Quintero, M. (Ed.), 2010. Servicios Ambientales Hidrológicos en la Región Andina. Estado del conocimiento, la acción y la política para asegurar su provisión mediante esquemas de pago por servicios ambientales. Instituto de Estudios Peruanos, Lima, Peru.
Website (intermediary NGO): http://www.naturabolivia.org/

Colombia:
Water Fund for Life and Sustainability:
Blanco, J., 2006. La Experiencia Colombiana en Esquemas de Pagos por Servicios Ambientales. CIFOR, Bogor Barat, Indonesia.
Goldman, R.L., Benitez, S., Calvache, A., Davidson, S., Ennaanay, D., McKenzie, E., Tallis, H., 2010. Linking People and Nature through Watershed Conservation in the East Cauca Valley, Colombia. TEEB case study. Available at: TEEBweb.org.
Goldman, R.L., Benitez, S., Calvache, A., Ramos, A., 2010. Water funds: Protecting watersheds for nature and people. The Nature Conservancy, Arlington, VA, USA.
Laurans, Y., Lemenager, T., Aoubid, S., 2012. Payments for Ecosystem Services. From Theory to Practice – What are the Prospects for Developping Countries? AFD, Paris, France.
Navarrete Le Bas, F., 2009. Annex III: Colombia Country Report, in: Greiber, T. (Ed.), Payments for Ecosystem Services. Legal and Institutional Frameworks. IUCN, Gland, Switzerland, pp. 187–238.
Porras, I., Neves, N., 2006. Valle del Cauca, Colombia. Valle del Cauca - land acquisition and land management contracts. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.

Costa Rica:
ESPH-PROCUENCAS
Dorado, V., Eguino, S., Octavio Ribera, M., Sangüeza, S., 2011. Mecanismos financieros del agua en América Latina. FUNDESNAP, La Paz, Bolivia and Fundación Avina, Panama City, Panama.
Gámez H., L., 2011. Internalización de servicios ambientales en tarifa de agua. Presentation given at the conference: '2do Encuentro Nacional de Mecanismos Locales de PSA', Mexico, F.D. 15 and 16 November.
Murillo, R., Kilian, B., Castro, R., 2011. Leveraging and Sustainability of PES. Lessons Learned in Costa Rica, in: Rapidel, B., Le Coq, J.F., Beer, J., DeClerck, F. (Eds.), Ecosystem Services from Agriculture and Agroforestry : Measurement and Payment. Earthscan Publications, London, UK, pp. 267–287.
Porras, I., Neves, N., 2006. Costa Rica - Empresa de Servicios Púpblicos de Heredia (ESPH). IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.
Redondo-Brenes, A. and Welsh, K., 2010. Procuencas Project, Costa Rica, Available at: TEEBweb.org
Rojas, M., Aylward, B., 2003. What are we learning from experiences with markets for environmental services in Costa Rica? A review and critique of the literature. (Working Paper), Environmental Economics Programme. International Institute for Environment and Development, London, UK.
Website ESPH: https://www.esph-sa.com/site/

PES La Esperanza:
Porras, I., 2010. PES as a strategy to minimize risk: The Case of La Esperanza Hydroelectric Power Company, Costa Rica. TEEB case, available at www.teebweb.org.
Porras, I., Neves, N., 2006. La Esperanza, Costa Rica. Hydroelectric Power (HEP) and cloud forest conservation. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.
Rojas, M., Aylward, B., 2003. What are we learning from experiences with markets for environmental services in Costa Rica? A review and critique of the literature. (Working Paper), Environmental Economics Programme. International Institute for Environment and Development, London, UK.
Rojas, M., Aylward, B., 2002. Cooperation between a small private hydropower producer and a conservation NGO for forest protection: The case of La Esperanza, Costa Rica, Land-Water Linkages in Rural Watersheds Case Study Series. FAO, Rome, Italy.
Website (provider NGO): http://www.acmcr.org/
PSA:
Arriagada, R.A., Sills, E.O., Pattanayak, S.K., Ferraro, P.J., 2009. Combining Qualitative and Quantitative Methods to Evaluate Participation in Costa Rica's Program of Payments for Environmental Services. J. Sustain. For. 28, 343–367. doi:10.1080/10549810802701192
Blackman, A., Woodward, R.T., 2010. User financing in a national payments for environmental services program: Costa Rican hydropower. Ecol. Econ. 69, 1626–1638. doi:10.1016/j.ecolecon.2010.03.004
Casas, A., 2008. Marcos legales para el pago por servicios ambientales en America Latina y el Caribe: Análisis de Ocho Países. Organization of American States, Washington DC, USA.
Castro Salazar, M., Peña Chacón, M., 2011. The Case of Costa Rica, in: Greiber, T., Schiele, S. (Eds.), Governance of Ecosystem Services: Lessons Learned from Cameroon, China, Costa Rica and Ecuador. IUCN, Gland, Switzerland, pp. 73–91.
Daniels, A.E., Bagstad, K., Esposito, V., Moulaert, A., Rodriguez, C.M., 2010. Understanding the impacts of Costa Rica's PES: Are we asking the right questions? Ecol. Econ. 69, 2116–2126. doi:10.1016/j.ecolecon.2010.06.011
Pagiola, S., 2008. Payments for environmental services in Costa Rica. Ecol. Econ. 65, 712–724. doi:10.1016/j.ecolecon.2007.07.033
Porras, I., 2010. Fair and green? Social impacts of payments for environmental services in Costa Rica. International Institute for Environment and Development, London, UK.
Porras, I., Neves, N., 2006. National Payment for Environmental Services (PES) programme, Costa Rica. Costa Rica's PES Programme - a financial mechanism for the recuperation and conservation of forest cover in Costa Rica. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.
Rojas, M., Aylward, B., 2003. What are we learning from experiences with markets for environmental services in Costa Rica? A review and critique of the literature. (Working Paper), Environmental Economics Programme. International Institute for Environment and Development, London, UK.
Zbinden, S., Lee, D.R., 2005. Paying for Environmental Services: An Analysis of Participation in Costa Rica's PSA Program. World Dev. 33, 255–272. doi:10.1016/j.worlddev.2004.07.012
Website: http://www.fonafifo.go.cr/

Ecuador:
FONAG:
Arias, V., Benitez, S., Goldman, R., 2010. Water fund for catchment management in Quito, Ecuador. TEEB case, available at www.teebweb.org.
Camacho, D.C., 2008. Payment schemes for environmental services in watersheds in Ecuador. Investig. Agrar.-Sist. Recur. For. 17, 54–66.
Echavarría, M., 2002. Financing Watershed Conservation: The FONAG Water Fund in Quito, in: Pagiola, S., Bishop, J., Landell-Mills, N. (Eds.), Selling Forest Environmental Services: Market-Based Mechanisms for Conservation and Development. Earthscan Publications, London, UK.

Espinosa Proaño, C., 2005. Payment for Water-Based Environmental Services: Ecuador's Experiences, Lessons Learned and Ways Forward (Technical Paper No. 2), IUCN Water, Nature and Economics Technical Paper. The World Conservation Union, Ecosystems and Livelihoods Group, Colombo, Sri Lanka.
Goldman-Benner, R.L., Benitez, S., Boucher, T., Calvache, A., Daily, G., Kareiva, P., Kroeger, T., Ramos, A., 2012. Water funds and payments for ecosystem services: practice learns from theory and theory can learn from practice. Oryx 46, 55–63. doi:10.1017/S0030605311001050
Goldman, R.L., Benitez, S., Calvache, A., Ramos, A., 2010. Water funds: Protecting watersheds for nature and people. The Nature Conservancy, Arlington, VA, USA.
Morales, M., 2011. The Case of Ecuador, in: Greiber, T., Schiele, S. (Eds.), Governance of Ecosystem Services: Lessons Learned from Cameroon, China, Costa Rica and Ecuador. IUCN, Gland, Switzerland, pp. 93–118.
Porras, I., Neves, N., 2006. FONAG, Ecuador. Water Conservation Fund in Quito - watershed protection contracts & land acquisition. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.
Quintero, M. (Ed.), 2010. Servicios Ambientales Hidrológicos en la Región Andina. Estado del conocimiento, la acción y la política para asegurar su provisión mediante esquemas de pago por servicios ambientales. Instituto de Estudios Peruanos, Lima, Peru.
Southgate, D., Wunder, S., 2009. Paying for Watershed Services in Latin America: A Review of Current Initiatives. J. Sustain. For. 28, 497–524. doi:10.1080/10549810902794493
Website: http://www.fonag.org.ec

