DISCUSSION PAPER / 2009.02

The Potential and Limitations of Markets and Payments for Ecosystem Services in Agricultural Landscape Restoration

Critical Reflections Inspired by an Assessment of the RISEMP Program in Matiguás-Río Blanco, Nicaragua

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Comments on this Discussion Paper are invited.
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Abstract

During the last two decades the concept of Payments for Ecosystem Services (PES) has gained ever-increasing attention among a wide public of scholars as well as conservation and development practitioners. The main premises of this innovative conservation approach are appealing: private landowners, which in normal circumstances -i.e. in absence of any direct incentives- are poorly or not motivated to protect nature on their land, will do so if they receive direct payments from environmental service (ES) buyers, which at least cover part of the landowners' opportunity costs of developing the land. In this paper, however, we warn for an overenthusiastic adoption of the PES approach. Based on an extensive literature review and a field study of the Nicaraguan component of the 'Regional Integrated Silvopastoral Approaches to Ecosystem Management Project' (RISEMP), one of the main GEF-World Bank funded pioneering pilot projects of PES in Latin America, we argue that the concept still has to deal with several theoretical and practical lacunae. We argue that the concept of PES rests on loose foundations, mainly because of (i) a simplistic view on ES as discrete, quantifiable and marketable entities; (ii) an abstraction of the required landscape approach to conservation and the corresponding collective action precondition; (iii) a simplistic and arbitrary one-sided approach to the externality problem with important implications on the desirability of different policy instruments; (iv) a simplistic perception of socio-institutional reality and negligence of institutional effects on human behaviour and environmental morale; (v) the problematic character of transaction costs and a misleading justification of the approach based on the efficiency criterion; and (vi) a potential continuation of regressive financing of global commons with important fairness and sustainability implications. As such, we argue that the concept of PES could distract the attention for environmental problems away from the more complex underlying causes, which generally require broader locally embedded political action for their solution and not merely market creation. We think more debate about the desirability and conceptual clarity of the PES conservation tool is necessary.

Key words: Payments for Ecosystem Services (PES), environmental conservation policy, incentives, institutions, motivation crowding, RISEMP, Nicaragua

Résumé

Au cours des deux dernières décennies, le concept de Paiements pour Services Environnementaux (PSE) a reçu une attention toujours croissante auprès d'un large public de spécialistes et de practitiens de la conservation et du développement. Les prémisses principales de cette approche innovatrice sur la conservation sont attrayantes: les propriétaires fonciers privés, qui, dans des circonstances normales -c'est-à-dire en l'absence de tout encouragement direct- sont peu ou pas motivés pour protéger l'environnement sur leurs terres le seraient en échange de paiements directs de la part des acheteurs de services environnementaux, ce qui couvrirait au moins une partie des coûts d'opportunité des propriétaires fonciers pour le développement de leurs terres. Dans cet article, néanmoins, nous mettons en garde contre l'adoption trop enthousiaste de l'approche PSE. Sur base d'une large documentation et d'une étude sur le terrain de la partie nicaraguayenne du ''Projet régional de gestion intégrée des écosystèmes sylvopastoraux'' (RISEMP), un des principaux projets pilotes sur le PSE en Amérique Latine, financé par la Facilité Globale de l'Environnement (GEF)-Banque Mondiale, nous soutenons qu'un certain nombre de lacunes théoriques et pratiques du concept doivent encore être traitées. Nous démontrons que le concept du PSE repose sur des fondements peu solides, principalement en raison de (i) une vision simpliste des services environnementaux comme des entités pouvant être isolées, quantifiées et commercialisées; (ii) une abstraction de l'approche du paysage nécessaire à la conservation et à la condition préalable correspondant à l'action collective; (iii) une approche unilatérale, simpliste et arbitraire du problème des externalités avec des implications importantes sur l'opportunité des différents instruments de gestion; (iv) une perception trop simple de la réalité socio-institutionnelle et une négligence des effets institutionnels sur le comportement humain et l'éthique environnementale; (v) le caractère problématique des coûts de transaction et une justification trompeuse de l'approche basée sur le critère d'efficacité; et (vi) une continuation potentielle du financement régressif des biens communs avec des implications importantes d'équité et de durabilité. Par conséquent, nous soutenons que le concept de PSE puisse détourner l'attention pour les problèmes environnementaux de leurs plus complexes causes sous-jacentes, qui, pour leur solution, requièrent en général des actions politiques plus vastes et enracinées localement, et pas simplement la création de marchés. Nous espérons stimuler le débat sur la désirabilité et sur la clarté conceptuelle de l'outil PSE.

Mots-clés: Paiements pour Services Environnementaux (PSE), conservation de l'environnement, primes, institutions, effet d'éviction, RISEMP, Nicaragua

Resumen

Durante las dos últimas décadas el concepto de Pagos por Servicios Ambientales (PSA) ha ganado una creciente atención entre un publico extenso de académicos así como entre profesionales en el ámbito de conservación y desarrollo. Las premisas principales de este enfoque innovador sobre la conservación son atractivas: propietarios de tierras privadas, que en circunstancias normales -es decir, en la ausencia de incentivos directos- no están motivados a proteger la naturaleza en sus propiedades, sólo lo harán si reciben pagos directos de compradores de servicios ambientales (SA), los cuales al menos cubrirían parte de los costos de oportunidad del desarrollo de su tierra. Sin embargo, en este artículo advertimos sobre la adopción sobre-entusiasta de este enfoque. Basada en una revisión extensiva de la literatura y un estudio de campo del componente nicaragüense del 'Proyecto Regional de Manejo Integrado de Ecosistemas Silvopastoriles' (RISEMP por sus siglas en inglés), uno de los principales proyectos pilotos de PSA en América Latina financiado por el GEF-Banco Mundial, argumentamos que el concepto de PSA debe aclarar todavía varios vacíos teóricos y prácticos. Demostramos que el concepto de PSA está basado en fundamentos sueltos, causados por (i) una visión simplificada de SA como entidades separables, cuantificables y comerciables; (ii) una abstracción del imprescindible 'enfoque de paisaje' para la conservación y la correspondiente precondición de acción colectiva; (iii) un enfoque parcial y arbitrario sobre el problema de externalidades con implicaciones importantes sobre la conveniencia de diferentes instrumentos de política; (iv) una percepción simplificada de la realidad socio-económica y una negligencia de los efectos institucionales sobre el comportamiento humano y sobre el 'moral ambiental'; (v) el carácter problemático de costos de transacción y una justificación falaz del enfoque basado en el criterio de eficiencia; y (vi) la potencial continuidad de financiamiento regresivo de los bienes comunes con implicaciones importantes sobre equidad y sostenibilidad. Por consiguiente, argumentamos que el concepto de PSA podría desviar la atención de los problemas medioambientales alejada de las causas fundamentales y más complejas, las cuales generalmente requieren de acciones de políticas más amplias y localmente 'enraizadas' para su solución, y no solamente de la creación de transacciones mercantiles. Esperamos poder incitar a un mayor debate acerca de la conveniencia y claridad conceptual del concepto de PSA.

1. Introduction

Agricultural activities are broadly considered to be one of the main direct causes of tropical deforestation and biodiversity loss (Kaimowitz and Angelsen, 1998). In Latin America, and in Central America in particular, the unfettered advance of the agricultural frontier, caused by expanding cropping areas and pastures, is deemed to be the root cause of these problems (Faris, 1999; Kaimowitz, 1996; Kaimowitz and Angelsen, 1998; Utting, 1997). Although there is no consensus on the proximate causes and underlying driving forces of these phenomena (Angelsen and Kaimowitz, 1999; Geist and Lambin, 2002), several policy scenarios have already been tried in order to change the land use of farmers, most of them, however, without the positive impact hoped for (Baland and Platteau, 1996; Ferraro, 2001; Ferraro and Simpson, 2002; Pearce, 2004). Often, farmers do not seem to be motivated to adopt alternative forms of land use. In some cases, this is attributed to a lack of know-how and technical assistance, but more often the main reason is taken to be the absence of direct economic incentives (Ferraro, 2001; Ferraro and Simpson, 2002).

At the same time an increasing body of research and on-the-ground experiences is focussing its attention on the role of ecosystems as ecosystem service (ES) providers[1] (see especially MEA, 2005). This relatively new approach stresses the importance of ecosystems as not only having an intrinsic value, but -perhaps even more importantly- as providing huge economic benefits to society (Costanza et al., 1997; MEA, 2005). Consequently, it is only a small step to argue for the integration of the ecosystems into the global market by attributing a monetary value to their services (Costanza et al., 1997; Farber et al., 2002; Pearce, 1993; Pearce and Turner, 1990). In this way, it can finally get a chance to "earn its own right to survive in a world market economy" (McAfee, 1999: 134).

In this context, the concept of Payments for Ecosystem Services[2] (PES) has gained ever-increasing attention among a wide public of scholars as well as conservation and development practitioners. The premises of this innovative conservation approach are appealing: landowners, which in normal circumstances -i.e. in absence of direct incentives- are poorly or not motivated to protect nature on their land, will do so if they receive direct payments from ecosystem service buyers, which at least cover the landowners' opportunity costs of developing the land (Engel et al., 2008; Pagiola et al., 2002; Wunder, 2005). The main theoretical underpinnings of this approach emanate from neoclassical environmental economics (Pearce, 1993; Pearce and Turner, 1990; Perman et al., 1999), which ascribes the main causes of environmental degradation to the chronic failure of markets to internalise environmental externalities, and the public good character of ecosystem services which stimulates free-riding behaviour. Following neoclassic economical reasoning, the PES concept is fundamentally based on the assumption that human beings are almost exclusively governed by self-interest; a fact which should be optimally taken advantage of by designing mechanisms that can convert this individual selfishness

[1] ES services include provisioning services (e.g. food and fiber, fuel); *regulating services* (e.g. pollination, flood control), *cultural services* (e.g. recreation, spiritual) and *supporting services* (e.g. nutrient cycling, soil formation) (MEA, 2005); For a more detailed overview, see for example Zhang et al., 2007.

[2] Often also referred to as Payments for *Environmental Services*. In this paper we use the terms *environmental* and *ecosystem services* interchangeably. For an overview of other terminologies of the same concept, see Wunder, 2005. Although there exist some ideological differences in using various terms, such as *compensations*, *markets*, or *rewards* instead of payments (Ravnborg et al., 2007), the differences in practice are rather semantic, but could have significant implications for the local framing and acceptance or resistance of the concept (Wunder and Vargas, 2005).

into maximum social welfare. Within this framework PES adherents claim that through the creation of markets and/or market-correcting tools the internalisation of environmental externalities can be achieved. Furthermore they believe that the use of market mechanisms and direct incentives will lead to the most efficient allocation of scarce conservation funds (Ferraro, 2001, Ferraro and Simpson, 2002, Pagiola et al, 2002; Pearce, 2004).

In this paper, however, we warn for an overenthusiastic adoption of the PES approach and discuss several shortcomings in both the theoretical underpinnings and their practical implications. Our research is based on an extensive literature review and a field study of the Nicaraguan component of the recently terminated 'Regional Integrated Silvopastoral Ecosystem Management Project' (RISEMP), implemented from 2002 until 2008 in three Latin American countries (Nicaragua, Costa Rica and Colombia), under the auspices of the Global Environment Facility (GEF) and the World Bank (World Bank, 2002). The project could be considered as one of the main pioneering pilot attempts to investigate the potentials of PES schemes in agricultural landscape restoration, and serves as a foundation for further replication throughout the world.

The rest of the paper is subdivided into five sections. In section 2 we briefly take a closer look at the definition, theoretical underpinnings, and classification of PES mechanisms. In section 3 we describe the characteristics of the RISEMP and the Nicaraguan pilot site. The main results of the project are presented in section 4, with a special focus on both the land use changes and the (underlying) motivations for these changes. Based on our field investigation and further theoretical reasoning, we argue in section 5 that the PES concept still has to deal with several theoretical and practical lacunae, which can be subdivided into six interrelated issues: (i) simplistic view of ES as discrete, quantifiable and marketable entities; (ii) abstraction of the required landscape approach to conservation and the corresponding collective action precondition; (iii) simplistic and arbitrary one-sided approach to the externality problem with important implications for the required policy instruments; (iv) simplistic view of socio-institutional reality and negligence of institutional effects on human behaviour and environmental morale; (v) problematic character of transaction costs and a misleading justification of the approach based on the efficiency criterion; and (vi) a potential continuation of regressive financing of global commons with important fairness considerations and implications on the sustainability of the approach. In section 6 we conclude and emphasise the need for clearer conceptual foundations of the concept.

2. The PES Concept and its Theoretical Foundation

2.1. Underlying Philosophy: Externalities and Missing Markets

The underlying logic for the creation of markets for ecosystem services in agricultural landscapes typically goes somewhat like this: ecosystems, such as natural or secondary forests, but also human-interfered landscapes like cultivated agricultural parcels, provide mankind not only with different sorts of marketed goods but also with ‘services’, such as carbon sequestration and storage, biodiversity protection, watershed protection, and scenic beauty which are provided at different local, regional, and global scales (Landell-Mills and Porras, 2002; MEA, 2005; Wunder, 2005). Since these ES are public or club goods, and thus externalities for which markets are usually inexistent and for which the beneficiaries are only rarely paying, society is systematically underprovided with these services (Binkley, 2005; Landell-Mills and Porras, 2002; Pagiola et al., 2002). Since the de facto ecosystem managers (farmers, loggers, etc.) often receive only few private benefits from land uses such as forest conservation or ES-enhancing investments, often the socially more optimal land uses cannot financially compete with other -from the land users’ private point of view- more attractive land uses, such as croplands or pastures (Engel et al., 2008; Pagiola et al., 2002). This problem of market failure is believed to be one of the main causes of environmental degradation (Pearce, 1993; Pearce and Turner, 1990; Richards, 1999; Richards, 2000). Or as Engel et al. (2008: 664) state it: “While not all conversion of capital is undesirable, the existence of many forms of market failure means that natural capital depletion is often much greater than would be socially optimal”.

Since, according to most PES adherents, other policy instruments, such as governmental regulatory frameworks (often referred to as command-and-control measures), taxes and subsidies, or more community-based voluntary and educational approaches, have often proven to be ineffective in correcting market failure and halting further degradation[1] (Ferraro, 2001; Pagiola et al., 2002), and offer too little value for declining funding, they propose exploring and often argue in favour of the creation of parallel markets for tradable environmental externalities[2]. In these markets, potential service providers could commercialise the positive externalities which they create by managing their land in an ‘adequate’ way. In this way the PES mechanism establishes a compromise between social conservation and private land user benefits (Pagiola et al., 2005) by shifting control of natural resources from states to markets (McAfee and Shapiro, 2008).

[1] It should be noted that some of the more recent PES literature is somewhat more nuanced and less caricatural and recognises the advantages and disadvantages of different (non-PES) approaches, as well as the need to use PES as part of broader policy approaches (see for example Enget et al., 2008). Nevertheless, it is often claimed that PES is “most useful in the intermadiate range of positive but numerically small opportunity costs: degraded pastures, marginal croplands, forests in slow-moving agricultural frontiers, etc.” (Wunder, 2005: 21). As we will see further below, these conditions are exactly the conditions met in the RISEMP context (see also Pagiola et al., 2007).

[2] This implies attaching a private property right or a price to public environmental goods and services (Hubermann and Leipprand, 2006).

