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Market and Non-market Instruments

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PAYMENTS FOR NATURE VALUES
MARKET AND NON-MARKET INSTRUMENTS

by

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Foreword

A central element in the international negotiations on resource mobilization under the United Nations Convention on Biological Diversity (CBD) is to broaden the basis for financing the implementation of the convention. The issue of payments for nature values as resource mobilization is at the center of the controversy in these negotiations. The first Quito dialogue on biodiversity finance in 2012 clarified that international trading in biodiversity is more controversial than local and national markets in biodiversity values. Even though the Quito dialogue was not a platform for creating consensus, it was a common view, that a further study of the "financialization of nature" as a basis for making decisions on the use of market-based instruments for international financing of biodiversity was needed. The statement was reiterated at the second Quito dialogue in April 2014.

The different views on payments for nature values vary according to the different world-views, weight given to monetary and non-monetary valuation of nature and the markets, economic tools and regulations and trading approaches proposed. The proponents see win-win possibilities by engaging business and the finance sector in environmental protection by creating efficient markets for trading in environmental benefits. The opponents see this as opening up for privatization of the common benefits of nature, licensing destruction of nature and leaving the last perils of our survival to the whims of the business and finance sector.

Norad commissioned this study with the aim of getting updated information on and an analysis of the challenges and opportunities for the implementation of different regimes for payment of ecosystem services (PES). A group of researchers led by professor Arild Vatn at NMBU in Norway won the contract for the study. The preliminary results were presented at the second Quito dialogue in April 2014 and three experts were invited by Norad to peer review the draft report. The final report covers the issues as outlined by Norad in the Terms of Reference for the study.

Norway has an active role in the negotiations on resource mobilization in the CBD and the United Nations Framework Convention on Climate Change (UNFCC). Different market based regimes are being negotiated in the CBD and the UNFCCC, but the regimes will both influence each other in practice and have some major common challenges. The practical experiences of international trade in carbon under the UNFCCC may contribute valuable lessons for the CBD discussion on biodiversity finance. On the other hand, lessons drawn from biodiversity finance will also be of relevance for area-based trade in carbon.

The market based regimes that have financed biodiversity so far are all site specific; linked to the management of a piece of land or water. In that respect they are similar to the regimes discussed under Land Use, Land-Use Change and Forestry (LULUCF), Reduced Emissions from Deforestation and forest Degradation(REDD) and Clean Development Mechanism for Afforestation and Reforestation (CDM-AR) in the UNFCCC. Carbon quotas based on land management will have a direct positive or negative effect on the biodiversity of the land. Such area-based quotas are not as liquid as the industry-based quotas. The likelihood that verified industry based quotas has to be withdrawn is regarded as smaller and as a result, get a better price.

Access and Benefit Sharing of the utilization of genetic resources (ABS) is the only market based biodiversity PES regulated by internationally agreed terms. The Nagoya Protocol took 9 years to negotiate and will most probably come into force in 2014. As it is not yet implemented, it is not included in this study. REDD and Biodiversity Offsets are the two most prominent biodiversity finance mechanisms. We had expected the study to unravel lessons learned on these topics. However, the literature search revealed that these instruments are still in the making, with rather limited documented experiences.

The report presents a classification of economic instruments including markets, an analysis of the concept of financialization and potential implication for markets for ES, the socio-economic impacts of different economic instruments and a series of unintended effects of these instruments.

The analyses and conclusions in this study are the independent work by the group of scientists, led by professor Arild Vatn, and do not represent an official Norwegian position on biodiversity and climate finance.