FORAGUA:

Camacho, D.C., 2008. Payment schemes for environmental services in watersheds in Ecuador. Investig. Agrar.-Sist. Recur. For. 17, 54–66.
Goldman-Benner, R.L., Benitez, S., Boucher, T., Calvache, A., Daily, G., Kareiva, P., Kroeger, T., Ramos, A., 2012. Water funds and payments for ecosystem services: practice learns from theory and theory can learn from practice. Oryx 46, 55–63. doi:10.1017/S0030605311001050
Goldman, R.L., Benitez, S., Calvache, A., Ramos, A., 2010. Water funds: Protecting watersheds for nature and people. The Nature Conservancy, Arlington, VA, USA.
Kauffman, C.M., Echavarría, M., 2013. The evolution of water trust funds in Ecuador, in: Bjornlund, H., Brebbia, C.A., Wheeler, S. (Eds.), Sustainable Irrigation and Drainage IV. Management, Technologies and Policies., WIT Transactions on Ecology and the Environment. WIT Press Publishing, UK.
Porras, I., Neves, N., 2011. Ecuador, various Payment for Watershed Services (PWS) schemes. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.
Quintero, M. (Ed.), 2010. Servicios Ambientales Hidrológicos en la Región Andina. Estado del conocimiento, la acción y la política para asegurar su provisión mediante esquemas de pago por servicios ambientales. Instituto de Estudios Peruanos, Lima, Peru.
Raes, L., Rengel, E., Romero, J., 2012. Inter-municipal cooperation in watershed conservation through the establishment of a regional water fund – FORAGUA – in Southern Ecuador. TEEB case, available at www.teebweb.org.
Website: http://www.foragua.org/

PES Pimampiro:

Camacho, D.C., 2008. Payment schemes for environmental services in watersheds in Ecuador. Investig. Agrar.-Sist. Recur. For. 17, 54–66.
Casas, A., 2008. Marcos legales para el pago por servicios ambientales en America Latina y el Caribe: Análisis de Ocho Países. Organization of American States, Washington DC, USA.
Echavarria, M., Vogel, J., Albán, M., Meneses, F., 2004. The impacts of payments for watershed services in Ecuador. Emerging lessons from Pimampiro and Cuenca (Working Paper), Environmental Economics Programme. International Institute for Environment and Development, London, UK.
Espinosa Proaño, C., 2005. Payment for Water-Based Environmental Services: Ecuador's Experiences, Lessons Learned and Ways Forward (Technical Paper No. 2), IUCN Water, Nature and Economics

Technical Paper. The World Conservation Union, Ecosystems and Livelihoods Group, Colombo, Sri Lanka.
Morales, M., 2011. The Case of Ecuador, in: Greiber, T., Schiele, S. (Eds.), Governance of Ecosystem Services: Lessons Learned from Cameroon, China, Costa Rica and Ecuador. IUCN, Gland, Switzerland, pp. 93–118.
Porras, I., Neves, N., 2006. Pimampiro, Ecuador. Nueva America forest management plan: Payments for Environmental Services (PES) component. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.
Quintero, M. (Ed.), 2010. Servicios Ambientales Hidrológicos en la Región Andina. Estado del conocimiento, la acción y la política para asegurar su provisión mediante esquemas de pago por servicios ambientales. Instituto de Estudios Peruanos, Lima, Peru.
Quintero, M., Wunder, S., Estrada, R.D., 2009. For services rendered? Modeling hydrology and livelihoods in Andean payments for environmental services schemes. For. Ecol. Manag. 258, 1871–1880. doi:10.1016/j.foreco.2009.04.032
Southgate, D., Wunder, S., 2009. Paying for Watershed Services in Latin America: A Review of Current Initiatives. J. Sustain. For. 28, 497–524. doi:10.1080/10549810902794493
Wunder, S., Alban, M., 2008. Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. Ecol. Econ. 65, 685–698. doi:10.1016/j.ecolecon.2007.11.004
PROFAFOR:
Albán, M., Argüello, M., 2004. Un análisis de los impactos sociales y económicos de los proyectos de fijación de carbono en el Ecuador. El caso de PROFAFOR-FACE (No. 07), Markets for Environmental Services. Internal Institute for Environment and Development, London, UK.
FAO, 2004. A review of carbon sequestration projects (No. AGL/MISC/37/2004). Food and Agriculture Organization of the United Nations, Rome, Italy.
Morales, M., 2011. The Case of Ecuador, in: Greiber, T., Schiele, S. (Eds.), Governance of Ecosystem Services: Lessons Learned from Cameroon, China, Costa Rica and Ecuador. IUCN, Gland, Switzerland, pp. 93–118.
Wunder, S., Alban, M., 2008. Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. Ecol. Econ. 65, 685–698. doi:10.1016/j.ecolecon.2007.11.004
Website: http://www.profafor.com
Socio Bosque Programme:
De Koning, F., Aguiñaga, M., Bravo, M., Chiu, M., Lascano, M., Lozada, T., Suarez, L., 2011. Bridging the gap between forest conservation and poverty alleviation: the Ecuadorian Socio Bosque program. Environ. Sci. Policy 14, 531–542. doi:10.1016/j.envsci.2011.04.007
Krause, T., Collen, W., Nicholas, K.A., 2013. Evaluating Safeguards in a Conservation Incentive Program: Participation, Consent, and Benefit Sharing in Indigenous Communities of the Ecuadorian Amazon. Ecol. Soc. 18. doi:10.5751/ES-05733-180401
Krause, T., Loft, L., 2013. Benefit Distribution and Equity in Ecuador's Socio Bosque Program. Soc. Nat. Resour. 26, 1170–1184. doi:10.1080/08941920.2013.797529
Raes, L., Mohebalian, P., 2014. The Socio Bosque Program for rainforest and páramo conservation, Ecuador. TEEB case, available at www.teebweb.org.
Website: http://sociobosque.ambiente.gob.ec/

Mexico:
PRONAFOR (Pro-Arból, PSA-CABSA and PSA-H):
Corbera, E., Soberanis, C.G., Brown, K., 2009. Institutional dimensions of Payments for Ecosystem Services: An analysis of Mexico's carbon forestry programme. Ecol. Econ. 68, 743–761. doi:10.1016/j.ecolecon.2008.06.008
Kosoy, N., Corbera, E., Brown, K., 2008. Participation in payments for ecosystem services: Case studies from the Lacandon rainforest, Mexico. Geoforum 39, 2073–2083. doi:10.1016/j.geoforum.2008.08.007

McAfee, K., Shapiro, E.N., 2010. Payments for Ecosystem Services in Mexico: Nature, Neoliberalism, Social Movements, and the State. Ann. Assoc. Am. Geogr. 100, 579–599. doi:10.1080/00045601003794833
Muñoz-Piña, C., Guevara, A., Torres, J.M., Braña, J., 2008. Paying for the hydrological services of Mexico's forests: Analysis, negotiations and results. Ecol. Econ. 65, 725–736. doi:10.1016/j.ecolecon.2007.07.031
Rodricks, S., 2013. Bundling of ecosystem services in agroforestry, Mexico. mainly based on Corbera, E., et al 2009.
Shapiro-Garza, E., 2013. Contesting the market-based nature of Mexico's national payments for ecosystem services programs: Four sites of articulation and hybridization. Geoforum 46, 5–15. doi:10.1016/j.geoforum.2012.11.018
Southgate, D., Wunder, S., 2009. Paying for Watershed Services in Latin America: A Review of Current Initiatives. J. Sustain. For. 28, 497–524. doi:10.1080/10549810902794493
Web site: http://www.conafor.gob.mx/web/apoyos/pronafor/

Scolel Té:

Corbera, E., Brown, K., Adger, W.N., 2007a. The Equity and Legitimacy of Markets for Ecosystem Services. Dev. Change 38, 587–613. doi:10.1111/j.1467-7660.2007.00425.x
Corbera, E., Kosoy, N., Tuna, M.M., 2007b. Equity implications of marketing ecosystem services in protected areas and rural communities: Case studies from Meso-America. Glob. Environ. Change-Hum. Policy Dimens. 17, 365–380. doi:10.1016/j.gloenvcha.2006.12.005
FAO, 2004. A review of carbon sequestration projects (No. AGL/MISC/37/2004). Food and Agriculture Organization of the United Nations, Rome, Italy.
Morrison, A., 2010. Carbon Sequestration- Scolel Té Plan Vivo Program. TEEB case, available at www.teebweb.org.
Ruiz-De-Oña-Plaza, C., Soto-Pinto, L., Paladino, S., Morales, F., Esquivel, E., 2011. Constructing Public Policy in a Participatory Manner: From Local Carbon Sequestration Projects to Network Governance in Chiapas, Mexico, in: Kumar, B.M., Nair, P.K.R. (Eds.), Carbon Sequestration Potential of Agroforestry Systems, Advances in Agroforestry. Springer Netherlands, pp. 247–262.
Websites (intermediaries): http://www.ambio.org.mx/ and http://www.planvivo.org/

Nicaragua:

PES San Pedro Del Norte:

Corbera, E., Kosoy, N., Tuna, M.M., 2007. Equity implications of marketing ecosystem services in protected areas and rural communities: Case studies from Meso-America. Glob. Environ. Change-Hum. Policy Dimens. 17, 365–380. doi:10.1016/j.gloenvcha.2006.12.005
Kosoy, N., Martinez-Tuna, M., Muradian, R., Martinez-Alier, J., 2007. Payments for environmental services in watersheds: Insights from a comparative study of three cases in Central America. Ecol. Econ. 61, 446–455. doi:10.1016/j.ecolecon.2006.03.016
Porras, I., Neves, N., 2006. PASOLAC: Nicaragua, El Salvador, Honduras. Pilot schemes in ten communities in Central America. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.

PHES Gil Gonzalez:

Flores Barboza, E.M., Somarriba, D., Esquivel, M., Sánchez, K., Centeno, E., 2011. Informe de Sistematización de la Experiencia, Pagos por Servicios Ambientales, en el Municipio de Belén, Rivas.
Hack, J., 2010. Payment schemes for hydrological ecosystem services as a political instrument for the sustainable management of natural resources and poverty reduction – a case study from Belén, Nicaragua. Adv Geosci 27, 21–27. doi:10.5194/adgeo-27-21-2010
Hack, J., Kosmus, M., Kräuter, H.J., Weiskopf, B., Somarriba, D., 2013. Payments for hydrological Ecosystem Services in the Gil González Watershed – A Public-Private-Partnership Case Study.

Porras, I., Neves, N., 2011. Gil-Gonzalez Micro-Watershed, Nicaragua, Rehabilitation of forest in Rivas, Nicaragua. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.

Peru:

Alto Mayo Water Initiative:

Casas, A., 2008. Marcos legales para el pago por servicios ambientales en America Latina y el Caribe: Análisis de Ocho Países. Organization of American States, Washington DC, USA.
Porras, I., Neves, N., 2006. Alto Mayo, Peru. Payment for Environmental Services (PES) scheme in the micro-watersheds of Rumiyacu, Mishquiyacu y Almendra. IIED Markets for Watershed Services -Country Profile. Available at http://www.watershedmarkets.org.
Quintero, M. (Ed.), 2010. Servicios Ambientales Hidrológicos en la Región Andina. Estado del conocimiento, la acción y la política para asegurar su provisión mediante esquemas de pago por servicios ambientales. Instituto de Estudios Peruanos, Lima, Peru.
Quintero, M., Wunder, S., Estrada, R.D., 2009. For services rendered? Modeling hydrology and livelihoods in Andean payments for environmental services schemes. For. Ecol. Manag. 258, 1871–1880. doi:10.1016/j.foreco.2009.04.032
Renner, I., 2010. Compensation scheme for upstream farmers in municipal protected area, Peru. TEEB case, available at www.teebweb.org.
Sandoval, M., Capella, J.L., 2009. Annex IV Peru Report, in: Greiber, T. (Ed.), Payments for Ecosystem Services. Legal and Institutional Frameworks. IUCN, Gland, Switzerland, pp. 239–286.

Annex 4

Supplementary Table 4.1: Users' ES

PES Scheme		ES Units	Direct ES	Indirect ES
Alto Mayo Water Initiative	*Initially*		*X*	
	Currently		X	
ESPH - PROCUENCAS	*Initially*		*X*	
	Currently		X	
FONAG	*Initially*		*X*	
	Currently		X	
FORAGUA	*Initially*		*X*	
	Currently		X	
NK-CAP	*Initially*	*X*		
	Currently	No longer sells carbon credits		
PES La Esperanza	*Initially*	*X*		
	Currently	X		
PES Pimampiro	*Initially*		*X*	
	Currently		X	
PES San Pedro del Norte	*Initially*		*X*	
	Currently		X	
PHES Gil Gonzalez	*Initially*		*X*	
	Currently		X	
Plan Vivo Scolel Té	*Initially*	*X*		
	Currently	X		
PROFAFOR	*Initially*	*X*		
	Currently	No longer sells carbon credits		
PRONAFOR	*Initially*		*X*	*X*
	Currently		X	X
PSA	*Initially*			*X*
	Currently	X	X	X
RWA Los Negros	*Initially*		*X*	
	Currently		X	
Socio Bosque Programme	*Initially*			*X*
	Currently			X
Water Fund for Life and Sustainability	*Initially*		*X*	
	Currently		X	
Total	*Initially*	*4*	*10*	*3*
	Currently	3	11	3

Supplementary Table 4.2: Providers' ecosystem services

PES Scheme		Output-based	Input-based
Alto Mayo Water Initiative	*Initially*		*X*
	Currently		X
ESPH –PROCUENCAS	*Initially*		*X*
	Currently		X
FONAG	*Initially*		*X*
	Currently		X
FORAGUA	*Initially*		*X*
	Currently		X
NK-CAP	*Initially*	*X*	
	Currently		X
PES La Esperanza	*Initially*	*X*	
	Currently	X	
PES Pimampiro	*Initially*		*X*
	Currently		X
PES San Pedro del Norte	*Initially*		*X*
	Currently		X
PHES Gil Gonzalez	*Initially*		*X*
	Currently		X
Plan Vivo Scolel Té	*Initially*	*X*	
	Currently	X	
PROFAFOR	*Initially*		*X*
	Currently		X
PRONAFOR	*Initially*		*X*
	Currently		X
PSA	*Initially*		*X*
	Currently		X
RWA Los Negros	*Initially*		*X*
	Currently		X
Socio Bosque Programme	*Initially*		*X*
	Currently		X
Water Fund for Life and Sustainability	*Initially*		*X*
	Currently		X
Total	*Initially*	*3*	*13*
	Currently	2	14

Supplementary Table 4.3: Users' participation mechanisms

PES Scheme		Voluntary			Compulsory		Government
		NGO	HH[a]	Bus.[b]	HH[a]	Bus.[b]	
Alto Mayo Water Initiative	*Initially*						*X*
	Currently				X	X	X
ESPH - PROCUENCAS	*Initially*				*X*	*X*	
	Currently				X	X	
FONAG	*Initially*	*X*		*X*			*X*
	Currently	X		X	X	X	X
FORAGUA	*Initially*	*X*			*X*	*X*	*X*
	Currently	X			X	X	X
NK-CAP	*Initially*	*X*		*X*			
	Currently	No longer sells carbon credits					
PES La Esperanza	*Initially*			*X*			
	Currently			X			
PES Pimampiro	*Initially*	*X*			*X*	*X*	*X*
	Currently				X	X	X
PES San Pedro del Norte	*Initially*	*X*	*X*				*X*
	Currently				X	X	X
PHES Gil Gonzalez	*Initially*			*X*			*X*
	Currently			X			X
Plan Vivo Scolel Té	*Initially*		*X*	*X*			
	Currently	X	X	X			X
PROFAFOR	*Initially*			*X*			
	Currently	No longer sells carbon credits					
PRONAFOR	*Initially*				*X*	*X*	*X*
	Currently				X	X	X
PSA	*Initially*	*X*			*X*	*X*	*X*
	Currently	X	X	X	X	X	X
RWA Los Negros	*Initially*	*X*					*X*
	Currently	X			X	X	X
Socio Bosque Programme	*Initially*						*X*
	Currently			X			X
Water Fund for Life and Sustainability	*Initially*	*X*		*X*			
	Currently	X		X	X	X	X
Total	*Initially*	*8*	*2*	*7*	*5*	*5*	*10*
	Currently	6	2	7	10	10	12