By trying to internalise nature's value into the wider economy, the main philosophy of PES can be traced back to neoclassical environmental economics (Pearce, 1993; Pearce and Turner, 1990; Perman et al., 1999), but moves beyond the Pigouvian philosophy[1] of taxing negative or subsidising positive externalities (in an attempt to reflect the 'true' value of a product to society) within existing markets, by directly focussing on the provision of certain (positive) services which are traded in newly-created markets. PES adherents perceive the problem not so much as a 'simple' pollution problem, but rather as a social cost problem (Salzman, 2005), and attempt "to put in practice the Coase theorem, which stipulates that the problems of external effects can, under certain conditions, be overcome through private negotiation between affected parties" (Coase, 1960, as cited by Engel et al., 2008). In other words, the PES approach recognises the externalities of certain production activities, but in contrast to the Pigouvian philosophy detaches the positive externalities from their marketable commodity and creates a parallel market for them, which should lead to the lowest-cost conservation and consequently the highest social welfare.

The explicit focus on positive externalities shifts the attention from the common-used 'Polluter Pays Principle' (PPP) to the 'Beneficiary Pays Principle' (BPP) (Pagiola et al., 2002; Pearce, 2004) or the 'Provider Gets Principle' (PGP) (Hanley et al., 1998; Hubermann and Leipprand, 2006). The land user is now considered to be a service provider instead of a polluter, and has the opportunity to add another 'cash crop' to his or her production activities (Salzman, 2005): the ES, which can either emerge as a joint product of other produced marketable goods, or be a product that is independently generated. Furthermore, reliance on the market mechanism and direct payments should secure the basic economic premise of efficiency optimisation of scarce conservation funds (Ferraro, 2001; Ferraro and Simpson, 2002; Pagiola et al., 2002; Pearce, 2004), by taking advantage of the landowners' knowledge of the cost of ES provision and to seek out the low-cost providers (Engel et al., 2008) or concentrate on the higher-benefit cases (Pagiola et al., 2005). In this context, and although poverty alleviation is not the main objective of PES schemes, it is often also considered to be an important positive side-effect of the environmental market (Grieg-Gran et al., 2005; Landell-Mills and Porras, 2002; Pagiola et al., 2005).

2.2. Definition and Classification

As with any other market, the creation of a market for ES implies four minimum conditions: the existence of a commodity, the existence and manifestation of demand, the existence and manifestation of supply (Pagiola et al., 2002), and -most importantly, but often overlooked- the intersection of the latter two. Based on these conditions, and in order to distinguish it from other economic conservation instruments, Wunder (2005: 3) provides a generally-accepted definition of the more specific concept of PES as "a voluntary transaction where a well-defined ES (or a land-use likely to secure that service) is being 'bought' by a (minimum one) ES buyer from a (minimum one) ES provider if and only if the ES provider secures ES provision (conditionality)". This definition presupposes some basic requirements which have to be fulfilled in order to belong to the PES family. First, the voluntary transaction assumes that ES providers have de

[1] Within the Pigouvian framework there is still need for state intervention (to determine the amount of taxes or subsidies and the quantity and quality of the (dis)services provided), which would lead to below-optimal solutions (Salzman, 2005; Pagiola et al., 2002). However, the minimisation of state intervention by PES proponents neglects the actual dependence of most PES schemes on state regulation and substantial public subsidies (McAfee and Shapiro, 2008). In this context some authors refer to markets' need of 'soft regulations' (Pearce, 2004).

facto or real land-use choices, which distinguishes it from command-and-control approaches (Ravnborg et al., 2007; Wunder, 2005). Secondly, the commodity itself should be well-defined (which implies measurability of the ES), or at least the land-use proxy which will likely result in the provision of the ES. Wunder recognises that this part of the definition can be problematic, especially because the lack of scientific knowledge on the relationships between a proxy and its real ES providing effect can undermine the sustainability of the transaction. Efforts are sometimes made to bundle various ES together or layer payments from different providers (Engel et al., 2008). Thirdly, a PES scheme requires resources going from the buyers to the providers, though the transfer can occur through an intermediary. It is in this criterion that the PES approach is particularly innovative; it ties the direct payments immediately to the investment goals instead of focussing on indirect conservation actions (Ferraro and Simpson, 2002). Finally, the hardest requirement to meet, according to Wunder, is the conditionality criterion of the scheme, which in practice implies the establishment of a baseline and (expensive) monitoring of compliance by the buyers or intermediaries, which might generate prohibitive transaction costs. Although the definition sketches the PES approach as a rather simple and straightforward mechanism which seems easy to create, a closer look at every criterion of the 'pure' approach leads even Wunder to the conclusion that there have been very few 'true PES' schemes satisfying all the criteria (Wunder, 2005). In section 5 we will further elaborate on several of these issues.

Within the growing PES literature several categorisations of PES programs can be found, depending on which underlying criteria are used to distinguish different schemes within the PES family. Landell-Mills and Porras (2002), for example, distinguish payment schemes by focussing on the resource contents of the service (biodiversity, carbon, water, and landscape beauty). Wunder (2005) grounds a possible categorisation on the basis of the actual conservation practice: conservation (paying for opportunity costs of conservation) versus asset-building schemes (paying for restoration activities). Another distinction commonly used is based on types of buyers (Engels et al., 2008): 'government-financed' schemes, in which the buyers of the ES are a third party acting on behalf of service users (typically a government agency or an international financial or conservation institution) versus 'user-financed' schemes, in which the buyers are the actual users of the ES (for example, downstream water users paying upstream farmers). Salzman (2005) asserts that in practice most of the PES schemes are monopsonies, in which there are multiple service providers, but only one ES buyer. Finally, another common distinction can be made by splitting the schemes into 'output-based' PES programs, in which payments are directly based on the ES provided, and the most commonly 'input-based' or 'area-based' schemes, which focus on land use proxies (Engel et al., 2008; Wunder, 2005).

2.3. PES Research and Experiences in Practice

During the last decade the documentation on PES experiences has been growing exponentially (see especially Landell-Mills and Porras, 2002 for an overview of 287 documented initiatives, or Ravnborg et al., 2007 for a recent extensive reference list). A large part of the literature is written from a neo-classical economic perspective by market instrument proponents who often take the creation of markets as desirable tout court (Landell-Mills and Porras, 2002). Although Engel et al. (2008) admit that PES is not a one-size-fits-all solution to every environmental problem and Kousky (2005) warns us for starting "to see every environmental problem as a nail waiting for the PES policy hammer", the creation of markets are increasingly considered as best or second-best solutions to a wide range of environmental externality problems.

According to Jordan et al. (2003), McAfee and Shapiro (2008), and McCarthy and Prudham (2004) this unquestionable pro-market promotion is inextricably associated with the rise of neoliberal rhetoric within supranational environmental policy-making institutions, emphasised by many PES adherents' claims that unattractive regulated nature conservation should be converted into alluring private business transactions (Salzman, 2005; Wunder and Vargas, 2005). As such, "market-oriented analysts see PES as one means of advancing the neoliberal goals of decentralising environmental management and of shifting control of resources from states to markets" (McAfee and Shapiro, 2008: 16). Especially the World Bank, which since the '90s has been working thoroughly at its 'greening' image, has been a fierce driving force behind the current global discourse of substituting unattractive and state-regulated conservation projects with profitable private business opportunities[1] (McAfee, 1999), following an approach which Robertson (2004) has described as using ecological arguments to expand market relations. To this end the World Bank has, among other things, promoted and co-developed various PES programs throughout the world[2], yet always with a special attention for the Latin American region[3] (Pagiola et al., 2005), where it also supported a pioneering PES investigation in landscape restoration: the RISEMP, which we will turn to in the following sections.

[1] McAfee (1999) has called these growing attempts to regulate natural capital flows through market institutions the rise of 'green developmentalism'.

[2] For an overview of World Bank sponsored PES programs, see the World Bank website http://web.worldbank.org/WBSITE/EXTERNAL/TOPICS/ENVIRONMENT/EXTEEI/0,,contentMDK:20487983~isCURL:Y~menuPK:1187844~pagePK:210058~piPK:210062~theSitePK:408050,00.html

[3] According to Pagiola et al. (2005) the main reason for this considerable interest in this region is attributed to the high dependence of the Latin American population on the environmental services and the protection provided by natural ecosystems, a fact which became increasingly recognised after the effects of Hurricane Mitch in 1998.

3. The RISEMP Description

3.1. Project Description and Site Characteristics

The RISEMP started in July 2002 and terminated in January 2008. The project, a full-sized GEF/World Bank initiative, was an innovative pilot experiment aimed at promoting silvopastoral practices[1] in degraded pasture areas through payments for environmental services (generated by these practices) and technical assistance. The targeted ES were biodiversity conservation and carbon sequestration. At a later stage, hydrological services were included as well. The project took place in three countries: Nicaragua, Costa Rica and Colombia, and was managed by the World Bank. Country sites were managed by local non-governmental organisations. The total cost of the project was an estimated US$8,7 million of which US$4,8 million was financed by a GEF grant and US$3,9 million through co-financing[2]. (GEF, 2007; Vaessen and Van Hecken, 2009; World Bank, 2002)

Despite their long-term on-site private benefits, silvopastoral practices often tend to be unattractive to farmers (Pagiola et al., 2007). The main barriers to the adoption of more environmentally-friendly silvopastoral practices are the relatively high requirement of investment in capital and labour and the long time lag between investment and higher productivity (Dagang and Nair, 2003). As such, the main objectives of the RISEMP were demonstrating and measuring (i) the effects of the introduction of PES to farmers on their adoption of integrated silvopastoral farming systems in degraded pasture lands; and (ii) the resulting improvements in ecosystems functioning, global environmental benefits, and local socio-economic gains resulting from the provision of said services (World Bank, 2002). Both issues were to a large extent unexplored territories of inquiry and thus illustrate the innovative nature of the project. But not only the objects of research were innovative; also the research methodology, which at least in theory was based on a randomized experimental design, with various participant groups receiving different incentives (payments and technical assistance) or no treatment (control group), was rather unique (Vaessen and Van Hecken, 2009). As such, through the use of scientific methods it was hoped to create a prominent precedent which would serve to promote further replication of the PES approach throughout the rest of the (developing) world.

In Nicaragua the project was executed by the research and development institute Nitlapán of the Universidad Centroamericana and the Costa Rican-based research centre CATIE, and took place in two micro-watersheds (Bulbul and Paiwas) in the central region of Matiguás-Río Blanco, department of Matagalpa, located at about 140 km northeast of the capital Managua. The site, which belongs to the so-called old agricultural frontier region, is situated in the buffer zone of the Cerro Musún natural reserve, and is also close to the Quirragua natural reserve. It has an undulating terrain, with an elevation of 300-500 m above sea level. It knows a semi-humid tropical climate, with average temperature about 25-30°C and average annual rainfall

[1] Dagang and Nair (2003: 149) define silvopastoral systems as "systems [that] integrate trees into livestock systems for multiple purposes including soil amelioration, shade, fodder, fruit, wood, and habitat for fauna". According to Pagiola et al. (2008: 304) silvopastoral practices include "(1) planting high densities of trees and shrubs in pastures, thus providing shade and diet supplements while protecting the soil from packing and erosion; (2) cut and carry systems, in which livestock is fed with the foliage of specifically planted trees and shrubs ('fodder banks') in areas previously used for other agricultural practices; and (3) using fast-growing trees and shrubs for fencing and wind screens."

[2] Following the categorisation described in section 2.2, the project is thus government-financed, trying to achieve global ecological benefits (with global GEF funds), rather than purely local benefits.

1300-2500 mm. (Pagiola et al., 2007; Yamamoto et al., 2007). The rainy season stretches from May to December. Until the beginning of the 20th Century, the area predominantly consisted of forests. However, increasing colonisation of the area, mainly by peasants in search of pasture land for their cattle, resulted in rapid deforestation from the 1920s onwards (Maldidier and Marchetti, 1996). In 2003, only 20% of the participating land[1] was covered with some type of forest, and most of the land was used as pasture for extensive grazing practices (about 63%), with about half of this degraded pasture. Silvopastoral practices were already used before project implementation: in 2003, pastures with high tree density covered 17% of the area (Pagiola et al., 2007), and fodder banks almost 3%. Nevertheless, the project tried to increase adoption of these practices through the application of different incentives.

3.2. Project Participants

The main hypotheses of the research that the adoption of silvopastoral practices can be attributed to PES, to technical assistance (TA) or to a combination of both, was investigated in Nicaragua by subdividing 123 households[2] (or 65% of total population in the micro-watershed of Bulbul, and 35% in Paiwas), mainly small to medium-sized farmers, into 3 different treatment groups (table 1). The largest part of the participants (98, or almost 80%) belonged to the PES group, which was further subdivided in two subgroups, depending on whether they received only payments (PES only) or also technical assistance[3] (PES + TA). In order to distinguish the effects of the different treatments, these groups would be compared to a counterfactual, which was established by assigning 25 farmers of the same region to a control group. During implementation, the use of a control group turned out to be practically impossible, given that it was politico-ethically unacceptable to exclude certain poor and medium-sized farmers from the project benefits in order to serve as the 'control group'[4]. Control group members therefore differed significantly from the treatment groups in many important characteristics (see table 1) and were generally chosen among more capitalized farmers (see also Vaessen and Van Hecken, 2009 for a discussion of this selection bias). Furthermore, the PES only and PES + TA groups were further subdivided into two payment schemes; 4-year scheme participants (n=72) would be paid during the whole project period, while farmers in the 2-year scheme (n=26) would only receive payments during the first two project years. This was done to monitor the land use practices of farmers after terminating the treatment, but within the project duration.

[1] The land use data cover the area that participated in the project, which according to Pagiola et al. (2008) corresponds to about 60% of the total area in Bulbul, and 40% in Paiwas.

[2] In fact the project started with 136 participants in the first year (2002), but due to various factors (such as decease or sale of participating farm) it ended with 123 remaining participants in 2008. In the further analysis we use data from the 123 remaining participants.

[3] Farmers receiving TA received help in adopting (new) land use practices and in applying (new) techniques, mainly focused on different aspects related to management of livestock and silvopastoral systems (World Bank, 2002). The TA component mainly consisted of monthly meetings and workshops (in which specific themes were discussed, such as sowing techniques, animal feeding, etc.), personal farm visits by the project staff (explaining certain practices on-field), and interchange of experiences (farmer-to-farmer knowledge extension: farmers visiting other farms, discussing certain experiences on old and new practices).

[4] One thing is indeed to design a scientific experiment behind a desk and quite another to implement it in the field where people tend to resist being treated as guinea pigs.

*Table 1: Project Participants by Treatment Group, 2002, Mean and Standard Deviation**

Treatment Group	Group Size	Area (ha)	Household Size	Number of Cattle
PES only	28	29,5 (25,1)	6,0 (2,7)	34,0 (35,5)
PES + TA	70	31,9 (25,8)	6,3 (2,5)	34,8 (30,9)
Control	25	46,7 (37,1)	5,3 (1,9)	53,6 (39,2)
Total	123	34,4 (28,8)	6,0 (2,5)	38,5 (34,4)

** Data refers to households that participated during whole project duration (n=123). Standard deviations indicated between parentheses.*

Source: Vaessen and Van Hecken, 2009

The project site is located in a region with a high degree of poverty (Levard, et al., 2001), which is reflected in a low average per capita yearly income of about US$340 (Pagiola et al., 2007), a low education level, and a limited access to basic services such as water and electricity. As can be seen in the same table 1, the average participating household is composed of six members, and possesses about 34 hectares of land, and almost 39 head of cattle. However, the high standard deviations for both area (28,8) and number of cattle (34,4) indicate large differences among households. It is therefore useful to subdivide the participants into three main types of farmers[1]. The poorest group of households, the so-called Campesinos Pobres con Tierra (CPT; poor peasants with land) possess a relatively small amount of land (maximum about 20 hectares), and generally lack capital to invest in self-sustaining agricultural production. They have a very small herd size of about 2 to maximum 10 animals. The richest group, the Finqueros Ganaderos (FG; cattle farmers), possess vast amounts of land, sometimes up to 150 or 250 hectares, and have large herds, with up to 200 or 300 head of cattle. The intermediate group of Campesinos Ganaderos (CG; cattle peasants) typically possess between 20 and 50 hectares of land, on which they keep around 20 to 100 animals. The latter two groups of households dedicate themselves almost exclusively to the breeding of dual purpose cattle, with only a small part of their land used for cultivation of basic staples (mainly corn and beans) for own consumption. The poorest group cultivates larger amounts of staple crops for own consumption, but often also depends on hiring out part of family labour to the other two -more capitalised- groups of farmers, and frequently also complements its income by renting out parts of its underused pastures to the more capitalised cattle farmers.