Oslo, 5th June 2014

Bente Herstad
Norad

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On behalf of the authors,

Aas/NMBU, 05.06.14

Arild Vatn

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Content list

Acronyms and Abbreviations

BDO – Biodiversity offsetting
BW – Bretton Woods
CBA – Cost-benefit analysis
CBD – The Convention on Biological Diversity
CDM – The Clean Development Mechanism
CDM_{AR} – Afforestation and reforestation projects under CDM
CER – Certified emission reductions
ES – Ecosystem Services
EUA – European Union Allowance
EU-ETS – European Union – Emission Trading System
OTC – Over-the-counter
PES – Payments for ecosystem services
PSA (Costa Rica) - Pago por Servicios Ambientales (Payments for Environmental Services)
PSAH (Mexico) - National Programme for Hydrological Environmental Services
tCO_2 – tons of carbon dioxide
TCs – Transaction costs
TDR – Tradable development rights
USD – United States dollar
VAT – Value-added tax
WTP – Willingness to pay

Summary

Ensuring conservation and sustainable use of biodiversity is challenging. After the 10th Conference of the Parties to the Convention on Biodiversity in Nagoya in 2010 there has been continued focus at how to increase funding for implementing the CBD through both market and non-market sources. In a previous report,[i] we offered insights into the role markets could play in expanding finances for biodiversity. We concluded that to ensure ample funding, public regulation has to play a key role. This is also the case to for expanding private funding. This is because biodiversity and the services it delivers are mainly public goods and services.

The present report includes further analyses of the above issues with emphasis on economic instruments more generally – both market and non-market. First, we offer a classification of economic instruments including markets. Second, we analyze the concept of financialization and potential implication for markets for ES. Third, we look at socio-economic impacts of different economic instruments. Finally, we study a series of unintended effects of these instruments.

Economic instruments can be divided in pure public instruments – like taxes and subsidies – and markets. The former implies payments between private actors and the state and are based on state power to command. The latter is characterized by parties trading over goods or services. We have trade where private actors pay e.g., a landowner to change her/his land use to reduce degradation of biodiversity. The responsibility to stop degradation is then taken on voluntarily – i.e., we can talk of a non-liability based market. These markets are small. The large markets are based on publicly defined caps that are tradable – hence, they are liability-based. The protection of the ES lies in the defined cap. The market is a way to reduce the costs of abiding by the cap. Regarding resources for ES, they therefore come mostly from the public purse. Such payments take dominantly the form of subsidies (no trade) or public auctions (trade) are used. Hence, public bodies are the main 'buyers' of ES as long as they are of the public kind – e.g., biodiversity.

Forest bonds are among the new financial products proposed to enhance private funding of biodiversity/ES. They provide resources for investments in e.g., enhancement of forests. Private as well as public bodies can issue them. As debt they demand, however, interest and a full payback upon maturity. Hence, financial flows from the asset – the forest – is needed for such an instrument to generate any sizeable private resources. Logging, carbon storage and ecotourism are key examples of ways to ensure that this happens, but especially in the case of logging, there is a conflict between the need to generate revenues and the capacity to protect biodiversity. Hence, it would seem that to attract private funding, states have to be involved to cover the gap between private claims on profits and what cash flows a forest aimed at biodiversity protection can deliver. We find it then more reasonable to use these resources to expand public funds and programs.

This conclusion is supported by observations regarding the dynamics of markets and the role of financialization. The latter implies turning tradable commodities into financial objects that can themselves be traded. This happens typically through securitization and trading in derivatives. These products exist not least to hedge against risks in commodity markets and markets for debt like bonds. It does not itself create any resources for e.g., ES production or protection. At the same time, the risks themselves, as well as opportunities for arbitrage between different capital markets, makes gains from speculation possible. Expanding profits through arbitrage demands increased borrowing, which creates new risks that are spread by bundling different debts and then trading these. This seems rational in a market context, but such securitization has together with trade in derivatives been shown to create a disconnect with the underlying asset, as well as increasing systemic risks that could create crises, respectively endanger the real values or assets (i.e. natural

habitats). There is also a tendency for intermediaries – by utilizing their strategic position – to capture a large part of potential gains from this trading. Finally, financialization is in itself costly to undertake. Hence, we advise care when turning towards markets for bonds to expand financing biodiversity protection, as it seems to 'demand' financialization. The costs that follow from such a development should not be underestimated. As a basis for that conclusion, we also note that by turning to the market, there is a fundamental change in the process of nature protection where calculation of risk and profit opportunities replaces political judgment.