[a] HH: Household; [b] Bus.: Business

Supplementary Table 4.4: Providers' participation mechanisms

PES Scheme		Voluntary			Government
		NGO	Household	Business	
Alto Mayo Water Initiative	*Initially*		*X*		
	Currently		X		
ESPH –PROCUENCAS	*Initially*		*X*		*X*
	Currently		X	X	
FONAG	*Initially*	*X*	*X*		*X*
	Currently	X	X		X
FORAGUA	*Initially*		*X*		*X*
	Currently		X		X
NK-CAP	*Initially*		*X*		*X*
	Currently		X		X
PES La Esperanza	*Initially*	*X*			
	Currently	X			
PES Pimampiro	*Initially*		*X*		
	Currently		X		
PES San Pedro del Norte	*Initially*		*X*		
	Currently		X		
PHES Gil Gonzalez	*Initially*		*X*		
	Currently		X		
Plan Vivo Scolel Té	*Initially*	*X*	*X*	*X*	
	Currently		X	X	X
PROFAFOR	*Initially*		*X*		
	Currently		X		
PRONAFOR	*Initially*		*X*		
	Currently		X		
PSA	*Initially*		*X*	*X*	
	Currently		X	X	
RWA Los Negros	*Initially*		*X*		
	Currently		X		
Socio Bosque Programme	*Initially*		*X*		
	Currently		X		
Water Fund for Life and Sustainability	*Initially*		*X*		*X*
	Currently		X	X	X
Total	*Initially*	*3*	*15*	*2*	*5*
	Currently	2	15	4	5

Supplementary Table 4.5: Users' payment setting mechanisms

PES Scheme		Bi-lateral agreement users and providers/ intermediaries	Price set by intermediary	Earmarked taxes or tariffs	Government budget allocation
Alto Mayo Water Initiative	*Initially*				*X*
	Currently			X	X
ESPH –PROCUENCAS	*Initially*	*X*		*X*	
	Currently			X	
FONAG	*Initially*	*X*			*X*
	Currently	X		X	X
FORAGUA	*Initially*			*X*	*X*
	Currently			X	X
NK-CAP	*Initially*	*X*			
	Currently		No longer sells carbon credits		
PES La Esperanza	*Initially*	*X*			
	Currently	X			
PES Pimampiro	*Initially*	*X*		*X*	*X*
	Currently			X	X
PES San Pedro del Norte	*Initially*	*X*			*X*
	Currently			X	X
PHES Gil Gonzalez	*Initially*	*X*			*X*
	Currently	X			X
Plan Vivo Scolel Té	*Initially*	*X*			
	Currently	X	X		X
PROFAFOR	*Initially*	*X*			
	Currently		No longer sells carbon credits		
PRONAFOR	*Initially*			*X*	*X*
	Currently			X	X
PSA	*Initially*		*X*	*X*	*X*
	Currently	X	X	X	X
RWA Los Negros	*Initially*				*X*
	Currently			X	X
Socio Bosque Programme	*Initially*				*X*
	Currently	X			X
Water Fund for Life and Sustainability	*Initially*	*X*			
	Currently	X		X	X
Total	*Initially*	*10*	*1*	*6*	*10*
	Currently	7	2	10	12

Supplementary Table 4.6: Providers' payment setting mechanisms

PES Scheme		Bi-lateral agreement providers and users/ intermediaries	Price set by intermediary	Government budget allocation
Alto Mayo Water Initiative	*Initially*	*X*		
	Currently	X		
ESPH -PROCUENCAS	*Initially*		*X*	
	Currently		X	
FONAG	*Initially*	*X*		*X*
	Currently	X		X
FORAGUA	*Initially*	*X*		*X*
	Currently	X		X
NK-CAP	*Initially*	*X*		
	Currently			X
PES La Esperanza	*Initially*	*X*		
	Currently	X		
PES Pimampiro	*Initially*	*X*		
	Currently	X		
PES San Pedro del Norte	*Initially*	*X*		
	Currently	X		
PHES Gil Gonzalez	*Initially*	*X*		
	Currently	X		
Plan Vivo Scolel Té	*Initially*		*X*	
	Currently		X	
PROFAFOR	*Initially*		*X*	
	Currently		X	
PRONAFOR	*Initially*		*X*	
	Currently		X	
PSA	*Initially*		*X*	
	Currently		X	
RWA Los Negros	*Initially*	*X*		
	Currently	X		
Socio Bosque Programme	*Initially*		*X*	
	Currently		X	
Water Fund for Life and Sustainability	*Initially*	*X*		*X*
	Currently	X		X
Total	*Initially*	*10*	*6*	*3*
	Currently	9	6	4

Supplementary Table 4.7: Use of contracts and/or ICDPs

PES Scheme	Contracts	No-contract (ICDP)
Alto Mayo Water Initiative	X	X
ESPH –PROCUENCAS	X	
FONAG		X
FORAGUA	X	X
NK-CAP		X
PES La Esperanza	X	
PES Pimampiro	X	
PES San Pedro del Norte	X	
PHES Gil Gonzalez	X	
Plan Vivo Scolel Té	X	
PROFAFOR	X	
PRONAFOR	X	
PSA	X	
RWA Los Negros	X	X
Socio Bosque Programme	X	
Water Fund for Life and Sustainability		X
Total	13	6

Supplementary Table 4.8: Intermediaries and their role

PES Scheme		Non-government Intermediary		Government Intermediary		Intermediary is buyer	Intermediary is provider
		Financial	Management	Financial	Management		
Alto Mayo Water Initiative	*Initially*			*X*	*X*	*X*	
	Currently			X	X	X	
ESPH - PROCUENCAS	*Initially*	*X*	*X*			*X*	
	Currently	X	X			X	X
FONAG	*Initially*	*X*	*X*	*X*	*X*	*X*	
	Currently	X	X	X	X	X	X
FORAGUA	*Initially*		*X*	*X*	*X*	*X*	*X*
	Currently		X	X	X	X	X
NK-CAP	*Initially*	*X*	*X*				
	Currently	Disappeared	X				
PES La Esperanza	*Initially*	No intermediary					
	Currently						
PES Pimampiro	*Initially*		*X*	*X*		*X*	
	Currently			X	X	X	
PES San Pedro del Norte	*Initially*		*X*	*X*		*X*	
	Currently			X	X	X	
PHES Gil Gonzalez	*Initially*		*X*	*X*	*X*	*X*	
	Currently		X	X	X	X	
Plan Vivo Scolel Té	*Initially*	*X*	*X*				
	Currently	X	X				
PROFAFOR	*Initially*	*X*	*X*				
	Currently		X				
PRONAFOR	*Initially*			*X*	*X*	*X*	
	Currently			X	X	X	
PSA	*Initially*			*X*	*X*	*X*	
	Currently			X	X	X	
RWA Los Negros	*Initially*		*X*	*X*			
	Currently		X	X	X		
Socio Bosque Programme	*Initially*			*X*	*X*	*X*	
	Currently			X	X	X	
Water Fund for Life and Sustainability	*Initially*	*X*	*X*			*X*	
	Currently	X	X	X		X	X
Total	*Initially*	*6*	*11*	*10*	*7*	*11*	*1*
	Currently	4	9	11	10	11	4

Annex 5

Supplementary Table 5.1: Standard Deviation of Experts' Ecosystem Service Indicator

Ecosystem	Hydrological Index	Erosion Control Index	Carbon Stock Index	Carbon Sequestration Index	Biodiversity Index	Connectivity Index
Primary forest	0.0	0.0	0.0	0.2	0.0	0.0
Páramo	0.0	0.0	0.0	0.3	0.0	0.0
Mature secondary forest	0.1	0.2	0.2	0.3	0.1	0.2
Young secondary forest	0.2	0.1	0.1	0.3	0.2	0.1
Forestry plantation	0.2	0.1	0.1	0.1	0.1	0.1
Organic coffee system	0.1	0.1	0.1	0.2	0.1	0.2
Abandoned pasture	0.2	0.2	0.2	0.2	0.2	0.2
Unmanaged pine/eucalyptus plantation	0.1	0.2	0.1	0.2	0.1	0.2
Conventional coffee	0.1	0.1	0.1	0.2	0.1	0.1
Pasture	0.2	0.3	0.1	0.1	0.1	0.1
Annual crops	0.0	0.1	0.0	0.1	0.1	0.0