3.3. Monitoring and Payments for Ecosystem Services

Because the measurement and verification of the provision of ES, such as biodiversity conservation and carbon sequestration, is laborious and would imply huge transaction costs, the project worked with land use proxies likely to provide the desired ES (Pagiola et al., 2004; World Bank, 2002), and thus can be categorised as an area-based PES program. Furthermore, the RISEMP recognised that a whole range of different land uses could deliver different 'amounts' of 'bundled' ES. As such, an 'environmental service index' (ESI) was elaborated, which

[1] The categorisation into different types of farmers is based on Maldidier and Marchetti (1996) and Levard et al. (2001), updated by own field observations.

was based on the aggregation of the estimated per hectare contribution of 28 different land uses to biodiversity protection and carbon sequestration. Farmers' payments were calculated on the basis of the net increase of this ESI -which ranged from value 0 (land use least effective in providing the ES) to 2 (land use most effective in providing the ES)[1]- as compared to the baseline land use data for their farm in 2003.

Payments were made ex-post, annually after the observed land use changes. According to Pagiola et al. (2007) the amount of payment per ESI point was calculated on the basis of the opportunity costs of more attractive land uses. The payments as such did not necessarily cover the full opportunity cost, but "could 'tip the balance' of profitability between current and costlier silvopastoral practices, by increasing the net present value of investments in silvopastoral practices and by reducing the initial period in which these practices impose net costs on farmers" (ibidem: 378). As such, the realised payments were not designed to compensate the full cost of the provisioning of the ES, but were meant to promote the introduction of silvopastoral practices by altering the cost-benefit relationships of alternative investment strategies, and calculating the payment net of the opportunity cost of the promoted land use (Pagiola et al., 2005). It was hoped that once these practices were adopted by the farmers and productivity would have increased, they would continue to use or even expand them after project termination (Pagiola et al., 2007). Based on calculations of the relative profitability of more attractive common practices, the payments were established at an annual US$75 (4-years scheme) and US$110 (2-years scheme)[2] per incremental ESI point. In order to eliminate perverse incentives (for example for farmers to cut down existing trees so as to raise the potential additional payments during project implementation) the baseline ESI points in 2003 were remunerated with a one-time initial payment of US$10 per ESI point (Pagiola et al., 2005).

[1] For a detailed explanation and overview of this ESI composition, see Pagiola et al. (2004) or Pagiola et al. (2007). For an overview and description of the different types of land uses, see Murgueitio et al. (2003) and World Bank (2002).

[2] According to project staff, this higher payment for the 2-year scheme was established to compensate these farmers for their shorter participation time.

4. Assessment of the RISEMP Results

This section deals with the main results of the RISEMP. Based on both the official project data and on our further inquiry -which is briefly explained in a first subsection-, it focuses especially on the implemented land use changes and the underlying motivation for these changes.

4.1. Research Methods

In order to enquire into the motivations of farmers (not) to change land uses, and more specifically into the role of PES as a potential incentive for these changes, we relied on different sources of primary and secondary data. First, we analysed the main surveys conducted during the project period by the CATIE, Nitlapán and World Bank staff. These surveys resulted in two main data sets: (i) a baseline data set (2002) and corresponding yearly socio-economic follow-up datasets, with detailed information on household characteristics and economic activities over the whole project period; and (ii) a land use data set, with an overview of the land use changes per farm (based on remote sensing imagery[1]) and the corresponding ESI scores and payments during the whole project period. Secondly, we analysed different internal and published documents on the project. Thirdly, we conducted an extensive field study after project termination. This field study consisted of (i) in-depth responsive interviews (Rubin and Rubin, 2005) with a sample of 35 former participating farmers during the month of April 2008; and (ii) in-depth interviews with the Nitlapán project staff in July 2008. For further considerations on the used methodology we refer to appendix 1.

4.2. Land Use Changes

One of the main yardsticks the RISEMP used to monitor its results was the change in land use practices among project participants and the control group. To this end, it subdivided the land uses into different categories, based on their potential contribution to the provision of the targeted ecosystem services. The results of the extensive land monitoring are displayed in table 2.

[1] Quickbird imagery with 61cm resolution. The corresponding land use maps derived from these images were ground-truthed to match each plot to one of the 28 different land uses recognized by the project (Pagiola et al., 2007).

Table 2: Land use RISEMP participating households (n=122), 2003-2007

Year / Land Use	2003		2004		2005		2006		2007		∆ 2003-2007	
	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%
Crops	310	7.4	207	4.9	146	3.5	130	3.1	123	2.9	-187	-4.4
Degraded pasture	1306	30.9	693	16.4	537	12.7	468	11.1	425	10.1	-881	-20.8
Natural pasture without trees	53	1.3	129	3.1	129	3.1	94	2.2	74	1.8	20	0.5
Improved pasture without trees	36	0.8	56	1.3	63	1.5	47	1.1	42	1.0	7	0.2
Semi-permanent crops	44	1.1	32	0.8	29	0.7	35	0.8	25	0.6	-19	-0.5
Natural pasture with trees	912	21.6	1179	27.9	1067	25.3	1088	25.9	1081	25.7	169	4.1
Improved pasture with trees	382	9.0	656	15.5	873	20.7	933	22.2	1002	23.8	621	14.8
Fruit crops	21	0.5	23	0.5	25	0.6	26	0.6	25	0.6	4	0.1
Fodder banks	104	2.5	178	4.2	227	5.4	288	6.9	324	7.7	220	5.2
Commercial tree plantations	1	0.0	3	0.1	5	0.1	12	0.3	5	0.1	4	0.1
Shaded coffee	2	0.1	3	0.1	20	0.5	9	0.2	6	0.1	4	0.1
Scrub habitats (tacotales)	221	5.2	211	5.0	234	5.5	216	5.1	207	4.9	-14	-0.3
Riparian forest	540	12.8	560	13.3	554	13.1	551	13.1	551	13.1	11	0.3
Intensive silvopastoral	4	0.1	1	0.0	6	0.2	9	0.2	11	0.3	7	0.2
Secondary forest (intervened)	184	4.4	195	4.6	184	4.4	189	4.5	183	4.3	-2	0.0
Secondary forest	43	1.0	45	1.1	63	1.5	61	1.5	65	1.5	22	0.5
Primary forest	41	1.0	37	0.9	40	0.9	33	0.8	37	0.9	-3	-0.1
Infrastructure, housing , roads	15	0.4	13	0.3	18	0.4	14	0.3	19	0.5	4	0.1
Total area	4221	100	4221	100	4221	100	4203	100	4206	100	n.a.	n.a.
Living fences*	127	n.a.	284	n.a.	448	n.a.	448	n.a.	479	n.a.	352	n.a.
Total forests	808	19.1	837	19.8	841	19.9	834	19.9	836	19.9	28	0.7
Total pasture	2693	63.8	2713	64.3	2676	63.4	2639	62.8	2636	62.7	-58	-1.1

** expressed in kilometres instead of hectares*

Source: Authors' own elaboration based on project data

Table 2 reflects the observed changes in land use. While the total area of pasture has remained stable (from 63,8% in 2003 to 62,7% in 2007), its composition has changed significantly. Degraded pastures decreased from 30,9% of total area in 2003 to 10,1% in 2007. They were replaced by improved pastures with trees (an increase of 621 ha, from 9% of the total area in 2003 to 23,8% in 2007) and fodder banks (which more than tripled in use, from 2,5% in 2003 to 7,7% in 2007). Annual crop areas (mostly for the cultivation of corn and beans) more than halved during the project period, while living fences almost quadrupled from 127 to 479 km. Forests and scrub habitats (tacotales) remained rather stable during the whole project period, covering about 25% of the total area.

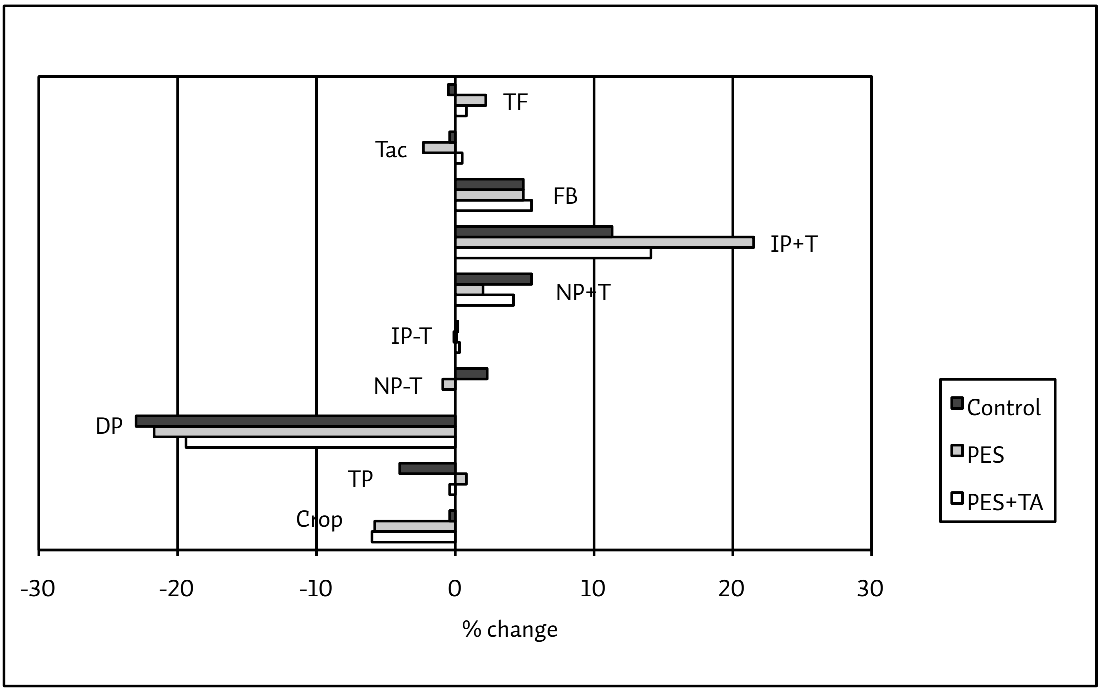
The most significant changes have occurred during the first two project years, mainly because the 'novelty' of the project incentives, the limited scope for improvements per farm (law of diminishing returns), and because about 26% of the participants from the two-year scheme only participated during the first two years. The system of payments, which each year repeatedly remunerated the incremental ESI points as compared to the 2003 baseline also stimulated farmers to change their land use as early as possible since this implied higher payments. Degraded pasture, for example, almost halved between 2003 and 2004 with a decrease of 613 ha, while it decreased by an additional 155 ha during the second year and with 'only' 111 ha during the last two years together. The same pattern, but in the opposite direction, occurred with living fences and improved pasture with trees, which increased over the whole project period, but -again- mainly during the first two years. Fodder banks increased rather constantly over the whole project period, with an average of 55 ha per year.

4.3. Project Incentives

The project aimed to motivate farmers adopting silvopastoral practices and to promote more environmentally-friendly land uses through various types of incentives. Simultaneously, it monitored the changes induced by every type of treatment. The project incentives can be subdivided into three categories: payments (PES), technical assistance (TA), and no incentives. Subdividing the project participants into three main groups receiving different (mixes of) incentives, it was hoped to attribute land use changes to the separate incentives (see also Vaessen and Van Hecken, 2009). Hypothetically, the group that would score best on the adoption of silvopastoral practices would be the group receiving both PES and TA, followed by the PES group, and ultimately the control group, which would only marginally adopt the targeted practices, or could be considered as the yardstick of the business-as-usual situation in the region. Furthermore, it was expected that farmers, triggered by the higher corresponding payments, would mainly invest in land uses with higher ESI scores, such as forests or scrub habitats (tacotales).

Figure 1 gives an overview of the land use changes among the treatment groups. It indicates that degraded pastures have decreased in all groups, with the highest reduction paradoxically in the control group (-23%). Also, living fences have increased most in the control group, eight times more than in the PES group. Fodder banks have known a very similar pattern among groups, while natural pastures with and without trees have known the highest increase in the control group.The establishment of improved pastures with trees however has been highest in the treatment groups (respectively 14,1 and 21,5 per cent for the PES+TA and PES group), with an increase of 'only' 11,3 per cent in the control group. The control group is also the only group which decreased its forest area (-0,5 per cent). The amount of scrub habitats has slightly increased in the PES+AT group (0,5 per cent), while it has slightly decreased in the control and the PES group.

Figure 1: Changes in land use, per treatment group (*), 2003-2007



	Crop	TP	DP	NP-T	IP-T	NP+T	IP+T	FB	Tac	TF	LF**
PES+TA	-6	-0.4	-19.4	0	0.3	4.2	14.1	5.5	0.5	0.8	213.5
PES	-5.8	0.8	-21.7	-0.9	-0.1	2	21.5	4.9	-2.3	2.2	164.4
Control	-0.4	-4	-23	2.3	0.1	5.5	11.3	4.9	-0.4	-0.5	1364.4

** Changes in land use: calculated as the additional percentage of the selected land use within the total land size of every treatment group.*
*** The % for LF is calculated as an increase in the length, compared to each group's initial LF length in 2003.*
Legend: Crop = annual crops; TP = total pastures; DP = degraded pastures; NP-T = natural pastures without trees; IP-T = improved pastures without trees; NP+T = natural pastures with trees; IP+T = improved pastures with trees; FB = fodder banks; Tac = tacotales (scrub habitats); TF = total forests; LF = living fences.

Source: Authors' own elaboration based on project data

These data clearly confirm that farmers have recognised the benefits of at least some silvopastoral practices and that they have increasingly adopted these practices. We believe however that the comparison of land use changes among different treatment groups does not provide unequivocal confirmation for the RISEMP hypothesis (Pagiola et al., 2005; World Bank, 2002) that observed changes are exclusively attributable to the payment incentive (see also Vaessen and Van Hecken, 2009).

Pagiola et al. (2007) argue that the similar results of treatment and non-treatment groups are attributable to the poorly chosen control group, which mainly consisted of more capitalised farmers from the FG and CG group (see also section 3.2). Thus, they concluded that the data on the control group were not useful, and decided to exclude them in their analysis. We recognise this problem with the control group, but do not think it justifies ignoring the information on this group alltogether. In fact, these data on the control group –maybe precisely because it is biased towards the less capital-constrained, richer farmers- are quite useful, because they indicate that there are complementary incentives triggering farmers' motivations to adopt silvopastoral practices. An alternative distinction among participants and control group on the basis of types of farms helps to shed more light on the reasons why certain groups have (not) adopted certain practices, and how this emanates from different opportunity costs and livelihood strategies among types of farmers.