Economic instruments comprise both market and non-market types, with payments as a common element – e.g., payments for ecosystem services (PES). In the third part of the report, we look at distributional effects of such payments – emphasizing experiences with PES and the part of the clean development mechanism directed at afforestation and reforestation (CDM_{AR}). While the former mainly takes the format of public subsidies, the latter comprises trade. We find that payments mainly go to larger land-owners – an effect of a) transaction costs (TCs) being high for low volume trades, and b) that small-holders often need the land for sustaining themselves. Payments are typically found to be lower than opportunity costs. Hence, PES and CDM_{AR} seem not to have realized much 'win-win' – reducing both deforestation and poverty. Focusing more at the community level and including intermediaries that are motivated towards supporting community development, seems to be effective means to include the poor, while safeguards are important to avoid elite capture. This strategy will also help reduce per unit TCs as the volume per contract affects these costs so heavily. We observe that in many situations public bodies have the capacity to reduce TCs substantially compared to private actors, as they can use command power.

In the last part of the report, we discuss potential sources of unintended effects on biodiversity conservation of economic instruments through ecosystem function and biodiversity interlinkages, motivational crowding, social networks, slippage, and policy interactions. Unintended effects at property and landscape level mean that there may be fewer 'win-win' conservation opportunities than some market-based instrument literature would suggest. Land-owners self-select conservation 'efforts' on cheap land – this is adverse for conservation where natural habitat biodiversity and agricultural land use capacity are positively correlated. 'Scarcity slippage' in the form of increased local agricultural prices and incentives for forest conversion due to scarcity of agricultural land is more likely to occur in countries where rigidities in credit, labor and land markets are more pronounced. Voluntary conservation motivations may be crowded out by economic incentives for conservation, but may not be recovered once incentives stop. In contexts with strong community organization, social network effects are likely, with copying behavior substituting net benefit rationales where land use decisions are complex and uncertain. Other instruments in a policy mix may interact unexpectedly with economic instruments. Almost by definition, empirical evidence for these unintended effects is scant - effects foreseen are more likely to be monitored. We present a number of cases where unintended effects have been observed, but they are often local and context specific.

The overall message from our analyses is the importance of public engagement. Private resources are very important, but will only be engaged in rather small volumes if not directed by action of states and municipalities. Biodiversity is moreover a 'local resource' that demands local adaptation. Therefore, markets – when being a reasonable solution – must be locally delimited.

1. Introduction

Since the conference of the parties to the convention of biodiversity (CBD) in Nagoya in 2010[2] there have been substantial discussions about the use of economic instruments to facilitate extended conservation and sustainable use of biodiversity. These debates have considered different ways to enhance the financial basis for CBD related actions. Involving the private sector and more use of markets have been emphasized. Discussions have centred on how effective and efficient such a strategy could be regarding meeting the goals of the CBD. There has also been emphasis on distributional effects of such a strategy.

In a previous report from 2011[3] we offered an overview of different economic instruments – including markets – used for protection of biodiversity. We analysed their legitimacy – both regarding process and outcomes. We noticed that while payments for ecosystem services were important, the resources came largely from public sources. We also noted that where markets were involved, public bodies – typically states – play a key role both in defining liabilities necessary to create an interest in trading and in controlling that the rules set are followed. We observed that in these cases, the protection of nature values lies predominantly in the definition of liabilities – e.g., caps. Trading is a way to reduce compliance costs among those liable. We noted that the strong role of states/public agents was caused by the public goods[i] characteristics of biodiversity and its attached ecosystem services. The embedded free rider problem seems to prevent any sizeable payments from private actors to public goods following from conservation. The observation relates also to the often very high transaction costs involved when using markets in areas like biodiversity. Hence, public payments were found to be the most efficient in several areas. Lack of empirical data regarding effects of different systems was noted as a limitation to parts of the analysis.