5.1. Start-up costs

Coffee label:

Supplementary Table 5.2: Coffee Label Start-up Costs, data 2003, 2004, 2009, 2010, 2011

Activity	Total Costs (US$) (120 coffee producers)	Cost Research Area (US$) (37 coffee producers)	Cost US$/ha
Consultancy	14,390	4,437	87.7
Promotional Activities	5,781	1,782	35.2
Capacity Building	37,882	11,680	230.9
Commercialization	69,333[a]	35,630	704.4
Implementation Production Systems and Infrastructure	93,798	28,921	571.8
Administrative Costs	4,125	1,272	25.2
Total	225,309	83,723	1,655.2

[a] Only for 74 producers that sold coffee through FAPECAFES in 2010 and 2011

Socio Bosque Programme:

Supplementary Table 5.3: Socio Bosque Start-up Costs, data 2008, 2009, 2010

Activity	Total Costs (US$) (912,137.63 ha)	Cost Research Area (US$) (990.47 ha)	Cost US$/ha
Diffusion	914,138	1,050	1.1
Incorporation of land	1,484,381	1,377	1.4
Administrative costs 2008	79,439	86	0.1
Focal Point Loja Province[a]	52,830	2,629	2.7
Costs Participants to enter	n/d	2,533	2.6
Total		7,676	7.8

[a] Only for Loja (19,902.70 ha in 2012)

FORAGUA:

Supplementary Table 5.4: FORAGUA Start-up Costs, data 2006, 2007, 2008, 2009, 2011

Activity	Total Costs (US$) (4,856.00 ha)	Cost Research Area (US$) (830.97 ha)	Cost US$/ha
Setting Up Secretariat	266,788	45,653	54.9
Studies, consulting, lobbying	192,742	32,983	39.7
Purchase of Land	380,624	216,672	260.8
Communication and Diffusion	15,035	2,573	3.1
Total	855,190	297,880	358.5

5.2. Recurrent Costs

Management and specific production Costs

Coffee Label:

Supplementary Table 5.5: Coffee label Management and Specific Production Costs, data for 2011

Activity	Total Costs (US$) (616.08 bags)	Cost Research Area (US$) (205.33 bags)	Cost US$/ha
APECAEL Management	4,313[a]	1,437	28.4
Costs Quality Control	n/a	2,325	46.0
Additional weeding costs	n/a	10,474	206.2
Total		14,236	280.6

[a] 7US$ for APECAEL Administration

Socio Bosque Programme:

Supplementary Table 5.6: Socio Bosque Management and Production Costs, data for 2011

Activity	Total Costs (US$) (912,137.63 ha)	Cost Research Area (US$) (990.57 ha)	Cost US$/ha
Financial Sustainability	15,274	20	0.02
Administrative Costs	234,057	258	0.3
Total	249,331	277	0.3

FORAGUA:

Supplementary Table 5.7: FORAGUA Management and Production Costs, data for 2011

Activity	Total Costs (US$) (4,856.00 ha)	Cost Research Area (US$) (830.97 ha)	Cost US$/ha
Costs FORAGUA Secretariat	40,000	6,845	8.2
Costs EMAAL-EP	158,171	27,067	32.6
Management Watersheds	16,119	2,758	3.3
Reforestation Costs	41,823	16,743	20.2
Total	256,113	53,413	64.3

Transaction Costs

Coffee Label:

Supplementary Table 5.8: Coffee label Transaction Costs, data 2011

	Cost US$/Unit	Cost Research Area (US$) (37 producers-205.33 bags)	Cost US$/ha
Transaction Costs Farmers	29/producer[a]	1,082	21.4
Farmers Certification Costs	41/producer[b]	1,500	29.7
FAPECAFES Certification	0.6/bag	129	2.6
Total		2,711	53.6

[a] Mean transaction costs based on survey data

[b] For only 74 producers

Socio Bosque Programme:

Supplementary Table 5.9: Socio Bosque Transaction Costs, data 2011

Activity	Total Costs (US$) (912,137.63 ha)	Cost Research Area (US$) (990.57 ha)	Cost US$/ha
Deforestation baseline	363,174	394	0.4
Monitoring	559,729	608	0.6
TC Participants	n/d	3,740	3.8
Total		**4,742**	**4.8**

FORAGUA:

Supplementary Table 5.10: FORAGUA Transaction Costs, data 2011

	Loja (US$) (4,856 ha)	Research Area (US$) (830.97 ha)	Cost US$/ha
Monitoring	43,788	7,493	9.0
Monitoring water quality	62,357	0.0[a]	0.0
Investment Plan	3,250	556	0.7
Total	**109,394**	**8,049**	**9.7**

[a] This water monitoring is not the one incurred by EMAALEP for normal potable water, but is a specific project of FORAGUA in certain watersheds

5.3. Opportunity costs Calculations

For the opportunity cost calculations we used compounded data of per hectare revenue from deforestation and costs of pasture establishment of Knoke et al. (2009). The per hectare net revenue from milk production for each Socio Bosque participant, as obtained through the surveys, was added to the surplus obtained from deforestation and pasture establishment (Supplementary Tables 5.10 and 5.11). Additionally one participant would have carried out sustainable forestry. The income from sustainable forestry was provided by one of the co-authors (Aguirre, N.).

Supplementary Table 5.11: Surplus after pasture conversion

Type	Quantity
Timber (m3/ha)	21
Timber Price[a] (US$/m3)	87
Financial Income (US$/ha)	1858
Costs Cutting Timber[a] (US$/ha)	670
Pasture Establishment Costs[a] (US$/ha)	704
Total Conversion Costs (US$/ha)	1373
Average Surplus (US$/ha)	485

[a] Data compounded using 5% discount rate (Knoke et al., 2009a; Wunder, 2000)

Supplementary Table 5.12: Landholders Total Opportunity Costs

Type	Quantity
Income Milk/ha Farmer 1 (US$/ha)	0
Income Milk/ha Farmer 2 (US$/ha)	28
Income Milk/ha Farmer 3 (US$/ha)	49
Opportunity Costs Farmer 1 (US$/ha)	483
Opportunity Costs Farmer 2 (US$/ha)	821
Opportunity Costs Farmer 3 (US$/ha)	2,043
Total Opportunity Cost Dairy Production in 2011 (US$)	3,347
Yearly Sustainable Financial Return Forestry (US$/ha)	34
For 92 hectares (US$)	3,145
Total Opportunity Costs 2011 (US$)	6,492
Per Hectare Opportunity Costs (US$/ha)	7

5.4. Cost-Effectiveness Calculations

Supplementary Table 5.13: Cost-Effectiveness Ecosystem Service Provision Total Costs

	Cost US$/ha	ESI/ha	Cost US$/ESI point	ICERI/ha	Cost US$/ICERI point
Coffee label	1,982	1.5	1321	0.5	3,964
Socio Bosque	19	1.0	19	0.3	67
FORAGUA	432	1.9	228	1.1	408

Supplementary Table 5.14: Cost Effectiveness Rural Development Indicators Total Costs

	Cost US$/ha	HIPI/ha	Cost US$/HIPI point	ERWI/ha	Cost US$/ERWI point
Coffee label	1,982	107	19	211	9
Socio Bosque	19	0.1	215	4	6
FORAGUA	432	0	n/a	4	100

5.6. Division of the Costs

Supplementary Table 5.15: Division of Total Costs (With Start-up Costs)

	Total Costs (US$)	Donor (US$)	%	Government (US$)	%	Own Revenue (US$)	%	Participants (US$)	%
Coffee label	100,670	83,723	83	0	0	1,567	2	15,381	15
Socio Bosque	19,187	0	0	6,422	33	0	0	12,765	67
FORAGUA	359,343	122,411	34	89,088	25	147,844	41	0	0

5.7. Origin of the payments to cover the costs

Supplementary Table 5.16: Origin Payment Total Costs

	Total Costs (US$)	Donor (US$)	%	Government (US$)	%	Consumers (US$)	%
Coffee label	100,670	83,723	83	0	0	16,947	17
Socio Bosque	19,187	0	0	19,187	100	0	0
FORAGUA	359,343	122,411	34	89,088	25	147,844	41

References

Knoke, T., Calvas, B., Aguirre, N., Román-Cuesta, R.M., Günter, S., Stimm, B., Weber, M., Mosandl, R., 2009. Can tropical farmers reconcile subsistence needs with forest conservation? Front. Ecol. Environ. 7, 548–554. doi:10.1890/080131

Wunder, S., 2000. The economics of deforestation: the example of Ecuador. MacMillan and St. Martin Press in association with St. Anthony's College.