4.4. Motivations and Incentives to Adopt Silvopastoral Practices

Costs to establish silvopastoral practices are relatively high. In the study region, they range from US$170/ha for sowing improved pasture on degraded pasture areas to about US$390/ha for converting degraded pasture into improved pasture with high tree density (>30 trees/ha). Establishment of fodder banks ranges from US$170 to US$270/ha. Establishing living fences costs about US$110-160/km (Pagiola et al., 2008). Their on-farm benefits are mainly linked to increased carrying capacity and thus higher milk and meat productivity. Yamamoto et al. (2007) showed that silvopastoral practices in the project region have effectively led to higher milk production, mainly through the establishment of pastures with moderate tree density (tree cover approximately 20%).

So what has been the precise effect of payments? Suppose a participating farmer decides to convert 1 hectare of degraded pasture to an improved pasture with high tree density. The establishment costs in this case add up to US$390/ha. The payments the farmer will receive the first year after establishing the improved pasture can be calculated as: (Δ ESI/ha x number of ha x US$75/point or US$110/point), which in this case equals 1,3 points/ha x 1ha x US$75/point (or US$110/point) = US$97,5 or US$143. In other words, the payments after the first year in this case cover only 25% to 37% of the establishment costs respectively. If the changes were implemented from the first year onwards, an additional part of the initial investment cost would be recuperated through the payments in the following years (adding up to a full -4 year payments scheme- or a 75% -2 year payment scheme- cost recovery). Similar results can be found for other land use changes. Since payments were ex-post, costs had to be pre-financed by the farmers, some of which to this end recurred to local microfinance institutions[1]. Payments were designed to alleviate the investment constraint and shorten the time lag to break-even, thereby convincing farmers of the long-term benefits of silvopastoral practices[2]. The capacity to access higher project payments however hinged on the farmers' capacity to pre-finance the investments. This creates a tendency towards the exclusion of the poorer farmers (mainly from the CPT group, see section 3.2), which have more difficulties to self-finance or to secure loans. As we will further discuss in section 4.5 this unintended implicit 'bias' against poorer farmers is not only socially undesirable, but also raises doubts about the environmental effectiveness of the project; The unintended exclusion of the poorer farmers certainly does not contribute to a reduction of the advancement of the agricultural frontier, which is at least partially the consequence of migrating poorer farmers, selling their relatively less productive land (for lack of capital investment) in regions like Matíguas-Rio Blanco to richer farmers in order to buy cheaper, unexploited land further down the frontier.

[1] In 2003 more than 50% of the participants were found to have used credit in the past five years. Interests typically fluctuate around 20%/year.

[2] Asked for how they financed investments in land use changes during the project, most farmers answered they had recurred to credits or had sold some of their cattle to obtain the required inputs.

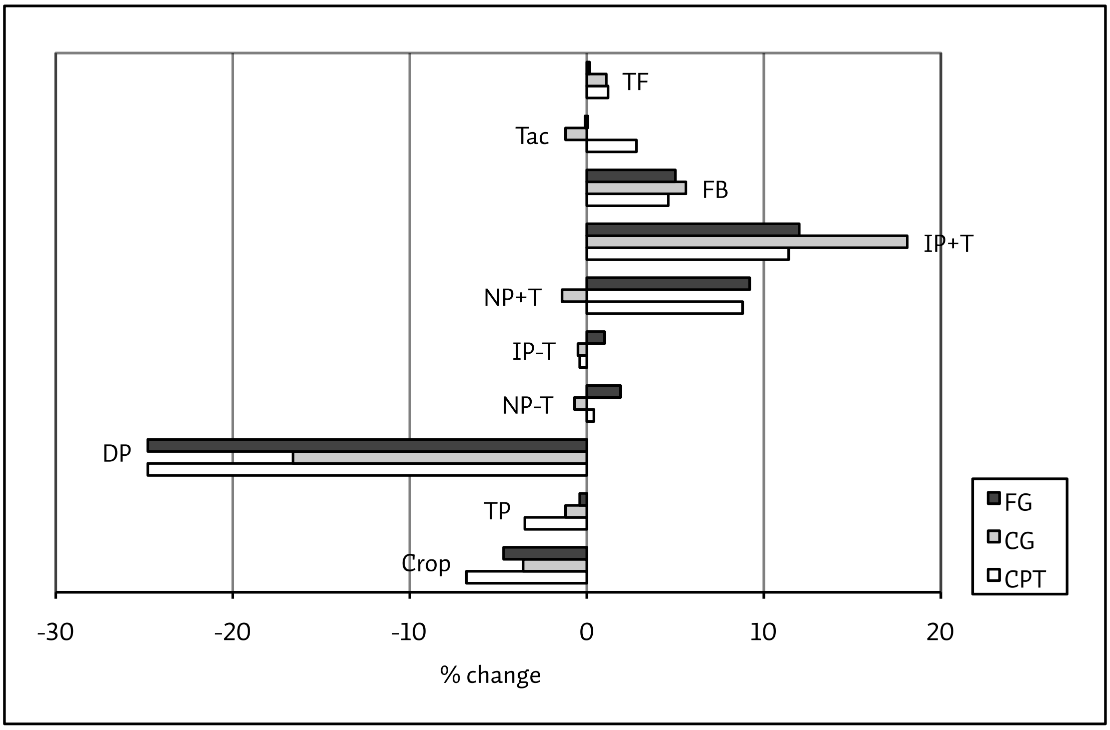
Together with the observation that control group participants - mainly consisted of more-capitalised farmers- also increasingly adopted silvopastoral practices, the foregoing discussion also indicates that the motivations to invest in the promoted land uses, are primordially related to exogenous processes and corresponding production strategies within the region Thus, the RISEMP PES payments are to be considered a kind of secondary incentive which make these processes more attractive at the margin, while also relaxing the capital constraints of some poorer and medium-sized farmers. In order to further inquire into this important observation, we turn to our qualitative data, and make a distinction between production strategies of the three main groups of farmers.

4.4.1. Differing Constraints among Different Types of Farmer Households

Figure 2 retakes the main land use changes, this time subdivided according to the type of household (CPT, CG and FG). The figure indicates that land use changes differ among different types of farmers. All farmers have invested in improved pastures and fodder banks, mainly by transforming degraded pastures to these more productive uses. Living fences have also been very popular among all groups[1]. The relative changes in productive uses (mainly improved pastures and fodder banks) have been highest in the more capitalised groups (FG and CG). But it is the CG group that accounts for the most significant intensification (an additional 5,6% of land in fodder banks and 18,1% in improved pastures with trees), and the lowest decrease in degraded pasture (-16,6%).

[1] This follows quite logically out of the fact that –apart from the productive benefits once established- living fences are up to 9 times cheaper to establish than 'dead' fences (Pagiola et al., 2008) and require far less labour.

Figure 2: Changes in land use, per type of household (*), 2003-2007



Land use / Group	Crop	TP	DP	NP-T	IP-T	NP+T	IP+T	FB	Tac	TF	LF**
CPT (n = 32)	-6.8	-3.5	-24.8	0.4	-0.4	8.8	11.4	4.6	2.8	1.2	167.6
CG (n = 67)	-3.6	-1.2	-16.6	-0.7	-0.5	-1.4	18.1	5.6	-1.2	1.1	263.0
FG (n = 23)	-4.7	-0.4	-24.8	1.9	1.0	9.2	12.0	5.0	-0.1	0.1	382.5

** Changes in land use: calculated as the additional percentage of the selected land use within the total land size of every household type.*
*** The % for LF is calculated as an increase in the length, compared to each group's initial LF length in 2003.*
Legend:
CPT = campesino pobre con tierra (poor peasant with land); CG = campesino ganadero (cattle peasant); FG = finquero ganadero (cattle farmer); Crop = annual crops; TP = total pastures; DP = degraded pastures; NP-T = natural pastures without trees; IP-T = improved pastures without trees; NP+T = natural pastures with trees; IP+T = improved pastures with trees; FB = fodder banks; Tac = tacotales (scrub habitats); TF = total forests; LF = living fences.

Source: Authors' own elaboration based on project data

At first sight it seems quite astonishing how the land use changes of the FG group resemble those of the much poorer CPT group (both a similar decrease in the amount of degraded pastures and annual food crops, and increase in fodder banks and in natural and improved pastures with trees). FGs and CPTs have invested relatively less than CGs in fodder banks and improved pastures, and have relied more on the use of natural pastures. The similar manifestations of land use changes have, however, different origins. The lower adoption rates of the more productive and intensive land uses (mainly improved pastures and fodder banks) among the CPT group reflect the often-experienced limitations they have, mainly due to labour, space and capital constraints[1]. During our interviews, for example, we found that smaller farmers often had tried to invest in fodder banks, but due to the long time lag before exploitation and little other

[1] Nevertheless, fodder banks adoption rates are amazingly high among the poorer farmers, which is probably attributable to this group's lower opportunity cost of labour; the establishment of fodder banks has a relative high labour capital ratio of between 1,3 and 4,8. Improved pastures, for example, have a much lower labour capital ratio of approximately 0,5 (calculations based on project data).

available pastures to bridge this time lag, they were often forced to allow their cattle to enter the newly established fodder banks, which meant destruction of the latter. This also explains the relative large increases in natural pastures, which do not require high capital and labour inputs, and save precious labour time for crop production or off-farm employment. In the case of the FGs the reliance on natural pastures and relative lower increases in improved pastures and fodder banks can be explained by the relative abundance of land of these farmers, which makes labour the scarce input factor, and stimulates the extensive use of land, with minimal investments in land intensification. However, this land extension strategy has been changing during the last few decades (see further below), which explains the -smaller than CGs, but nevertheless positive- increases in the adoption of more intensive practices.

Quite contrary, increases of areas in scrub habitats (tacotales) have only taken place in the case of the poorest CPT group, an observation which should not be much of a surprise; since this group's main constraint is capital availability and since parts of the land are underutilised mainly because of these capital constraints, a cheap way to obtain project payments without having to invest much capital, nor labour, has been the simple regeneration of 'low-pressure' or underutilised parts of the farmers' land into scrub habitats. Considering that this land use is to be converted regularly into new productive land uses, and is used as a natural fertiliser for future crop production, the project payments might have just created somewhat more space for letting land fallow for a longer time.

Finally, the general decline in food production (annual crops) among all groups could be an indicator that relative prices of other agricultural activities in the region have changed (mainly towards dairy production, see below) and that labour was increasingly (and temporarily) needed for activities related to these more economically interesting activities (improving pastures and establishing fodder banks).

4.4.2. Moving beyond Direct Project Incentives: Revealed Motivations for Land Use Changes

In our field interviews most farmers expressed to have changed land use for several reasons. Payments were a welcome additional incentive, but according to them did not play the decisive role. Surprisingly, they attribute much more importance to the provision of the technical assistance, which deepened their knowledge on silvopastoral practices and more importantly strengthened collective motivation to engage in the pathway of silvopastoral intensification, articulated to the on-going milk-dairy boom (see below). Various other studies have emphasised the importance of TA in the adoption of silvopastoral practices (see Pattanayak et al., 2003 for an overview). Although most farmers in Matiguás-Río Blanco already knew some of the silvopastoral techniques and the majority applied them to some extent before project implementation, the technical assistance and the social momentum is held to have offered a stimulus to experiment with new or expand already known practices, and at the same time to have lowered the perceptions of risks. At first sight, the quantitative project data (

Figure 1) do not seem to provide evidence that TA played such a decisive role. Comparison between the PES and PES+TA group even reveals that adoption rates were somewhat higher in the group without technical assistence. However, since the experiment did not take place in a laboratory, but in the real world, where farmers cannot simply be isolated from other interconnected community members, participants (not receiving TA) inevitably interchanged newly acquired knowledge by interacting with other participants (receiving TA), experienced demonstration effects from neighbouring farms, and often even attended workshops as substitutes for eligible participants (with TA) (Vaessen and Van Hecken, 2009). Also, several other extension organisations were active in the region, some joining the RISEMP momentum and offering similar services as the RISEMP. Rather than being an indication that TA did not have effect, this precisely suggests that adoption of new practices is not solely dependent on individual cognitive capacities and decision-making, but also supported by the emergence and articulation of sufficient social momentum crystallising into coherent collective action that enables widely shared joint pathways of change (de Haan and Zoomers, 2005).

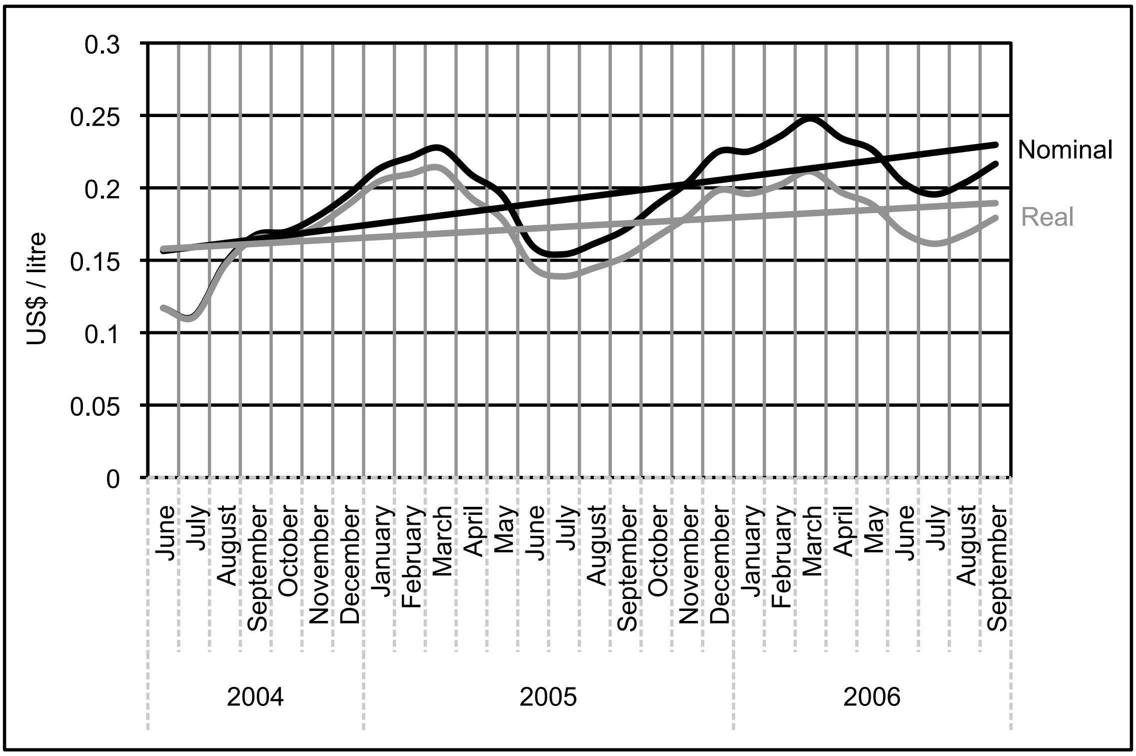
But apart from the interconnected individual economic and shared social incentives induced by the project, there have also been strong exogenous incentives which have -contrary to the more traditional production logic in the region, which mainly favoured extensive land use among all three types of farmers[1]- motivated farmers to intensify their farms. The main incentive derives from the boom of the national milk market and improved local access of farmers to fresh milk collection centers and (semi)industrial cheese factories. This translated in high demand for milk (volume), a significant increase of regional milk prices and the widespread perception that this rise will continue in the future, in particular as a consequence of a series of free trade agreements that will further reduce barriers to lucrative export markets, including the USA. Until the '70s most communities in the study region were relatively isolated and lacked access to milk storage facilities, which made the commercialisation of milk unattractive. Increasing interest from international dairy companies through the '80s and '90s led to a rapid growth of roads and basic milk storage facilities, which opened up the milk market for many farmers and led to a relative intensification of cattle production. In the late '90s Salvadoran traders further commercialised the dairy production and offered higher, but seasonally fluctuating milk prices. (Levard et al., 2001). The increasing presence of local agricultural extension and development organisations has further promoted and developed milk commercialisation among farmers.