This report can be seen as a follow up of the 2011 publication. It analyses a set of issues regarding market/non-market instruments that have surfaced in the CBD community over the last couple of years – not least as part of the so-called Quito dialogues. First, we respond to a need for developing a consistent classification scheme for economic instruments involved in conservation and sustainable use of biodiversity/ecosystem services. How to draw the line between what is a market and what is not, is a core topic here. Similarly, there is a need to establish a common language regarding different types of markets to facilitate clarity in the discussions.

Regarding expanded use of market instruments in the field of biodiversity, there has been substantial emphasis on the issue of financialization. The second part of the report looks at this topic, explaining what financialization is, and what it might imply regarding environmental effects, distribution of revenues and transaction costs. Since the development of financial products of this kind seems to be on the 'drawing table' in the area of biodiversity, this chapter takes largely the form of a principal analysis with some references to the experience with carbon markets/CDM.

Next we focus on evaluating socio-economic impacts of different economic instruments, following up on the discussion on distributional impacts from the first report. Here we emphasize distribution of economic gains and losses and how to explain the variations observed in level of transaction costs across instruments.

The issue of unintended impacts on biodiversity from various payments systems has also been emphasized in the debate. The last part of the report discusses such impacts at two levels a) unexpected behavioural responses at farm level to different economic instruments, and b) side-

[i] In this report we use the concept of public goods as short for both public goods and common-pool resources. See Appendix 1 for definitions and explanations.

effects on biodiversity at the landscape level of payments to support other ecosystem services like those from water, carbon and landscape amenities.

While the main focus is on biodiversity, the report expands beyond this field. Experiences with markets in ecosystem services is at present more extensive outside the field of biodiversity. Hence, to evaluate potential effects for an expansion within the area of biodiversity, utilizing knowledge from other fields is important. Note also that we in the following will use two concepts regarding the environmental aspects we look at. The wider concept – that of 'nature values' – covers the totality of anthropocentric and 'intrinsic' values of nature. Ecosystem services is a sub-concept only emphasizing the anthropocentric aspects. According to MEA (2005) these services can be grouped in four – i.e., i) provisioning services (like food, fresh water, wood and fibers, fuel); ii) regulating services (like climate regulation, flood regulation, disease regulation, water purification); iii) supporting services (like nutrient cycling, soil formation, primary production); iv) cultural aspects (like aesthetic, spiritual, educational, recreational). While many of the provisioning services can be categorized as private services, the others are typically of the public kind.

The report is largely based on a review of documented research in the different fields assessed. A set of appendices are attached to the report to support the main analysis. A core element here is an overview of concepts that may not be very familiar to all readers. All concepts that are underlined the first time they appear in the following text are further explained in Appendix 1.

2. Classifying economic instruments

We have developed two classification schemes for this report. First, we specify the main types of policy instruments. While largely being non-market, some markets are included to the extent states/ public bodies participate in or are key in establishing these. The second classification regards markets for ecosystem services (ES). The chapter concludes with a brief overview of existing core instruments for ES payments/trading.

Cooking with the guests - Co-management and participatory policy design with landusers
Illustration Javier Sáez
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2.1 A conceptual basis for the classification

We have chosen to use the concept of a governance structure[4] as basis for our classification. It consists of two elements:

- The types of actors involved with their motivations, rights/responsibilities – e.g., private vs. public actors
- The type of interaction between the involved actors – e.g., trade vs. command vs. cooperation/reciprocity

The above elements facilitate capturing key aspects regarding how the instruments function. First of all, the involved motivations depend on who the actors are, what liabilities they have and what kind of interaction is facilitated. A firm is primarily oriented at maximizing profits. When operating in markets, it is expected to look for gains from trade. A public body is instituted to serve the public interest. It may commands its citizens. It may, however, also engage in trades. Finally, civil society organizations operate on the basis of a societal cause – often a social or environmental one.

Regarding motivation, the theory of rational choice – i.e., the concept of the economically rational agent who maximizes utility or profits – is a core reference. As already indicated, there may, however, be many different motivations driving choices. Regarding land use, individual/ household decisions are typically very important. The issue of profits may be central to such actors. However, as we will see later on, actions may be based also on other types of motivations.