Annex 6

6.1. Milk production data

Data available for milk production calculations for the 95 farms were hectares of pasture, heads of cattle (cows, calves, heifers), time of sale of calves and average milk production per cow (see Table 5.2). Production cost data (pasture maintenance costs per hectare, fodder and veterinary costs per unit of cattle, and costs per litre of milk) were based on averages from the 19 farmers first interviewed. Fodder and veterinary costs were calculated by converting the heads of cattle into units of cattle (Supplementary Table 6.1) which were then multiplied by the average costs per unit of cattle obtained from the 19 farmers initially interviewed (Supplementary Table 6.3).

Although locally it is not uncommon for the same pasture to be in use for over 40 years ,(Knoke et al., 2009), we assumed that in order to maintain productivity, a cost for replanting pasture should be added. Data for these costs were provided by Knoke et al. (2014). We considered those systems with a cattle density of less than 0.75 units cattle/ha as low-input, and those with a higher density, as high-input systems[42]. We took into account the annual costs of pasture maintenance and fertilization for high-input pastures. This cost is constant for pastures with a density equal to or less than the average density of one unit of cattle per ha and increases gradually for those farms with a higher density. Only two farms in the sample had a density of two units of cattle per ha. Additionally, the cost of repairing or replacing fences is taken into account. Because we did not have a precise timing for pasture establishment and fencing, we calculated the average annual costs of these activities

As for cattle replacement rates, we assume cows produce milk for eight years, after which they are sold and replaced. Some farms would keep bull calves to sell in the second year, others would sell all their calves in the year they were born (Supplementary Table 6.2). To take into account productivity drops due to disease or other misfortunes, we assumed a conservative estimate of the lactation period of 220 days per 400-day cycle. We assumed that an average 25% of milk production goes to feeding calves. Assumed mortality rates were based on Ecuadorian averages (Ministry of Agriculture, Ecuador, 2012).

Supplementary Table 6.1: Cattle conversion units

	Conversion units
Calf 0-6 months	0.2
Calf 6-12 months	0.4
Calf 12-18 months	0.6
Pregnant heifer	0.8
Dry cow	1
Milk cow	1.2

Supplementary Table 6.2: Production data (see also Table 5.2)

	Duration/amount
Lactation period	220 days
Time between pregnancies	400 days
First birth	24 months
Production time	8 years
Annual replacement	16.7%
Mortality cattle (<12 months)[a]	7.52%
Mortality cattle (12-24 months)[a]	5.86%
Mortality cattle (>24 months)[a]	5.35%

[a] Ministry of Agriculture of Ecuador, 2012

[42] Knoke et al. (2014) considered two systems: low-input with a density of 0.4 heads/ha, and high-input with a density of 1.1 heads/ha. We considered the mathematical average (0.4 + 1.1)/2 to split farmers into one of these groups.

Supplementary Table 6.3: Cost and price data

	Price/cost per unit
Cost fodder non-milk cow	US$ 50/unit of cattle
Cost fodder milk cow producing 6l/day	US$ 87
Cost veterinary	US$ 13.5/unit of cattle
Cost insemination	US$ 15 (1 bull/day or 2 straws)
Cost planting pasture	US$ 240 (20 years)
(low density < 0.75 units cattle/ha)[a]	Annuitized: US$ 12/year
Cost planting pasture	US$ 640 (20 years)
(high density > 0.75 units cattle/ha)[a]	Annuitized: US$ 32/year
Weeding costs	US$ 30/year (2 paid labour days)
Cost fertilization high density pasture[a]	US$ 73/year
Cost fencing year 1 and 11/ha	US$ 200
Cost fencing other years/ha	US$ 35
Annuitized fencing cost	US$ 51.5
Fixed material costs (shovels, etc.)	US$ 11/year
Additional costs (costs, buckets, milking chair)	0.01/litre (min. US$ 33.5)
Income deforestation[b]	US$ 484.5/ha
Milk/calf rate	250 litres
Price calf < 12 months	US$ 70[c] and 180
Price calf > 12 months	US$ 0300[c] and 450
Price cow	US$ 250[c] and 350
Milk price (range)	US$ 0.29-0.45/litre
Milk price (average)	US$ 0.364/litre

[a] Knoke et al, 2014

[b] Knoke et al., 2009

[c] Data from Nature and Culture 2006 survey

Supplementary Table 6.4: Compounded milk prices

Price (US$/l)	Times occurred in data series	Occurrence probability
0.29	1	0.0060
0.30	2	0.0119
0.32	1	0.0060
0.33	20	0.1190
0.34	33	0.1964
0.35	38	0.2262
0.36	15	0.0893
0.37	10	0.0595
0.40	34	0.2024
0.42	5	0.0298
0.45	9	0.0536

Source: data provided by the Ministry of Agriculture of Ecuador, 2014

5.2. Tree plantation data

A production cycle of 20 years was assumed for timber plantations (Dunn et al., 1990; Knoke et al., 2009; Raboin and Posner, 2012). Risk in plantations may result from a (partial) failure to establish plantations due to seedling failure, fire damage and price fluctuations. The maximum profit from timber came in year 20. In the other years, smaller amounts of revenue were achieved from the sale of wood from thinning (Supplementary Tables 6.5, 6.6 and 6.8). In addition 75% of the establishment costs (subject to tree survival rates) were paid back through the incentive programme. A degree of uncertainty in reforestation was taken into account due to seedling survival uncertainty and fire damage (Weber et al., 2008) (Supplementary Table 6.7). The probability of being hit by fire was set at the annual rate of 0.0135 (averaging probabilities of fire around Podocarpus National Park from Knoke et al. (2009). For Andean alder fire damage of 5% for the first five years was assumed, 2% for the next five years and 1% for the other ten years. Damage from

fire was higher for pine, with damage of 10% in the first five years if fire occurred and 5% in the next 15 years (Supplementary Table 5.7) (Hofstede et al., 2002).
Based on data collected at 20 local saw mills, both pine and Andean alder prices were set between US$/m3 1 and 30. The local demand for pinewood currently exceeds that for Andean alder.

Supplementary Table 6.5: Alder plantation establishment and maintenance costs

Activity	Year	Unit	Quantity	US$/unit	USS/ha
Purchase and transport seedlings	1	Plant	1,111	0.25	278
Planting	1	Paid labour/day	10	15	150
Fertilization	1	Paid labour/day	3	15	45
Fertilizer	1	kilos	50	1	50
Area maintenance	2	Paid labour/day	6	15	90
Cleaning of the crown	2	Paid labour/day	5	15	75
Cleaning of the crown	3	Paid labour/day	5	15	75
First thinning	5	Paid labour/day	2	15	30
Felling trees	5	Paid labour with chainsaw/day	2	50	100
Area maintenance	5	Paid labour/day	4	15	60
Second thinning	8	Paid labour/day	1	15	15
Felling trees	8	Paid labour with chainsaw/day	2	50	100
Pruning	10	Paid labour/day	5	15	75
Area maintenance	10	Paid labour/day	4	15	60
Area maintenance	15	Paid labour/day	4	15	60
Felling trees	20	Paid labour with chainsaw/day	8	50	400
Transport	20	Paid labour/day	15	15	225
	20	Truck	350	4	1,400
Total					3,288