Figure 3 shows the evolution of milk prices (in US$/litre), both in nominal and in real terms, in the project region for the period June 2004-September 2006. The cyclical movement represents the seasonal character of milk market prices, while the linear trend line suggests that prices are in an upward evolution. Comparisons between different years reveal an increase of almost 10% in nominal prices in the positive peak price months of March 2005 and March 2006 (dry season), which however corresponds with a real price decrease of 1%; and a nominal increase of 38% and 27% (real increase of 25,3% and 16,2% respectively) between the negative peak months of July 2004 and July 2005 and July 2005 and July 2006 respectively (rainy sea-

[1] Traditionally it has never been very interesting for the three types of farmer households to intensify their land use, but each group for different reasons. Generally, CPTs maintain extensive land use practices (with low cattle carrying capacity of pastures), because they lack capital to intensify their land use. CGs and FGs, on the other hand, generally base their production logic on the maximisation of the relative scarce input factor labour, which in practice translates into extensive cattle breeding, with –at least in the absence of attractive dairy markets- little motivation for land use intensification, and a constant drive to purchase more land, which is the cheapest way to enhance productivity. (Maldidier and Marchetti, 1996).

son)[1]. Compared to the changes in some important agricultural input prices, these changes in milk prices are substantial; for example, the cost of barbwire, an important input for managing pastures and fodder banks, decreased with 6% in real terms between 2004 and 2006, while herbicides (used for 'cleaning' pastures from undesired vegetation) decreased by almost 13% in real terms. The real cost of hiring day labourers decreased with more than 10% in this same period.

Figure 3: Nominal and real milk prices (in US$/litre) to farmers in Matiguás-Río Blanco, 2004-2006



Source: Authors' own elaboration based on project data

The emergence of and the better access to dairy markets and the corresponding higher milk prices, make intensification of land use through the adoption of silvopastoral practices more attractive for farmers. In order to benefit from these opportunities, dairy farmers permanently need to keep (and feed) their milking cows close to the roads (where the milk is collected), dissuading extensive land use in cheaper interior areas and rather favouring land intensification near milk collecting centres and access roads. Milk collection centers, owned or connected to cheese exporters and national processing plants that are strongly interested in maintaining year-round stable production, also pay significant price premium for a stable supply of milk. The latter requires avoiding the usual decline in milk production as a consequence of lower food availability during the dry season. Furthermore, the more productive milk cattle breeds are also less resistant than the traditional meat cattle breeds to heat and therefore require and benefit from the protection of trees. Since easy farm access was one of the inclusion criteria during the project's participant selection phase, and since the majority of the participant's main economic activity is oriented to dairy production, it should not be much of a surprise

[1] Milk prices could potentially rise even more in some communities where purchase is currently mainly controlled by middle-men.

that most farmers have increasingly adopted silvopastoral practices which allowed them to intensify their land use.

Empirical evidence on the local relation between milk production and land use practices was provided by a study of Yamamoto et al. (2007), who analysed the link between milk production and silvopastoral practices in the project region. One of their conclusions is that an increase in the meat milk price ratio is likely to decrease tree cover, or reversely, a decrease in this ratio will probably foster the adoption of certain silvopastoral practices, such as tree planting in pastures. Our interviews with participants confirmed the perception among farmers of certain silvopastoral practices having positive effects on cattle production, especially by an increase in milk productivity during dry season (for example by planting trees which provide cattle with shade and avoid dehydration and weight loss) and a higher carrying capacity of their pastures. Together with the observation that since various decades farmers in this region are often affected by animal alimentation scarcity during dry season (Maldidier and Marchetti, 1996), and a growing perception among farmers of more extreme climate conditions (longer and more severe dry seasons) during the last five to ten years, these factors have increasingly motivated farmers to try to change animal alimentation strategies, which in practice often means recurring to more drought resistant types of pastures and fodder banks. As such, farmers who recur to silvopastoral practices potentially benefit disproportionately more from milk price increases than other farmers. Project data revealed that between 2004 and 2006 daily milk production remained rather stable during the rainy season (on average 3,9 litres/cow), but increased with almost 10% during the dry season (from 3,3 to almost 3,7 litres/cow); according to interviewed farmers a consequence of improved animal alimentation during summer, which tempers the differences in cattle productivity between both seasons.

Griffith and Zepeda (1994) and Nicholson et al. (1995), however, emphasise that higher milk prices do not necessarily lead to land use intensification, but could rather "tend to bring more land into production" (Griffith and Zepeda, 1994: 130), and as such "do not necessarily reduce the total land in cattle production, although it likely increases total output" (Nicholson et al., 1995: 731). Especially if farmers have access to cheap land in close vicinity, intensification is not a profitable alternative. In a research study of three Latin American countries, White et al. (2001) found that land price is one of the main drivers in farmers' decision of investing in improved forage technologies. They stress that increasing land prices make it harder for farmers to increase their farm size, mainly because of capital constraints. "Instead, they adopt improved forage alternatives to enlarge their herds for less money." (ibidem: 103).

The former reasoning is perfectly applicable to the RISEMP study site. In Matiguás-Río Blanco land prices have been increasing considerably during the last two decades. Based on our interviews and existing project data, it is estimated that real land prices have increased with slightly more than 100% between the end of the '90s and today, heavily constraining the option of further land expansion within the region. These higher land prices, which are attributable to increasing population pressure, but are mainly reflections of growing (dairy) market opportunities and access, and the gradual disappearance of non-exploited land in close vicinity, have led to a general perception among farmers that land use intensification is a necessity. In other words, the land constraint, which in the past could be partly evaded by expanding the agricultural frontier, has become increasingly binding (reflected in an increasing land labour price ratio), also for farmers with relative high amounts of land (CGs and FGs). Consequently, one of the most obvious ways to enhance productivity under this constraint is the establishment of higher productive uses per unit of land (White et al., 2001), mainly through the implementation of improved pastures and fodder banks.

In conclusion, even though payments covered a substantial part of the investments costs and probably have played a positive role in motivating and enabling beneficial land use change, the motivation for this land use change has to be located in a broader process ensuing from the exogenous incentives created by the milk boom and the related social momentum of knowledge creation and social learning. This provides an indication that farm decisions, evidently affected by market conditions and individual cost-benefit calculus, are also dependent on broadly diffusing change processes that are collectively deemed beneficial for economic and possibly other reasons. Evidently, the payments have been welcomed by all farmers, and they have probably been both a real as well as a symbolic factor in creating collective and individual motivation for silvopastoral intensification. Yet, in the absence of credit constraints many of the investments would probably have taken place anyway, precisely because they 'made sense,', economically and socio-culturally. In our interviews, farmers indeed claimed that the project did not alter their farm management strategies and that it only promoted the faster adoption of practices which they were already adopting -or at least trying to adopt. Project payments thus appear to have played a less prominent role in behavioural change than expected or claimed, which more or less concurs with the claim of Scherr (1995: 788, emphasis in original) that "financial profitability, per se, may be of a secondary importance" in the adoption of silvopastoral practices. Following de Haan and Zoomers' (2005) pathway concept, it is much more probable that the high adoption rate, especially during the first two years of the project, was not triggered by simply offering individual payments to farmers, but rather also by the introduction of a new project in the region, which through the promotion of a new environmental narrative and the provision of TA and new knowledge and its unintentional farmer-to-farmer extension spill-over, legitimated existing but uncertain perceptions and discourses and as such generated a collective momentum that 'tipped the balance' in favour of adopting the practices which were already known to a great extent by the farmers before project implementation. The promoted practices were largely compatible with the farmers' livelihood strategies and represented the most logical economic trajectory for relaxing emerging constraints within the newly-emerging collective pathway. This also explains the high adoption rate among control group participants and the observation that technical assistance was perceived as the highest valued project component.

4.5. The Link with Ecosystem Service Provision

Silvopastoral practices are broadly considered to produce both productivity and environmental benefits. The previous section already dealt with the productive part. Additionally, the RISEMP tried to gain more site-specific knowledge on the effects on the provision of the targeted ES. Systematic monitoring at various project sites in Nicaragua showed positive effects of certain silvopastoral practices and corresponding land uses on carbon sequestration (Ibrahim et al., 2007) and on certain biodiversity indicators, such as bird richness (Sáenz et al., 2007). Additionally, the effects on water quality and quantity were monitored and found positive as well (Cárdenas et al., 2007; Ríos et al., 2007). As such, it seems that the ESI tool could be effectively used as a yardstick for the environmental service improvements in the region. Over the whole project period the ESI score increased with slightly more than 48% (from 3005 in 2003 to 4467 in 2007). The carbon index increased in this same period by 47% and the biodiversity index by almost 50%[1]. An important question then arises; should we conclude from these numbers that the project region is offering almost half as much of the targeted ES in 2007 as compared to 2003? This question will be discussed in section 5.1.

Nevertheless, it would be rather short-sighted to limit the analysis merely to a change in the project ESI score. Often it is argued that the intensification of productive land uses could relieve pressure on the agricultural frontier and existing ecological land uses, such as forests and scrub habitats[2], and in this way foster biodiversity conservation and other ES (see for example Green et al., 2005). This presumption was also made in the RISEMP: the intensification of productive land, it was believed, would lead to a decrease in space needed for cattle, which could liberate important and fragile parts of the farmers' land and even stimulate reforestation. (World Bank, 2002). While very appealing in theory, in practice this potential effect is still widely debated, mainly because of contradicting empirical evidence (Dorrough et al., 2007; Kaimowitz and Angelsen, 1998; Swift et al., 2004). In the case of Matiguás-Río Blanco the actual effect is not very clear. At first sight it seems that the overall amount of land dedicated to forests and scrub habitats has remained rather stable (see table 2), which could be a sign of reduced pressure. However, since the project limited monitoring land use changes only to the participating farms, it did not account for possible intra- and inter-regional leakage effects, in which environmentally-damaging activities are shifted elsewhere in space[3]. Soaring land prices in the project region have pushed at least some -mainly smaller- farmers to sell their farm and move eastward to the new agricultural frontier where land is still relatively cheap and unexploited, questioning the overall sustainability of the promoted measures. Furthermore, the lack of detailed pre-project land use data, which impedes contextualising land use changes into a potential broader evolution, makes it difficult to assess the exact influence of the project incentives on pressure relief[4].

It is very probable that the promotion of silvopastoral practices has only led to a temporal break on local environmental degradation, without dealing with broader (non-local) issues, such as the negative spill-over effects on neighbouring regions, mainly caused by poorer farmers hoping to find new (and cheaper) opportunities in the new agricultural frontier. As long as there is abundant and cheap land available in the vicinity and without a significant change in the underlying drivers of agricultural expansion, the overall positive effects of silvopastoral practices or other practices which can lead to land use intensification might at least have mixed results, in particular if –as we indicated- the actual adoption (and PES subsidies) favor the mid-

[1] Subdivision in treatment groups of the change in ESI/hectare between 2003 and 2007 revealed that the PES group scored best (53,9% increase), while the control group had the lowest relative increase of 'only' 42,2%. Subdivision in types of households revealed that the poorest group (CPT) increased most per hectare (56,7%), closely followed by the FG group (56%). The CG group scored lowest with an increase of 42,1%.

[2] In fact, scrub habitats should not be interpreted as fixed parts of the landscape being in transition to secondary forests, but as cyclical and temporary land uses which serve to recuperate soil fertility in the agricultural production process (Maldidier and Marchetti, 1996).

[3] A good example is that several participants possessed various pieces of land, but could participate with only one of them. The non-participating pieces of land (in and outside the project study region) were not monitored, which allows for leakage effects during project period in order to obtain a better ESI score at low cost.

[4] It seems that pressure on ecological land uses had already approached a saturation point before project implementation; most interviewed farmers asserted leaving a certain amount of ecological land uses on their farm (usually pieces of forests on parts of their land which are close to water sources or which are difficult to cultivate or convert into pastures, for example because of steep slopes), mainly for the provision of different ecosystem goods (such as firewood and construction materials) and perceived on-farm services (such as water provision). We would call these 'protected' areas on their land critical ecological parts.

dle-sized and richer farmers[1] Analysis of the precise effects of certain policies needs to assess environmental effects beyond direct observable outcomes at the project site and move beyond the technical or 'engineerical' closed-system approach. It has to take into account potential leakage effects and (future) farm production strategies[2]. In the Matiguás-Río Blanco region, for example, the main future plans of bigger cattle farmers (mainly FGs and CGs) are first intensifying the land and improving dairy cattle and pasture quality (through the introduction of genetic improvements and silvopastoral practices), and later on, when a new productive optimal equilibrium is perceived to be reached, the goal is expanding production by buying more land and cattle. Poorer farmers, mainly from the CPT, but also from the CG group, often mentioned their first priority was obtaining more cattle, and then expand their land size or move to the new agricultural frontier, hoping to find better conditions to expand their cattle activities.

[1] White et al. (2001) emphasise that intensification of pastures is only interesting for a farmer if this option is cheaper than simply expanding pasture size: "More intensive technologies will only help maintain forest cover if they are a less expensive option than extensive growth" (ibidem: 92). Vosti et al. (2001) show that pasture intensification in the Western Brazilian Amazon does not automatically halt deforestation and can even increase pressure on forested lands: "Greater profitability will create a demand for larger milking and beef cattle herds and pasture to support them" (ibidem: 129). As such, the promotion of silvopastoral practices, which increase cattle productivity per unit of land, could even be considered as a potential pressure on further deforestation.

[2] Nicholson et al. (1995), for example, recognise these leakage effects and emphasise that "a policy that only promotes intensification of cattle production is unlikely to reduce deforestation rates in Central America because incentives for forest clearing transcend the market demands for livestock products" (ibidem: 723).

5. Are PES Mechanisms up to the Conservation Job?

The previous section dealt with the main empirical results of the RISEMP. It indicated that the study region has experienced significant land use changes, mainly towards higher productive silvopastoral practices. While the quantitative data indicated that PES might have spurred investments in intensification, mainly through relaxing the capital constraints of the poorer farmers, further inquiry suggested that the effect of PES must be understood and interpreted in a broader context of change. In this section we now further reflect on the PES approach, in view of its effectiveness and long-term effects in promoting more environmentally-sound agricultural practices.

5.1. Payments, Markets and the Need for Quantification

Paying for environmental services implicates two minimum requirements. First, environmental services need to be quantified in measurable units, so people know what they are actually paying for or being paid for. Second, the measurable units need a (market) negotiated price, which normally should be based on both the willingness to pay (WTP) and the willingness to accept (WTA) of the ES buyers and ES providers respectively. While it is beyond the scope of this paper to discuss the controversial issues around environmental valuation techniques (see for example Foster, 1997 for a broad overview of these controversies), we do want to emphasise some of the main shortcomings related to the quantification of both ES units and ES prices, especially in the context of the RISEMP.