Nature values/ES are dominantly public goods or services. This has some important implications because actions undertaken to ensure protection or sustainable use will create a benefit not only for the ones that e.g., protect, but also all others that gain from such acts. Hence, the economically rational actor may not want to voluntarily engage in protection – while societally favorable – since costs are typically high relative to the gains for the individual her-/himself.

The type of interaction between actors – its power basis and its costs – is also important to understand when evaluating policies. While trades are between formally equal parties, command is based on legitimated hierarchical power. Reciprocity is also between equal parties, but the logic is different from trades as it is based on community trust. It is also notable that the interaction structure influences how easy it is to interact and control what is delivered etc. A key concept here is transaction costs. While these costs depend on the kind of goods/services involved, they also depend very much on the type of interactions instituted.

There is a 'command' element to all instruments – be they market or non-market – as rights must be defined. Even 'voluntary' action is made with reference to a specified property right for the resource in mind. The following classifications avoid some inconsistencies implied by distinctions like market vs. state and market-based instruments vs. command-and-control.

2.2 A general classification scheme for policy instruments

A classification of the main types of policy instruments – distinguishing between legal, informational and economic – are found in Table 1. Examples are also offered. The categorization is quite similar to the one found in OECD (2013). The concept of a policy instrument relates to public action – dominantly state action. Some markets are still included, like markets where states trade with private actors – e.g., public auctions – or where the state establishes the market as in the case of cap-and-trade systems.

The categories of Table 1 are generic. In practice, we observe combinations. Legal instruments are typically underpinning any policy – e.g., defining rights and responsibilities; what the public actor is allowed to do. Payments in the form of compensations for costs incurred are often linked to legal regulations like area protection. Similarly, information packages typically follow the introduction of both legal and economic policy instruments.

We note that the 'power base' for the instruments is quite different. Legal instruments are based on the state power to command – i.e., to rule out or put high costs on certain acts (e.g., fines or imprisonment). Informational instruments are directed at changing behavior through altering actors' knowledge about their options. It is directed at voluntary actions. Both legal and informational instruments may, however, also change people's values and preferences – i.e., how they view the choice set. This is the aim of normatively directed information campaigns. Such changes may also follow from new laws – as is shown in the case of e.g., smoking regulations[5].

Table 1. Classification scheme for policy instruments in the field of the environment

Legal rules		**Information**	**Economic instruments**	

Public provisioning: e.g., rules regarding resource use/protection on public land	Legal protection
	- Prohibitions
	- Mandated solutions
	- Protection
	  - National parks
	  - Nature reserves
		Information
		- Technical
		- Normative Education/ development of skills
			Pure public instruments
			- Taxes and fees
			- Subsidies
			- Fiscal transfers
				Markets:
				- Contract based payments
				- Public auctions
				- Cap-and-trade systems

Economic instruments are divided in two – pure public instruments and markets where public agents/states are a party to trades. The former is based on the command power of the state, although being offered a payment in the form of a subsidy may not seem to represent much forcing. Markets are based on voluntary transactions. The basis for the trade may, however, vary. When states trade with private actors – either through e.g., individual contracts or through auctions – the private party is free to engage. In the case of cap-and-trade systems, the state defines a liability – like a cap on an emission or on land 'development'. The state then facilitates trading of the liability. Trade will be of interest to private actors if it reduces the costs of abiding by the cap.

In our classification, we have deliberately avoided the use of the concept of market-based instruments. We think that it is better to talk about market and non-market instruments. Hence, we use the concept of economic instruments to cover all instruments where monetary transfers/incentives are involved, whether they happen through markets (trade) or are the result of public command (taxes, subsidies etc.). The concept of a market is this way exclusively reserved for trading.