Own data

Supplementary Table 6.6: Pine plantation establishment and maintenance costs

Activity	Year	Unit	Quantity	US$/unit	USS/ha
Purchase and transport seedlings	1	Plant	1,111	0.25	278
Planting	1	Paid labour/day	10	15	150
Fertilization	1	Paid labour/day	3	15	45
Fertilizer	1	kilos	50	1	50
Area maintenance	2	Paid labour/day	6	15	90
Cleaning of the crown	2	Paid labour/day	5	15	75
Cleaning of the crown	3	Paid labour/day	5	15	75
First thinning	4	Paid labour/day	2	15	30
Felling trees	4	Paid labour with chainsaw/day	2	50	100
Area maintenance	5	Paid labour/day	4	15	60
First pruning	5	Paid labour/day	10	15	150
Second thinning	8	Paid labour/day	1	15	15
Felling trees	8	Paid labour with chainsaw/day	2	50	100
Second pruning	9	Paid labour/day	5	15	75
Area maintenance	10	Paid labour/day	4	15	60
Area maintenance	15	Paid labour/day	4	15	60
Felling trees	20	Paid labour with chainsaw/day	12	50	600
Transport	20	Paid labour/day	15	15	225
	20	Truck	400	4	1,600
Total					3,838

Own data

Supplementary Table 6.7: Additional data for tree plantation calculations

	Andean alder	**Pine**
Probability of survival of seedlings	0.8	0.90
Probability of fire	0.0135	0.0135
% plantation destruction	Year 1 to 5: 5 Year 6 to 10: 2 Year 11 to 20: 1	Year 1 to 5: 10 Year 6 to 20: 5
Growth/ha/year:	15 m^3	17.5 m^3
Timber price	US$ 1-30/m^3	US$ 1-30/m^3

Data based on own estimations, Dunn et al. (1990); Hofstede et al. (2002); Knoke et al. (2009); Olschewski and Benítez (2005)

Supplementary Table 6.8: Income tree plantations

	Andean alder				Pine			
Incentives:	Without		With		Without		With	
Year	Expected return	Standard deviation	Expected return	Standard deviation	Expected return	Standard deviation	Expected return	Standard deviation
1	-523	0	-208	156	-523	0	-170	117
2	-132	645	-33	80	-149	48	-37	60
3	-60	29	-15	36	-68	21	-17	27
4	0	0	0	0	300	119	388	140
5	219	136	219	136	-188	58	-188	58
6	0	0	0	0	0	0	0	0
7	0	0	0	0	0	0	0	0
8	531	266	531	266	590	203	590	203
9	0	0	0	0	-67	19	-67	19
10	-108	43	-108	43	-54	15	-54	15
11	0	0	0	0	0	0	0	0
12	0	0	0	0	0	0	0	0
13	0	0	0	0	0	0	0	0
14	0	0	0	0	0	0	0	0
15	-48	17	-48	17	-54	13	-54	13
16	0	0	0	0	0	0	0	0
17	0	0	0	0	0	0	0	0
18	0	0	0	0	0	0	0	0
19	0	0	0	0	0	0	0	0
20	2,077	922	2,077	922	2,618	796	2,618	796
Total	1,956	970	2,414	987	2,410	833	3,008	847

6.3. Socio Bosque Programme data

6.3.1. Conservation

The payments in the Socio Bosque programme depend on the size of the farm, and the area enrolled in the programme.

Supplementary Table 6.9: Incentive scale for the Socio Bosque programme

Individuals with more than 20 hectares in their overall land title			Individuals with less than 20 hectares in their overall land title		
Range of hectares		Amount (US$)	Range of hectares		Amount (US$)
1	50	30.00	1	20	60.00
51	100	20.00			
101	500	10.00			
501	5,000	5.00			
5,001	10,000	2.00			
> 10,001		0.50			

6.3.2. Restoration

The incentives for restoration were based on the method to be implemented (a total of five modes existed, each with different incentives, (Supplementary Table 6.10). The main interest in the research area is natural regeneration, which is understood by Socio Bosque's management as natural ecological succession accompanied by protection, management and control activities (Ministry of the Environment, Ecuador, 2014). The incentive was spread over the first three years follows: 40% the first year and 30% in each of the following two years. From the fourth year onwards, the farmer was eligible to participate in the (other) Socio Bosque programme for the next 17 years and to receive an incentive for conservation.

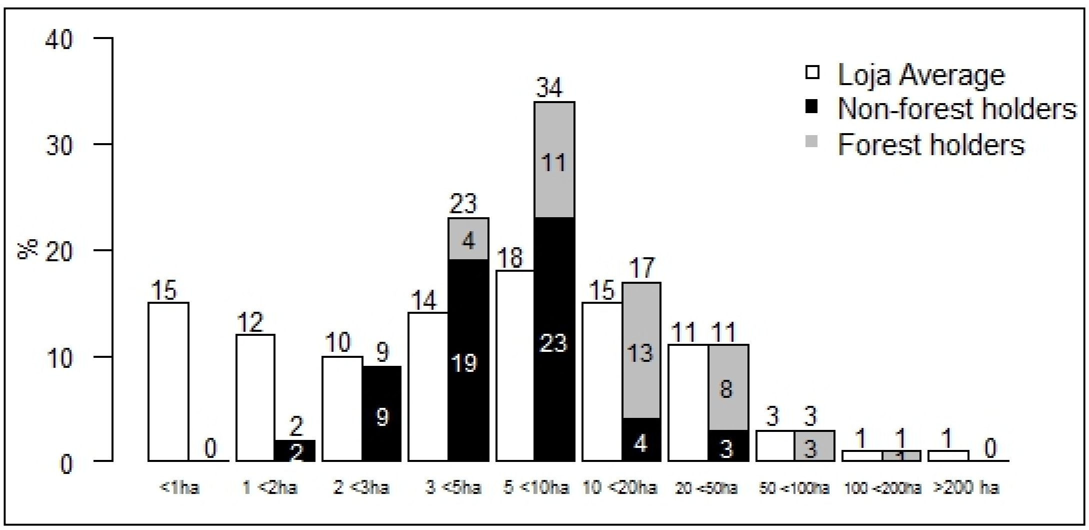
Supplementary Table 6.10: Incentives for restoration first three years

Modalities	% establishment costs paid back	Coast & Amazon	Sierra (Andes)
		US$/ha/3 years	
Natural regeneration	100	412	404
Revegetation with native species:			
In block	100	889	742
Enrichment	60	533	445
Ecological strips	65	578	482
Sustainable ecosystem use	75	667	556

Source: Data provided by the Ministry of Environment of Ecuador, 2015

6.4. Farm Size

Supplementary Figure 6.1 compares farm size categories for the province of Loja[43] with the 95 farms within the sample. The data on farm sizes was obtained from the Ministry of Agriculture of Ecuador (2012).



Supplementary Figure 6.1: percentage of farms according to farm size class for the province of Loja and sample

6.5. Calculation of household income

6.5.1. Off-farm income

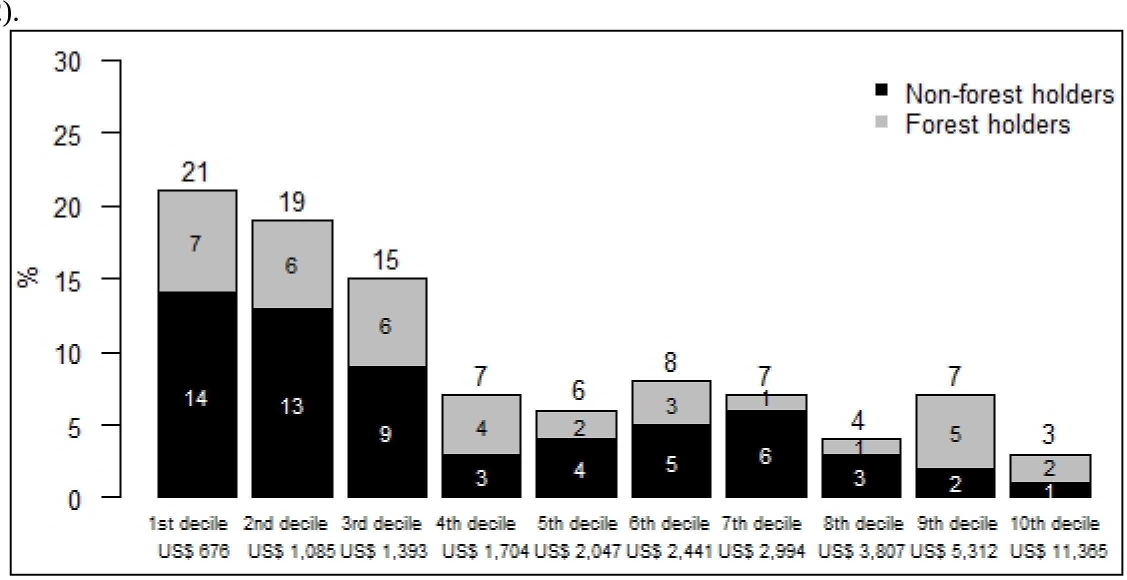
Respondents were often either unable or unwilling to estimate the wages earned from off-farm income by all household members. Instead, we asked about the type of work, and days per week worked, for each household member. Knowing their age and education, this allowed us to estimate off-farm income based on national data of income per job type in Ecuador (INEC, 2012). For income from farm labour we also used the agricultural labour wage of the area, which at the time of data collection was estimated at 10 US$ per day for women and 15 US$ per day for men.