As was already explained in section 2 and 3, quantification of ES is usually based on the use of ES proxies, such as the land use likely to provide the targeted ES. While the site-specific and often uncertain links between land uses and their effects on ES provision usually turn the land use index much more into an oeuvre of generalised speculation than of scientific accuracy[1], the RISEMP tried to avoid this problem by dedicating a considerable part of project resources to the monitoring of land use effects on effective carbon sequestration and biodiversity indicators (see section 4.5). To some extent, the on-site studies confirmed the positive relationships of the promoted practices and the targeted ES, which also led to modifications of the ESI by the CATIE staff. Although this exercise might have been useful to get some more site-specific knowledge on the influence of certain types of vegetation on biodiversity or carbon, it is impossible to derive general rules on the effects of these land uses on the ES (Swift et al., 2004). The complex relationships between ecosystem functions[2], their context dependency (Kremen, 2005), and the still limited knowledge on their mutual interactions (Swift et al., 2004) make an approach based on attributing scores and corresponding payments to discrete or 'segregable' land uses, separable from their context, a quite arbitrary exercise. It reduces an ecological landscape to easy-manageable numbers (Robertson, 2004), which subsequently can be 'optimised' in an 'objective' way. Needless to say that in reality the biodiversity value of for example 100 successive hectares of forest is of a much higher quality than 100 fragmented terrestrial 'islands' of forests (MacArthur and Wilson, 1967). This site-specificity and practical immeasurability of

[1] Rojas and Aylward (2003), for example, criticise that most payments in existing PES programs are not based on scientifically proven relationships, but rather arbitrary on local perceptions.

[2] Ecosystem functions could be defined as the underlying capacity of natural processes and components to provide goods and services that satisfy human needs (De Groot, 1992).

land-use tied ES undermine the conceptual underpinnings of the use of ESI scores and their corresponding payments[1].

The intangible character of most ES also complicates their valuation. In the context of existing PES schemes, this problem is generally solved by implicitly or explicitly basing the monetary value on the farmers' opportunity costs of the main alternative land uses (Pagiola et al., 2007). The use of opportunity costs, however, does not always imply their full remuneration, accentuated by the 'tipping the balance' argument of PES adherents, as explained in section 3.3. This rather arbitrary juggling with opportunity cost and 'tipping the balance' concepts is at least theoretically confusing, but more importantly undermines the use of the term payments for environmental services; the ES as such are not actually paid for, but payments are rather directed to cover part of the opportunity costs of economically more, but often ecologically less attractive practices.

This conceptual confusion only grows if we further analyse what was actually paid for in the RISEMP, especially if we include a time dimension in ES provision. Suppose a farmer changes 1 hectare of degraded pasture into 1 hectare of scrub habitats (tacotales) during the first year of the project and then leaves this new land use unchanged during the remaining project duration. Making abstraction of the biodiversity payments, the farmer will receive the following yearly carbon payments (Table 3):

Table 3: Hypothetical example of yearly carbon payments for changing 1 ha of degraded pasture to 1 ha of scrub habitats (tacotales)

Year	Land use	Carbon score	Carbon payment (US$)
2003	DP	0	0
2004	Tac	0,8	60*
2005	Tac	0,8	60*
2006	Tac	0,8	60*
2007	Tac	0,8	60*

** Calculated as: carbon payment = [increase carbon points/ha as compared to 2003 x amount of ha] x US$75/carbon point*

Legend: DP = degraded pasture; Tac = tacotales (scrub habitats)

The payments illustrate that the farmer will receive yearly payments of US$60 once he or she has changed the land use to scrub habitats. Since payments for carbon sequestration were equivalent to about US$7,5 per ton of carbon sequestration (Pagiola et al., 2007), this payment suggests that the conversion of the degraded pasture into scrub habitats sequesters an average of about 8 ton of carbon per year, at least during the four project years[2]. Following the RISEMP 4-year scheme, the payment scheme thus compensates farmers for a total of 32 tons of carbon sequestrated, valued at US$240. The yearly payment seems logical as there is

[1] Of course, also PES-adherents are aware of this problem and considerable efforts have been made to adapt ESI scores to site-specificity. It can be doubted however to what extent this can solve the problem without increasing transaction costs to prohibitive levels.

[2] This seems to correspond to the results found in other studies, such as CATIE (1999, as cited by Pagiola et al., 2007) which found that silvopastoral practices can accumulate up to 13-15 tons of carbon ha-1 year-1.

an annual flow of sequestrated carbon, stocked away in growing above-ground vegetation and structural root biomass. There are however two caveats. The first is that the cutting point of four years, after which no further payments are made, does not seem to have a sound basis in technical analysis of carbon sequestration processes. Although these are still poorly understood, as they depend on specific (ecological) context as well as vegetative species and their management regimes (Albrecht and Kandji, 2003), there are indications that the net carbon fixation process continues well beyond these first four years (see for example de Jong et al., 1995, which base their calculations on rotation cycles of more than 25 years). Because of the limited bureaucratic lifespan of the RISEMP-project there might thus be undervaluation of the actual long term carbon sequestration service produced.

The second, more important caveat is that this kind of temporary carbon sequestration requires the carbon not to be released afterwards, e.g. by burning down trees or scrubs, or by gradual degradation of the new land uses through a lack of maintenance. So although annualized payments make sense, as gradually more net carbon is stocked away in vegetation and soil, these payments more fundamentally refer to a carbon sequestration service that stretches well beyond the year in which it is initially sequestrated. In other words, when a farmer receives his or her US$60 for 8 ton carbon sequestrated during one year (or the 240 US$ for the four years), he or she receives this as a once-and-for-all compensation for capturing this carbon and leaving it there for a long time. This is recognised by Pagiola et al. (2007) as they compare the once-and-for-all RISEMP-payment of US$240 with equivalent long-term annual payments (at a 10% discount rate), indicating that this would equal a yearly payment for the carbon sequestration service of US$24[1]. Obviously, this raises the question of the long-term sustainability of the investment in land use changes which are paid for. If this sustainability is guaranteed by the long-term economic rationality of the investments, as seems to be the case for the more productive silvopastoral investments (such as fodder banks or improved pastures with trees) at current prices and opportunity costs, there is not necessarily a problem and the once-and-for-all payments might indeed play a role in 'tipping the balance' in favor of such investments. Should these economic conditions change or in the case of other more temporary ecological (and less productive) investments (such as the scrub habitat in our example), this sustainability is far from guaranteed and a danger exists that farmers are paid a once-and-for-all price for a long-term service (demand) which is however only met with a short-term provisioning of it (supply). In such cases, we would be faced with only an apparent intersection of supply and demand in the market for environmental services.

So can we reasonably expect that US$240 for 32 ton of carbon sequestration through the economically unattractive use of scrub habitats can be materialised in effective and sustainable market transactions, especially if the time horizon is stretched to 20 or 50 years? In other words, would farmers still offer these services if they would receive yearly payments of US$12 or US$4,8 per hectare, spread over respectively 20 or 50 years? Or conversely -assuming an infinite and constant net accumulation capacity of ecosystems- would service buyers be willing and able to pay US$60 a year for a one-time change in land use, which supposedly constantly accumulates extra carbon, without any long-term certainty of permanent sequestration?

[1] White et al. (2006: 167) indicate that the insecurity about the future provision of the service gives rise to a significant negative price discount for carbon sequestrated by organic sinks, mentioning a tentative price of only US$4/tC.

A key concern that emerges from these tentative calculations and the analysis of the inconsistencies of the logic behind the ES payments in RISEMP, when viewed from the perspective of possible ES markets, is therefore whether the condition of an intersecting demand and supply for ES (see section 2.2) can always be met, and thus whether the perspective to establish ES markets with a real impact on farm decision-making can materialise, without the need for additional non-market governance. Especially within the current attempts of creating highly-localised markets for ES (see section 5.6), the former calculations suggest that opportunity costs and the corresponding WTA of farmers could be well above the actual WTP of local service users, considerably undermining the long-term sustainability of autonomous PES schemes in the assurance of the permanence of the ES, requiring infinite and high payments to be made by poor local dwellers (for services they hitherto have always received for free), as well as possibly prohibitively high monitoring and transaction costs (see section 5.5).

In conclusion, the question which was raised in section 4.4 on what an increase of 50% in the ESI score actually means, and -at least from the economically preferred efficiency point of view- maybe more importantly, what the corresponding payments have actually 'bought', seems rather redundant. The increase in the ESI score at most indicates that farmers have adopted some of the project-promoted land uses, but due to the non-contextualised attribution of scores and corresponding payments, and the opacity of what was actually paid for in a broader time dimension, it does not permit to claim something meaningful about the actual provision of the ES. It would be an illusion to think that the project bought about 50% extra biodiversity or carbon. So, is the use of the term of payments for environmental services not an abusive assertion for an approach that in reality does not buy any services as such? The use of a fuzzy mix of opportunity and investment cost considerations further indicates that payments are rather targeted to the use of supply-side determined environmentally-friendly agricultural practices (Smith, 2006) and are deceitfully justified by the use of a popular ES discourse based on a conceptually unsatisfying foundation, which creates the illusion that through the use of markets society can effectively buy ES.

5.2. Conservation and the Need for Collective Action

As discussed in section 2, the PES approach to landscape restoration suggests that the inherent collective action problem of environmental conservation can be solved by mainly focussing on the individual interests of farmers. As such, markets and payments should lead to a social optimum, in which conservation and agricultural activities can co-exist on a negotiated basis. Negotiation through the market implies that given the market price of ES, farmers have the choice to individually decide the amount of services they will provide to society. This decision will be largely based on the economic calculus of the farmers, which will take into account the opportunity costs of different land uses (and corresponding ES), and protect the low-opportunity cost parts of their land (which would lead to maximal cost-efficiency of the scarce conservation funds).

One of the main conditions for the protection and provision of ES is a landscape approach to conservation (Goldman et al., 2007; MEA, 2005). The Matiguás-Río Blanco project site was chosen because of its potential function as a corridor between two important natural reserves (Pagiola et al., 2004). At first sight the project could thus be considered to take a landscape approach to conservation. However, land use decisions were entirely left to the decisions of individual separate households, which obviously has some important implications for conservation. The low opportunity cost pieces of land which from the farmer's economic point of view are most interesting to conserve, do not automatically coincide with ecological priority areas. The creation of markets and corresponding payments only focus on landscape composition, and neglect the importance of landscape configuration. Especially in heavily cleared landscapes, such as Matiguás-Río Blanco, the promotion of corridors and other connecting habitat routes is indispensable for improving habitat quality (Haila, 2002). This requires coordination beyond farm level, implying mechanisms beyond markets in which farmers always have the possibility to "hold out for prices well above market rates" (Salzman, 2005: 939).

While coordination is less important for carbon sequestration, which is a non site-specific activity (i.e. it does not matter where carbon is captured)[1], it is crucial for biodiversity conservation, which requires a minimal amount of contiguous habitat, in which spatial patterns are one of the main characteristics (Gottfried et al., 1996). It is these 'economies of configuration' (Wear, 1992, as cited in Gottfried et al., 1996) which undermine the effectiveness of markets for solving these collective externality problems. Gottfried et al. (1996) summarised the general failure of the market approach in its attempt to ensure ES provision as:

> "The location of each landowner's parcel of land plays a critical role in determining the landscape's mix of goods and services. Aggregating landowners' contributions to ecological goods or bads, in order to determine some optimal landscape configuration via a damage or benefit function, loses this critical information. Because landowners jointly affect the landscape's ecological processes, scale problems emerge when taxes, subsidies, or other economic policy instruments attempt to internalize individual landowners' externalities. Instead, owners must be dealt with as a group, for it is at this scale that landscape level processes emerge. Because of these scale problems, individual owners acting alone cannot provide the socially optimal mix of ecologically-provided goods and services. Rather, this requires orchestrating human endeavors across a landscape and across landowner boundaries." (ibidem: 136).

In sum, exclusive reliance on the market approach, which leaves land use decisions entirely up to individual landowners, without a mechanism that simultaneously stimulates across-farm decision-making and cooperation, fails to offer a sustainable solution to environmental conservation. Some market proponents have increasingly recognised this problem of spatial configuration and coordination and have tried to deal with it by the implementation of so-called *agglomeration* or *coordination bonuses* (see for example Goldman et al., 2007 or Parkhurst and Shogren, 2007), which encourage connectivity by extra remunerating land use changes which border other 'conserved' areas. Nonetheless, empirical evidence on these systems is still very limited and falls short on the practical applicability of this approach on-field. In the case of the RISEMP, the bonus system was considered too complex (Pagiola et al., 2004).

[1] Goldman et al. (2007), however, claim that over the long term configuration of trees for carbon sequestration do play an important role, especially in the tropics where "trees in small fragments have experienced high mortality from wind exposure, microclimatic stresses (changes in moisture, temperature, or light), and proliferating lianas" (ibidem: 337).

5.3. Externalities, Entitlements and Their Policy Implications

At the heart of neoclassical environmental economics is the concept of externalities, which was partly discussed in section 2. Externalities, which refer to costs or benefits imposed on people outside an economic transaction, also lie at the core of the PES approach. In general terms, PES adherents often believe that markets in which positive externalities (termed as ES) can be traded will prove beneficial to the environment, and solve the funding problem by attaching monetary values to ES for which beneficiaries have to pay. In this way, PES mechanisms no longer conceive farmers as causes of environmental disservices, but rather as potential ES providers. As such, this approach breaks away from the Polluter Pays Principle (PPP) and holds to a mix of the Provider Gets (PGP) and the Beneficiary Pays Principle (BPP) (Hanley et al., 1998). While these principles should not be addressed as scientifically sound theoretical concepts, but rather as practical rules of thumb, it is remarkable how rarely this change in focus from negative to positive externalities has been explicitly covered and justified in the PES literature[1], which has some important implications for the sustainability of the approach.

The widespread use of the externality terminology implicitly encloses the environmental problem in a (neoclassical) economics framework, which also suggests the possible solutions to the problem (Vatn, 2005). Following this framework, it is the inexistence of markets for externalities which cause increasing environmental degradation. While this narrow vision of environmental problems has been criticised in the literature (see for example Paavola and Adger, 2005 who advocate for a broader view on environmental problems as interdependencies[2] instead of externalities), it can be argued that it is still the mainstream way of thinking about environmental problems among many economists. But even within the externality framework, there is still room for contradicting visions on (solutions to) environmental degradation.

One of the main inherent contradictions -at least in the context of PES- is how to assess an externality as being positive or negative. In the context of the RISEMP it was argued that farmers who change their land use practices towards more environment-friendly ways should be (at least partly) compensated for the positive externalities they provide to society. But what if we reverse this way of perceiving reality and instead claim that farmers have a moral or social duty to preserve the environment such that when they deviate from this obligation, they should be considered as causing negative externalities to society, engendering the need for the farmers to compensate society for this harm. The latter would imply that paying for ES in reality means that society is 'bribing' farmers to secure a supply of environmental goods (Hanley et al., 1998), and in the context of Nicaraguan regulations, for rights they arguably never had. Moreover, from the efficiency point of view, it is often argued that priority payments should go to the most degraded farms, which would convert the PGP in a de facto Pay the Polluter Principle (Hanley et al., 1998), which could be normatively disturbing (Salzman, 2005).

[1] One of the main exceptions is Salzman (2005), who devotes a large part of his article to the discussion and justification of the focus on positive versus negative externalities.

[2] This concept of interdependency, which surges from the New Institutional Economics paradigm, is explained by Paavola and Adger (2005: 355) as: "Interdependence exists when a choice of one agent influences that of another –a situation overlooked in conventional economic analysis which assumes that agents are independent." They add that "Economists have failed to recognise 'externalities' as instances of interdependence despite the obviousness of this in the classic externality examples… [which show that] one agent's choice limits the range of choices available to other ones or influences the choices made by them" (ibidem: 355).