2.3 A classification scheme for markets in ecosystem services

We define a market as a constellation of actors involved in trades over specific goods or services. Some define markets as a place where goods are exchanged at fixed prices[6] – i.e., markets are competitive. We will, however, define markets by the format of the interaction only – whether it is a trade or not – emphasizing the specific motivations involved through the characteristics of the interaction. We moreover note that the competitive market is a theoretical concept – assuming standardized commodities and zero transaction costs. Using this as a basis for defining markets in ES actually circumvents most challenges that markets may face in this field.

Markets are a subcategory of economic instruments. Not all of them are policy instruments, as trades between private actors dominate. Hence, the category of markets in Table 1 is only covering a sub-set of markets that are relevant to this report. At the same time, markets are based on states defining who holds various rights. Often public regulations go beyond this, defining quality standards as a basis for trading, having responsibility for controls, etc.

A market in ES[ii],demands at minimum one seller and one buyer. A trade directly between the two can be termed a 'direct market' – see Table 2. More typically, we observe that there is an intermediary operating between sellers and buyers. Hence, two or more transactions will be involved, and if both are trades, we term the market 'complete'. If only one is trade and the other is not – e.g., based on command like raising the money through taxes – we term the market 'incomplete'. There is an important 'grey zone' to be mentioned here – i.e., the case of brokers as opposed to traders. Brokers may, in the same way as traders, buy and sell. They may, however, also operate only as facilitators of negotiations. Strictly spoken they are then not intermediaries in the sense of Table 2.

The second dimension in Table 2 regards liability. In some cases no liabilities are defined. Actors are free to act, despite the fact that it may bring costs upon others through loss of nature values/ES. Nevertheless, victims of e.g., pollution (or other actors) may decide to pay polluters to change their behavior. While trade is per definition always voluntary[iii], private actors take in this case on the responsibility also for ensuring environmental protection. If the one paying is a public body/a state, it is politically decided that 'polluters has the right'/the 'providers' gets principle' rules. The opposite rights situation implies that the ones damaging nature values/ES are liable.

[ii] Note here that we refer only to the anthropocentric aspect of nature as trade in other nature values seems a contradiction.

[iii] Certainly, the basis from which a party trades may be weak. Options are fewer for the poor (Martinez-Alier 2002)

This may involve the introduction of an environmental tax. It may, however also take the form of a cap on e.g., pollution or land conversion. If that cap is tradable, a market is established.

Table 2. Classification scheme for markets in environmental services with examples

	Direct market (between seller and buyer)	**Market with intermediaries**	
		Complete (all transactions – both the one between sellers and the intermediary and the buyer and the intermediary – are trade based)	**Incomplete** (combination of trade-based and non-trade based transactions)
Non-liability based	Vitel case (PES)	Some market **PES** systems: water, biodiversity, carbon **Certification** schemes	Most market **PES** systems: water, biodiversity, carbon
Liability based	EU ETS – bilateral trades	**EU ETS** **CDM** (private buyers) Biodiversity **offsets as banking**	Some **CDM** projects (public buyers)

In Table 2 we have tried to categorize some existing markets for ES according to the classification scheme. Payments for ecosystem services (PES) is as a case where the buyer 'voluntary' pay a seller – with or without an intermediary. However, few of these trades seem to be direct. A very notable case is that of Vitel/Nestlé contracting with farmers in an area in Northern France for increased water quality[7]. More important are transactions with intermediaries. Based on data from Milder et al.[8] we conclude that of all land-related PES for public goods in 2009 – i.e., water, landscape beauty, biodiversity and carbon – about 99 % of the resources involved – in total about 23 billion USD – are raised by public bodies through taxes, fees etc. It has not been possible to establish how large a fraction of the public money has been used for trading with sellers – e.g., through auctions. We conclude, however, that a majority of public payments are subsidies. Hence, the fraction that can be termed trade is clearly below 50 %. From this we conclude that most of PES is not a market according to the above classification. Moreover, just 1 % seems to qualify as a direct or complete. Here the so-called 'voluntary carbon market' dominates – i.e., land-related carbon trades amounting to about 158 million USD in 2009. This figure should be compared to the total carbon market, which was about 144 billion USD in that year[9]. Hence, trade is mainly the result of publicly defined caps, allowing the implied obligations to be traded.