[43] Data for the canton of Loja were not available

33% of respondents (versus 44% for the whole of Ecuador, ECLAC, 2011) received additional income from the Ecuadorian state through a conditional cash transfer "*Bono del Desarrollo Humano*" programme. The programme provided 35 US$ per month per household member at the time of research (Ponce et al., 2013), accounting for about a third of household income. To obtain information on participation, we asked respondents whether they received the bono or not, and also checked participants' information with the programme's eligibility requirements. Support goes to households who live below the poverty line (not including income from the bono), and is for households with children, for people over 65 years who do not receive another pension, and for disabled people (Ministry of Economic and Social Inclusion, Ecuador, 2012).

6.5.2. Non-dairy farm income

Different household income sources (net income from crops, milk production, livestock sales and off-farm income) were considered. Farm production consumed by the household was valued at farm-gate prices. Cost estimates were made from data collected from the 19 farmers interviewed in the first round. For the calculations of current milk income we used the 2011 milk price of US$ 0.4/litre (Ministry of Agriculture, Ecuador, 2013), and cattle prices collected during field research (see Supplementary Table 6.2). Supplementary Figure 6.2 compares our sample of households with the Ecuadorian income deciles (INEC, 2012).



Supplementary Figure 6.2: Distribution of sampled households classified according to Ecuadorian per capita income deciles

5.6. Output portfolios with pine plantations

Supplementary Table 5.11: Outcome of portfolios with Pine for milk producers without forest

		Current situation	Change through minimum variance portfolio		Change through optimal portfolio	
	Incentives:	Without	Without	With	Without	With
Average area (ha)	Pasture	5.7	- 0.6	- 4.5	- 2.3	- 4.0
	Plantation	/	+ 0.6	+ 0.1	+ 2.3	+ 0.2
	Restoration	/	/	+ 4.4	/	+ 3.8
Average portfolio weights	Pasture	1.00	- 0.13	- 0.83	- 0.24	- 0.63
	Plantation	/	+ 0.13	+ 0.01	+ 0.24	+ 0.03
	Restoration	/	/	+ 0.82	/	+ 0.60
Average expected return (US$/ha)		1,542	- 108	- 388	+ 152	+ 37
Average standard deviation of ER/ha		215	- 15	- 83	+ 66	- 53
Average change Sharpe ratio		4.4			+ 1.5	+ 3.1

Supplementary Table 5.12: Output portfolios for milk producers with forest considering pine plantations

		Current situation	Changes compared to current situation			
			Minimum variance portfolio		Optimal portfolio	
	Incentives:	Without	Without	With	Without	With
Average area (ha)	Pasture	11.6	- 1.5	- 9.6	- 3.5	- 8.5
	Plantation	/	+ 1.5	+ 0.2	+ 3.5	+ 0.2
	Restoration	/	/	+ 9.4	/	+ 8.3
	Deforestation	/	+ 8.4	+ 2.7	+ 8.4	+ 4.0
	Conservation	8.4[a]	- 8.4	- 2.7	- 8.4	- 4.0
Average portfolio weights	Pasture	0.68	- 0.11	- 0.56	- 0.27	- 0.55
	Plantation	/	+ 0.11	+ 0.01	+ 0.27	+ 0.01
	Restoration	/	/	+ 0.55	/	+ 0.54
	Deforestation	/	+ 0.32	+ 0.11	+ 0.32	+ 0.14
	Conservation	0.32[a]	- 0.32	- 0.11	- 0.32	- 0.14
Average expected return (US$/ha)		613 (902[b])	+ 300	+ 305	+ 483	+ 475
Average standard deviation of ER/ha		119 (175[b])	+ 10	- 46	+ 102	- 34
Average Sharpe ratio		0.2 (2.1[b])			+ 4.8	+ 9.5

[a] Current standing forest, not active conservation through Socio Bosque

[b] Milk production data not taking into account forest area

References

Dunn, W.W., Lynch, A.M., Morgan, P., 1990. Benefit-cost analysis of fuelwood management using native alder in Ecuador. Agrofor. Syst. 11, 125–139. doi:10.1007/BF00838725

ECLAC, 2011. Programas de Transferencias Condicionadas. Balance de la experiencia reciente en América Latina y el Caribe. Cuadernos de la CEPAL 95. United Nations, Santiago, Chile.

Hofstede, R.G.M., Groenendijk, J.P., Coppus, R., Fehse, J.C., Sevink, J., 2002. Impact of Pine Plantations on Soils and Vegetation in the Ecuadorian High Andes. Mt. Res. Dev. 22, 159–167. doi:10.1659/0276-4741(2002)022[0159:IOPPOS]2.0.CO;2

INEC, 2012. Encuesta Nacional de Ingresos y Gastos de los Hogares Urbanos y Rurales [WWW Document]. Inst. Nac. Estad. Censos. URL http://www.ecuadorencifras.gob.ec/encuesta-nacional-de-ingresos-y-gastos-de-los-hogares-urbanos-y-rurales/ (accessed 4.6.15).

Knoke, T., Bendix, J., Pohle, P., Hamer, U., Hildebrandt, P., Roos, K., Gerique, A., Sandoval, M.L., Breuer, L., Tischer, A., Silva, B., Calvas, B., Aguirre, N., Castro, L.M., Windhorst, D., Weber, M., Stimm, B., Günter, S., Palomeque, X., Mora, J., Mosandl, R., Beck, E., 2014. Afforestation or intense pasturing improve the ecological and economic value of abandoned tropical farmlands. Nat. Commun. 5, 5612. doi:10.1038/ncomms6612

Knoke, T., Weber, M., Barkmann, J., Pohle, P., Calvas, B., Medina, C., Aguirre, N., Günter, S., Stimm, B., Mosandl, R., von Walter, F., Maza, B., Gerique, A., 2009. Effectiveness and distributional impacts of payments for reduced carbon emissions from. ERDKUNDE 63, 365–384. doi:10.3112/erdkunde.2009.04.06

Ministry of Agriculture, Ecuador, 2013. Acuerdo No. 394.

Ministry of Agriculture, Ecuador, 2012. III CENSO NACIONAL AGROPECUARIO [WWW Document]. SINAGAP. URL http://sinagap.agricultura.gob.ec/censo-nacional-agropecuario (accessed 5.7.14).

Ministry of Economic and Social Inclusion, Ecuador, 2012. Agenda Social 2009–2011.

Ministry of the Environment, Ecuador, 2014. Plan Nacional de Restauración Forestal 2014-2017.

Olschewski, R., Benítez, P.C., 2005. Secondary forests as temporary carbon sinks? The economic impact of accounting methods on reforestation projects in the tropics. Ecol. Econ. 55, 380–394. doi:10.1016/j.ecolecon.2004.09.021

Ponce, J., Enríquez Bermeo, F., Molyneux, M., Thomson, M., 2013. Hacia una reforma del bono de desarrollo humano: algunas reflexiones. Abya Yala, Quito, Ecuador.

Raboin, M.L., Posner, J.L., 2012. Pine or Pasture? Estimated Costs and Benefits of Land Use Change in the Peruvian Andes. Mt. Res. Dev. 32, 158–168. doi:10.1659/MRD-JOURNAL-D-10-00099.1

Weber, M., Günter, S., Aguirre, N., Stimm, B., Mosandl, R., 2008. Reforestation of Abandoned Pastures: Silvicultural Means to Accelerate Forest Recovery and Biodiversity, in: Beck, E., Bendix, J., Kottke, I., Makeschin, F., Mosandl, R. (Eds.), Gradients in a Tropical Mountain Ecosystem of Ecuador, Ecological Studies. Springer Berlin Heidelberg, pp. 431–441.