So, which claim should be supported? Should we punish polluters or pay providers? The academic literature shows that both claims are to some extent defendable, and that the categorisation of externalities -which also determines the applied policy- is fundamentally based on the historic and socio-institutional evolution of entitlements, in particular property rights over land[1]. Ellickson (1973), for example, proposed the use of the 'normalcy' concept, which categorises externalities by referring to a socially accepted normal or zero reference state and its corresponding environmental entitlements, which allow the categorisation of deviations from this reference state as being harmful (negative) or beneficial (positive). Young et al. (2003) further elaborate on this concept by using the 'duty of care' approach, in which landowners have the obligation to "take all reasonable and practical steps to prevent harm arising from their activities" (ibidem: 4, emphasis in original). These concepts allow for potential 'grey' zones in between the two extreme principles; landowners should pay society if management falls below the socially desired level and should be compensated if their management produces benefits above the minimum duty of care (Bromley and Hodge, 1990, as cited by Young et al., 2003). Wittman (1984), however, recognises that externality problems are inherently symmetric, but shows that the existence of administrative costs tied to governance structures changes this symmetry and therefore calls for the use of the 'negative' approach:

> "...charging one person, X, for being inefficient involves much lower administrative costs than compensating X for being efficient. With either method, X would typically act efficiently. Payment of compensation, however, entails high administrative costs: it would be called for in numerous instances, because the efficient outcome would be typical. In contrast, there would be low administrative costs if X were charged only for acting inefficiently, a type of behaviour rare in comparison to acting efficiently" (Wittman, 1984: 61).

This result implies that the ever-used economic criterion of cost efficiency -on which a large part of the PES approach is based (see section 2)- cannot automatically justify the use of market creation in the presence of high transaction costs (Vatn, 2005); it rather suggests that the compensation of every landowner for the produced positive externalities would provoke prohibitive transaction costs, an issue we will further discuss in section 5.4.

Recognising that the efficiency criterion is just one among many other criteria, we could expand the externality discussion to yet another level. Using the fairness criterion, for example, addresses some other important questions, especially in the context of localised user-financed PES programs; is it fair to expect 'environmental service users' to pay for services they used to get for free, especially if they do not contribute (neither in a direct way, nor in an indirect way) to environmental degradation? Is clean water and air, for example, not the normal reference situation and even a basic human right? In order to clarify these important remarks, it could be useful to turn to a highly simplified but very illustrative example. Suppose farmer A has 2 ha of land, subdivided into 1 ha of forest and 1 ha of pasture on which she keeps 1 cow. Further downstream live two urban dwellers (B and C) which for the provision of their clean water are dependent on good upstream land stewardship. Deforestation of the 1 ha would give A an extra income of 1 cow. Introduction of the BPP, which assesses the 1 ha of forest as a land use creating positive externalities, would require both urban dwellers to pay the equivalent opportunity cost

[1] Although this does not necessarily have to be the case, the PES approach often seems to implicitly assume an unrestricted private property right over land and associated natural resources (see also section 5.4) without any social responsibility as care-taker of these resources.

of 1 cow to A in order to 'save' the forest and continue clean water provision. The distribution of the costs could be established according to the amount of water each urban dweller A and B uses (this methodology is in fact often used in localised hydrological PES schemes). Suppose B consumes 100 litres of water and C 200 litres. The BPP mechanism would lead to the solution that B pays A the equivalent of 0,33 cows and C pays A the equivalent of 0,66 cows. But what if B is a carnivore and C is a vegetarian? Is it fair to expect a higher contribution of C, just because he consumes more water? Is it not fairer to tie the contributions of B and C to the potential negative externality caused by raising an extra cow -by charging a fee on the producer (which can shift this on to consumer B)-, which is a more direct underlying driver of deforestation, and as such make B -who indirectly steers the upland land use- contribute more to the proposed compensation payments?

The example shows that the BPP (which underlies the PES approach) does not seem to 'take the bull by the horns', but rather looks for solutions on a level which is hardly linked to the actual causes of the environmental problem. In other words, the PES approach looks for demand side funds by using an impact logic (who benefits more from clean water?), but neglects that fairness would imply dealing with the funding problem at the level of the driving forces (who is directly or indirectly responsible for the contamination of water?). In this context the PPP seems to be a fairer conservation tool, at least, if it is tied to the commodity that (indirectly) causes the negative externality. The latter principle would allow raising environmental funds, independently of which services this piece of forest is offering to society, and partly avoid the free-riding problem, which in fact is artificially created by focusing on impacts instead of on drivers of environmental degradation.

The implementation of PES mechanisms has important implications on the 'de facto' legitimacy of certain actions. In theory, the use of PES to supplement weakly enforced laws and "provide a carrot that makes the stick of regulations more palatable" (Engel et al., 2008: 669) seems quite attractive. However, creating a new institution by paying for certain land uses, changes the locus of responsibility and ultimately leaves the choice up to the landowners who will base their environmental contribution efforts on the condition of receiving sufficient payments. The landowner's decision of cutting down trees, for example, which according to Nicaraguan law is an illegal act, can then justify his or her act by the foregone monetary payments of this land use, which de facto legitimises illegal land use practices (Ravnborg et al., 2007; Young et al., 2003), and gives the farmer a de facto entitlement to pollute. As such, "there is a danger that existing institutional actors may take less responsibility for biodiversity if they see a new institution created for this purpose" (Wells, 1998: 830). Furthermore, PES approaches often implicitly allow farmers to demand compensations for actions they do not undertake (and were perhaps not even thinking of undertaking), and thereby implicitly presumes an unrestricted private property right over land and resources of the individual owners. This claim is however not self-evident as property rights will typically be restricted by a number of state and non-state rules of entitlements.

The former discussion shows that determining the appropriate characterisation of an externality is not an easy and objective assignment. Salzman (2005) concludes that the categorisation of externalities "turns less on biophysical measures or ecological modelling than on our sense of what the allocation and definition of entitlements ought to look like and how they should change over time. These questions, in the end, are value judgements" (ibidem: 960, emphasis added; see also Paavola, 2007). The imposition of certain criteria over others is thus utterly based on value judgements, which are often deceitfully presented as scientifically-proven 'truths'. As such, it would be wrong to blindly accept the positive externality approach or the efficiency criterion as scientifically 'correct', and consequently treat the environmental problem as a pure technical optimisation within this subjectively chosen framework. Paying farmers for the provision of ES is not a pure technical matter, but could implicitly affirm or redefine their entitlements, with the potential danger of undermining social norms and environmental ethics, transforming these in marketable environmental entitlements, an issue we will turn to in the next section. One should therefore be aware that the application of certain frameworks to specific problems is utterly based on the political power to impose certain values, which in the age of neoliberal globalisation are mainly determined by the global elites (McAfee, 1999).

5.4. PES, Institutions and Human Behaviour

> "The environment is not a commodity like produced goods and services; as the context within which all life occurs, it is part of what constitutes the common human good, and as such (as well as for its intrinsic value) it is the subject of ethical consideration" (Jacobs, 1997: 229).

Market-approaches to nature conservation build on the rational actor paradigm, which assumes that people act out of self-interest, and will not undertake conservation actions if the latter do not contribute to the actor's private utility. In other words, market proponents claim that nature should be protected by the use of market institutions which capitalise on human self-interest (Karp and Gaulding, 1995). The introduction of markets alters human behaviour mainly by using extrinsic motivation, which changes relative prices and make some actions more beneficial than others (Frey and Stutzer, 2006). In the RISEMP, this principle was applied by attributing differing payments to each of the promoted land use practices, altering the relative price structure of certain land uses and practices as compared to the situation before project implementation.

The sheer reliance on this relative price effect for environmental conservation neglects, however, that institutions are not simply objective incentive transmitters, but also exercise effects on people's intrinsic motivations, which may relate to a sense of enjoyment, satisfaction, (social) responsibility or obligation (Reeson, 2008). As Reeson further explains, "the way in which a situation is perceived can determine the extent to which intrinsic motivations are applied" (ibidem: 18). Especially in cases where certain conservation practices are already exercised voluntarily or under social pressure, mainly because of existing intrinsic motivation, the introduction of price mechanisms can partially destroy the relative price effect or in some cases even reverse it, resulting in a reduced supply of environmental goods and services (Frey and Oberholzer-Gee, 1997). This effect of extrinsic motivations destroying intrinsic motivations and environmental morale, in the literature known as 'motivation crowding-out' (Frey, 1997), reflects the 'hidden costs of rewards' (Lepper and Greene, 1978), and adds some important remarks to the potential sustainability of PES approaches. If the relative price effect of markets is 'exceeded' by the opposite motivational crowding-out effect, then the creation of PES mechanisms might create more harm than good to nature. But even in the case where relative price effects are higher than the possible negative crowding-out effects, the sustainability of the PES approach is still very fragile; because if payments become the primary cause for behavioural change, the discontinuation of this incentive is likely to become a cause for reversal (Enters et al., 2004), with -due to the path dependency of policy measures- possibly lower levels of conservation than before.

In line with these concerns, Heyman and Ariely (2004) show that monetary payments invoke monetary marketplace frameworks, which could irreversibly erode 'social markets' in which efforts are mainly expended on the basis of social relationships. In practice this means that efforts, which were previously framed in a social market and largely independent of compensation levels, are subsequently framed in monetary markets which are "characterized by a monotonic [positive] relationship between payment and effort" (ibidem: 792). This implies that "not paying at all in the context of social market relationships can create higher levels of incentives than low levels of compensations in the context of money-market relationships" (ibidem: 788), and warns for an increasing need of conservation funds to achieve certain conservation goals[1]. In the context of PES, Reeson (2008: 20) further warns for the potential implication that market-based payments might mean "that landholders will come to expect to be paid for actions they are currently doing voluntarily". The previous considerations imply that once payments are introduced, farmers can be irreversibly 'drawn' into a path dependent market logic (Salzman, 2005), which implies that in order to have sustainable conservation the payments need to be on-going, rather than short term (Pagiola and Platais, 2007). Following the logic of motivation crowding-out, once the payments are ended, efforts to conserve nature could even drop to below pre-project levels.

In the case of the RISEMP there are some anecdotal evidences of former project farmers who after project termination cut part of the trees or burned down scrub habitats they had been planting or protecting during the project. But, as was already extensively explained in section 4, most of the promoted practices during the RISEMP were widely perceived to be in the farmers' private interest, which substantially lowers the risk of destroying what was conserved or added during the project. Another danger which was also observed through some anecdotal evidence, is what Young et al. (2003) have called the use of the 'environment as a ransom', by which former project farmers, but also former non-participants in the region threaten to cut down currently protected forests on their land if they do not receive compensations for (further) protecting them. Especially in the neighbouring Quirragua natural reserve farmers are increasingly using this threat. As such, the introduction of the discourse of a market logic could imply a dangerous precedent which could lead to 'token economies' (Frey and Stutzer, 2006) in which people would expect to be paid for actions they previously did out of moral obligation or social pressure. This is exactly what Heyman and Ariely (2004) warn for when they assert that "the social aspects of reward are fragile and a social reward can easily be made into a non-social extrinsic reward by merely mentioning monetary circumstances or perhaps just promoting comparisons to other tasks or other individuals' reward levels" (ibidem: 793).

A key question is however to what extent there are actually pre-existing intrinsic motivations and social norms that contribute to environmentally-sound practices. If these are few, there is not much to destroy in terms of 'social markets' and 'intrinsic motivation'. In the

[1] This claim was researched by Gneezy and Rustichini (2000) in an experimental context. The authors concluded that "the usual prediction of higher performance with higher compensation, when one is offered, has been confirmed: but the performance may be lower because of the introduction of the compensation" (ibidem: 807, emphasis in original). Wunder (2005) recognised this potential danger when he states that "At worst, conservation effort in exchange for a low monetary PES could be lower than for 'no payment'. This is noteworthy, since in most cases PES amounts paid have actually remained low." (ibidem: 15).

Nicaraguan agricultural frontier, a strong individualistic peasant-work ethic prevails, collective action and mutual control are often weak, and with respect to the environment the historically dominant logic was to consider trees as a hindrance for production and cleared parcels as 'mejoras' (improvements) (Bastiaensen et al., 2006: 15-16). In a context of gradually changing local perceptions, one could wonder whether there could not also exist a positive interaction between extrinsic and intrinsic motivation, whereby the strong PES signal that outsiders value and are willing to pay for environmental protection could lead to changes in local perceptions, values and norms concerning 'accepted' and 'desired' agricultural practices. At the level of the individual such positive 'motivational crowding-in' rather than 'crowding-out' does not seem a priori impossible, but it remains unlikely that individualized PES-payments alone could contribute to the strengthening and emergence of 'social markets', even when they could play a positive role in a broader, combined approach.

While it would be rather naïve to presuppose that cooperation problems could be solved by relying exclusively on environmental morale and voluntary action, it is important to acknowledge that people are predisposed to follow social norms, and that institutions can "both shape preferences and determine which social norm or behavioural paradigm is followed" (Reeson and Tisdell, 2006: 21). In particular, it could be very important to work on changing the informal, non-state perceptions about land property rights, which in the agricultural frontier were typically associated with cutting down trees and exploiting the land (Bastiaensen et al., 2006), rather than with responsible exploitation and conservation of natural resources. The use of an experimental design set-up, which only allows to focus on the behavioural effects of certain predefined incentives, only marginally touches upon the importance of the existing socio-institutional landscape in Matiguás-Río Blanco. The effect of other variables on pro-environmental behaviour, such as social capital and existing formal and informal institutions (e.g. local rules and social norms), are unduly neglected in such an approach.

Our analysis of land use changes among different communities, however, showed some interesting results. Farmers in the community of San Ignacio, for example, which upon analysis proved to be the best organised project community (only community with a community council -including a committee for environmental themes- and a water committee, as well as a well-functioning cooperative milk collection centre), were the ones who most increased forests as well as improved pastures with trees[1], and according to former project staff made the best quality changes in their farm. Farmers in the communities of Patastule or El Gavilán, where any significant form of organisation is absent, on the other hand, have made the least changes in their farm and generally their changes were of poor quality. These findings suggest that social capital and collectively shared (environmental) ideas might have played an important role in the type of changes that were made in the farms, which reinforces our previously mentioned pathway argument.

The preceding discussion emphasises the need to understand the whole socio-institutional landscape before simply 'pushing' the provision of ES in a market framework. It emphasises that "people apply different behaviors in different institutional settings" (Vatn, 2005: 215). The often unconditional reliance on markets as 'objective' tools to secure environmental conser-

[1] In San Ignacio forests covered an extra 4,1% of total land, while all other communities knew a change of maximum 0,7%. Improved pastures with trees increased from 11% to 33,8% of total land use, which means an additional cover of 22,9%; the closest follower knew an increase of 14,4%.

vation through exploitation of the self-interested nature of human beings, actually neglects the fact that the basic premise of self-interest is often exactly a consequence of placing people in a market framework, which may "trigger people to behave in a self-interested way, rather than in the more cooperative or reciprocal ways in which they behave in other situations" (Reeson and Tisdell, 2006: 20). In the context of a need for collective action among all land users, it would be a categorical mistake to always invoke the market perspective (Vatn, 2005). In this context it is important to approach human beings not simply as selfish utility maximisers, but take due account of the existence of environmental morale and how the latter can be further developed and utilised through deliberative institutions that enable cooperative actions and foster what Agrawal (2005) has called 'the making of environmental subjects'[1]. Vatn (2005: 215) perfectly recapitulates the previous discussion by emphasising that "Choosing policy instruments is thus not simply about changing incentives. First of all it is about instituting certain logics, about understanding which institutional frames people apply, and about influencing these frames."

5.5. Transaction Costs and Efficiency

PES schemes are often criticised for their potential high transaction costs, especially when working with many small-scale ES providers (Ravnborg et al., 2007; van Noordwijk et al., 2004; Young et al., 2003). In the case of the RISEMP transaction costs were indeed very high. Total payments in the three participating RISEMP countries were estimated to cover about 15% of total project costs[2], while the remaining 85% went to other activities, such as detailed monitoring and research, capacity building, and project management (World Bank, 2002). This low share of effective payments in total project costs can be explained to a large extent by the pilot nature of the project. According to Pagiola et al. (2007) the high transaction costs, mainly tied to the detailed monitoring and some other activities which would not be necessary in a scaled-up project, could be reduced by simplifying the ESI index and using proxy indicators "that are highly correlated with biodiversity conservation but are easy and cheap to monitor" (ibidem: 382). Nevertheless, as discussed in section 5.1, the complex relationships between land uses and ES and between different ES mutually and the high site-specificity of these relations, require extensive and continuous contextual research with inevitable corresponding high transaction costs, especially in the context of market creation, where real (and not only perceived) relationships have to be very clear in order to guarantee sustainability of the mechanism. The latter was also emphasised by Kroeger and Casey (2007: 328), who state that "While some of the gaps could be closed with current knowledge, effective implementation of high-quality markets that capture the full economic value of traded services would most likely be very expensive and complex, and perhaps prohibitively so, because of the attendant measurement, valuation, and monitoring requirements".

An important question is thus if PES -which claims to be more cost-efficient than other approaches through the use of direct incentives (Ferraro and Simpson, 2002)- really offers higher conservation value for every invested dollar. High transaction costs potentially conflict with efficiency criteria. But even if we would make abstraction of high transaction costs, and

[1] It is somewhat striking that most former RISEMP participants in Matiguás-Río Blanco did not understand the logic of the ESI point system and the underlying ecological logic of the project; during our interviews we found that a significant part of the interviewed farmers perceived the project as a mere support in productive agricultural practices and did not make an implicit link to the environmental advantages of the promoted activities. It is thus very questionable whether the project has contributed to the fostering of an environmental morale as such.

[2] Total project payments to farmers in Matiguás-Río Blanco amounted to US$ 244.245

only focus on the cash payments, the approach is still likely to provide low value for the invested conservation money. Our analysis in section 4 showed that -at least in the case of the RISEMP in Matiguás-Río Blanco- the additional value of payments has been rather marginal, especially if compared to exogenous processes in the region, which seem to have favoured the promoted practices anyway. The analysis also stressed the importance of higher valued services such as technical assistance and other forms of cross-farm education in the adoption of more environmentally friendly, but mainly higher productive practices. It follows rather logically then that the scarce conservation funds which were used to pay farmers could have improved the environment even more and in a more sustainable way by investing it in more technical assistance or in institutions that promote cross-farm learning or help farmers through the extension of market access, just to give some examples. As was discussed in section 5.2, efforts that stimulate collective action among farmers could prove much more sustainable, while market approaches which promote costly land use changes, but do not explicitly focus on the coordination requirement, could be a waste of money. Equally important could be investments in the creation and social maintenance of sufficiently shared, environmentally-conditioned perceptions and rules of the game, which can become 'actionable knowledge' to support such required collective action (see section 5.4).

Furthermore, in section 3.1 we already mentioned that the main barriers to the adoption of silvopastoral practices are the relative high investment costs and the long time lag until financial profitability. But if practices are privately profitable after a certain time, and if capital constraints are the main barriers to investments, wouldn't it have been a more cost-efficient option to work with credit instruments? The local rural microfinance institution FDL[1], for example, offers cheap 'green credits' targeted to environmentally-friendly agricultural investments. The project could certainly have improved its impact by helping poorer and capital-constrained farmers to secure access to these (subsidised) credits, instead of only paying them ex post for privately profitable investments. Additionally, this would have lowered the risk of motivational crowding-out, leading to higher overall efficiency and sustainability.

The effectiveness of payments was also questioned in other studies. Kosoy et al. (2007), for example, emphasised that it was mainly landholders already committed to forest conservation who were participating in three Central American PES schemes, which according to these authors "brings about some doubts about the cost-effectiveness of the payments vis-à-vis alternative tools for upstream-downstream concerted action" (ibidem: 452). Muñoz-Piña et al. (2008) found similar results in Mexico, where payments for hydrological services were mainly benefiting landholders in areas with low deforestation risk. Strict reliance on the PGP means that scarce funds could be wasted to farmers who would have conserved certain land uses anyway, which of course increases inefficiency. While PES adherents have increasingly recognised this weak spot and tried to solve it by elaborating sophisticated targeting methods (see for example Wünscher et al., 2008), we believe the main reasons behind the low additional value of payments are rather related to the motivational and behavioural underpinnings of the concept (see previous section), which undermines the long-run efficiency of the approach. Indeed, claiming that 'direct' PES approaches are more efficient than indirect approaches to conservation manifests a rather static notion of both conservation and efficiency characteristics. In the short run, direct incentives will probably display more readily observable results than indirect approaches

[1] FDL stands for 'Fondo de Desarrollo Local' or 'Fund for Local Development'.

(Giger, 1999), but in the longer run this apparent higher efficiency could be only an illusion, as it does not take into account possible spill-over or crowding effects, which were already discussed in the previous section.

5.6. Long-term Funding and the Burden of Conservation

Section 5.4 already explained that by pushing conservation into a market context, PES programs generally need ongoing, rather than finite payments. The RISEMP, however, could be considered as an exception to this rule. According to Pagiola et al. (2007) the long-term private profitability of the silvopastoral practices could justify short term payments which would lift farmers to a sustainable track. In section 5.5 we already suggested that credits might have been a more efficient instrument to promote silvopastoral practices. Most PES schemes, however, deal with unprofitable land use conservation instead of productive land use changes. In addition, Pagiola et al. (2007: 382) state that "it is highly unlikely that the profitability of silvopastoral practices in a country as large and varied as Nicaragua, let alone regionwide, is always such that a short-term payment would 'tip the balance' in their favour. [...] Even in the case of farmers for whom short-term payments are sufficient to induce long-term adoption of silvopastoral practices, longer-term payments may still be desirable because of the conditionality they allow on other land use decisions, such as preventing burning fields or cutting trees in other parts of the farm". As such, the logic goes, the short-term character of most existing PES schemes[1] proves unsustainable in the long run. Additional long-term funds should be found.

The PES philosophy suggests that payments should be secured through negotiations with service users. Those who benefit from ES should bear the cost. In the case of the RISEMP most funding was secured by 'global society', represented by the multilateral GEF institution. Since they mainly produce global benefits, it seems fair that biodiversity and carbon sequestration are financed through global funds. But there is the rub. Since global funding mechanisms, such as the Clean Development Mechanism in the case of carbon sequestration, often prove difficult in reality, mainly because of limited funding, high transaction costs and strict rules and conditionality tied to the funding, expectations for 'fund raising' in the context of PES are increasingly focussed on local communities. Wunder et al. (2008), for example, concluded in a broad overview of current PES initiatives that there is an increasing attempt to change government-financed PES programs to more efficient user-financed programs, which "are attempting to develop additional financing sources from individual ES users to complement their public financing" (ibidem: 851). In the context of the RISEMP, for example, Pagiola et al. (2007) suggested that potential long-term funding could be secured through the establishment of local markets for water services[2], which "offer the most promising avenue for financing long-term PES programs" (ibidem: 383). These attempts mean in practice that (potential) local ecosystem benefits are identified, the local willingness to pay for them is assessed, and a new mechanism for funding them is established.

[1] Pagiola et al. (2007) give some examples of short-term PES programs in Costa Rica and Ecuador, where it was hoped that conservation would still be sustainable after ending project payments, mainly by raising the profitability of tree plantations.

[2] Because of the site-specificity of water services, and the fact that neither of both targeted watersheds in the study region contributes to the water supply in Matiguás and Río Blanco, this possibility proved to be unfeasible (Pagiola et al., 2007).

Nevertheless, this increasing search for and faith in local funds raises some critical concerns. Expecting that local people will pay for locally generated ES makes a dangerous abstraction of the 'joint production' nature of different ecosystem benefits. As was already discussed in section 5.1, the complex relationships within and between different ecosystems make it impossible to meaningfully segregate or 'unbundle' different ES, and sell them each as separate services on different markets. A problem indeed seems to be that a PES scheme can hardly ever take into account and compensate for all the ES provided. It could well be true that a forest is responsible for local people's water provision, but simultaneously this same forest also offers global biodiversity and carbon sequestration benefits. If sheer reliance on local demand of ES is then applied to guarantee ecosystem conservation, then it could be argued that the extraction of funds out of the local community only widens the existing regressive financing of local conservation efforts that produce global benefits (Balmford and Whitten, 2003; Barrett et al., 2001). In a sense it provides a mechanism that discards global responsibility and makes local people pay for their own foregone economic development (Karsenty, 2004). As was already discussed in section 5.3, this logic emanates from applying the BPP, which focuses on the impact instead of the drivers of environmental degradation.

Can we morally expect that local people in developing countries, which often already need to struggle for mere survival, should bear the burden of conservation, just because there happens to exist a local demand for some specific ES? Isn't it hypocritical to arbitrarily pick out some of the locally created externalities and expect local people to pay for the opportunity costs of conserving a certain ecosystem, merely based on the argument that the same local people are the principal beneficiaries of one or a few ES among many other simultaneously generated services? Is the creation of locally user-financed PES systems not a perverse example of global free-riding and an opportunistically applied additionality principle at the cost of poor communities? Is it fair to expect that local ES users pay for externalities while Western beef consumers are at least co-responsible for the existence of these externalities in the first place? These questions suggest that the PES concept still has to deal with some important politico-ethical issues. They also display that local conservation is probably not so much a question of scarce money per se, but rather of having the global political will to change.

6. Conclusion

We started this article by recognising that the PES approach provided us with an innovative, appealing and attractive narrative about the causes and possible market solutions to the problem of the underprovisioning of ES related to agricultural activities. Yet, despite claims of PES advocates that their approach is theoretically well-founded as well as applicable in practice, our contribution has indicated that a too narrow market-based approach might not adequately explain the dynamics that are taking place, nor do they automatically contribute to effective and sustainable improvements in ES provision. While the RISEMP results confirm the positive role of monetary incentives in changing farmers' land use, we argued that the underlying motivations for these dynamics were also linked to other exogenous processes in the study region. PES payments interacted with exogenous economic factors and broader local institutional processes, which together generated renewed and environmentally-sound collective pathways towards intensified silvopastoral milk production. The PES payments were an additional objective and symbolic factor in this broader dynamic.

We further indicated that the PES market approach is difficult to be linked with the required 'economics of configuration', necessary to create connected biological corridors or to protect sites critical for water management, and it does not provide incentives in the direction of the collective action that this requires. Rather we found that the spread of the market-based logic of monetary rewards discouraged such collective action and might in particular cases even tend to erode existing environmental ethics and social norms, unless PES is matched to effective local institutions.

As such, PES adherents need to (better) understand that producing socio-perceptional change is not simply a matter of playing with relative prices and tipping financial balances. Due account needs to be taken that creating new 'failure-absorbing' markets to supposedly 'correct' failures of other markets, implicitly approves the structures that allowed for the emergence of these failures in the first place. As such, extensive reliance on market tools to solve environmental problems implicitly depoliticises environmentalism and, as McAfee (1999: 151) has persuasively claimed, "it offers a rationale for the illusion that biological diversity can be 'saved' without fundamental changes in present distributions of political power". An explicit focus on PES implies a potential danger of deviating our attention away from the real drivers behind environmental degradation, which usually require broader political actions and not merely market creation.

So is there any role to play for PES? Is it able to solve (at least some of the) environmental problems in the developing world? Maybe, but –as is increasingly recognised in the PES literature- certainly not on its own. Complex problems require well thought-out solutions, and generally invoke a mix of policy instruments. We think, however, that PES has a much more marginal role to play in this 'policy package' than adherents of the approach would like us to believe. The PES concept has undoubtedly contributed to the growing recognition that environmental problems cannot be solved by simply applying 'no-touch' conservation policy instruments, but that sustainable environmental protection in an agricultural context requires a negotiation-based compromise between farmers and the rest of society. As such, PES advocates have emphasised the positive role that farmers could play in providing crucial ecosystem services to society, and have rightly stressed that farmers will need support in doing so.

Nevertheless, our arguments in this paper lead us to the conclusion that despite the apparent attractiveness of the PES story, its implicit promotion of the market as the 'Magic Bullet', which can and should replace 'ineffective local or state governance' could eventually do more harm than good[1]. A more sophisticated and ideologically flexible approach that recognises the advantages and disadvantages of 'market', 'state' and 'community' governance as well as their varied and complex local manifestations in each particular circumstance and that modestly tries to generate ways to build better institutional mechanisms which improve overall outcomes might thus be more appropriate. As an expression of joint (world) responsibility for the limited natural resources of our planet, principles and mechanisms of PES and ES markets can still be part of improved institutional governance, but a narrow individualistic market-based application should not be treated 'ex ante' as the superior governance alternative. Other, more collective institutional mechanisms that allow to transfer resources to ES providers might have to be considered.

[1]

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Annex I: Methodological Explanations

Our field study was conducted with the objective of getting more insight in the participants' experiences with this project. We wanted to get to know the underlying motivations for (not) changing land use practices and the possible influence of extrinsic incentives on intrinsic motivation, and assess whether there were potentially other factors which could have played a role in motivating farmers to adopt new land use practices. In order to obtain as much diversity as possible in characteristics thought to be of interest for this study, we selected the participants through maximum variability sampling (Glaser & Strauss, 1967), in which all farmers were ranked on the basis of high, low, and median values for certain variables in the beforementioned data sets, such as participant group (PES, PES + TA, Control) and scheme (2 versus 4 years); received payments; farm size; location and accessibility (communities, and sectors within communities); gender; herd size; and type of land use changes. On the basis of this exercise we obtained a list of about 90 participants, among whom we randomly chose a first group of farmers to be interviewed. Since we were not aspiring quantitative representativity, but rather theoretical saturation (Glaser & Strauss, 1967), we conducted the selection of respondents with an open attitude (e.g. based on recommendations from former interviewed participants of other successful or unsuccessful cases), without limiting us to the preliminary selection.

The interviews were conducted in the absence of former project staff and in the name of the University of Antwerp in order to avoid being associated with the project and therefore eliciting socially desired responses. Basically, the first part of every interview was devoted to getting to know the life history and livelihood strategies of every farmer (on average 1 hour), after which the interview was gradually directed to the topic of projects and institutions with which the respondent had been collaborating, eventually talking about the RISEMP (on average 1 hour).

The interviews with the Nitlapán staff were conducted with the main objective of getting to know the (more technical) on-field experiences of the project staff and the main project results according to them, and to cross-check some of the main findings of the farmer interviews. As such, by bringing up some experiences or anecdotes from the earlier interviewed farmers, we tried to form a more nuanced context for some of the narratives provided by the farmers.

In conclusion, many of the arguments and conclusions in this paper rely heavily on triangulation between interviews. This means that they are supported by interviews with different people who independently have arrived at the same impression about a certain aspect of the project. Where possible, claims are further supported by (quantitative) project data and documents.

One referee of this paper made the observation that PES-theoricists fully agree with this conclusion and never claim that PES are such a 'Magical Bullet', but that they are often forced to present and 'sell' the PES-story to funders in a simplified version. The key question here is however which of both, the simple story or the more sophisticated and nuanced theoretical reflections, has more influence when translating PES-ideas to policy and interventions in the field.

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