Another important non-liability based system is certification. A wide number of such programs exist – not least in fields like agriculture and forestry. The total value of agricultural products included in a certified program is in the order of 64 billion USD – about 2.5 % of the total market for food.[10] We have found no overall information neither about the overall extra margin nor about what fraction would accrue to an environmental/public goods component. Note that individual health considerations are important here. In forestry the volume of certified products is estimated to be in the order of 54 billion USD[11]. We classify these markets as dominantly complete.

Moving next to the liability based systems, cap-and-trade based carbon markets and biodiversity offsets are the most important. According to the World Bank[12] the value of carbon trades under a liability scheme was in 2011 around 175.5 billion USD. Of this, the EU ETS accounted for about 85 % (148 billion USD). The next largest volume was the secondary CDM market – 22 billion USD – while the primary CDM market is much smaller, 3 billion USD. While some of the EU ETS trades are direct, most are of the complete ones with intermediary. In the case of CDM, we find both complete and incomplete markets with intermediaries. The former dominates, as public agents are only involved in a minor part of trades.

Regarding biodiversity offsets, one may distinguish between systems with a liability to compensate for biodiversity loss, and systems where this liability may be traded.[13] Habitat banking system are the core example of the latter type. It is mainly used in the US. In other systems, the degree of trading varies. One example of a liability system without trading is the German Impact Mitigation Regulation System. Madsen et al.[14] indicate that biodiversity offsets are in a yearly order of 1.8-2.9 billion USD. Programs in the US dominate. Australia has also put much effort into establishing biodiversity offsets. The instrument is spreading, and there are now offset programs in a large number of countries e.g., Brazil, Canada and several European countries.

3. Financialization of trades in ecosystem services – a principal analysis

In this chapter, we analyze present knowledge about financialization regarding conservation and sustainable use of biodiversity/ecosystem services. The analysis is mainly principal. In the case of biodiversity, financialization is just emerging. Hence, empirical experience comes foremost from financial markets in general, respectively from carbon markets. The issue is, however, important, since increased use of markets in biodiversity values will most probably result also in financialization. Whether that is good or bad is a heated topic.

Different styles of cooking – experimental versus predictive policy mix design
Illustration Javier Sáez
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3.1 What is financialization?

The definition of financialization in the field of ES varies across the literature. Some include monetization of nature values into the definition – i.e., financialization happens already at the stage of monetary assessments/pricing of such values. Others emphasize that as soon as there is trading in nature values/ES, there is financialization. While both these aspects are necessary for financialization to take place, we think it is wise to delimit the concept to the fact that there is trade in financial products themselves. Hence we define financialization as the turning of tradable commodities' values – in our case commodities in ES – into financial objects that can themselves be traded.[15] Core examples of financialization are securitization and derivatives.

Securitization implies pooling different debts into tradable products. They may take the form of bonds. It is seen as a way to e.g., increase liquidity, expand revenues on equity/own capital, reduce funding exposure and transfer risk. A derivative, on the other hand, is a financial instrument where investors 'bet' on the development of an asset/commodity, an index, an interest rate etc. – the so-called 'underlying'.[16] The basic logic of derivatives is more narrowly to hedge against risk in the development of prices and/or quantities of the 'underlying'. Hence, early forms are forwards and futures linked to trades in grain where sellers and buyers hedge against risks related to fluctuation in prices and/or uncertainties regarding the volumes to be delivered. They may take the form of forwards, futures, options or swaps.

Securitization and derivative creation are phenomena whose scale[iv] is substantially increased after the break down of the Bretton Woods institutions from 1971[v], resulting in a deregulation of

[iv] Such instruments are, however, observed used as far back as in the 17th century. They have been a contentious issue from the very beginning.

[v] The start was the devaluation of the USD by the Nixon administration. A key element of the Bretton Woods (BW) order was linking other currencies to the dollar. The BW system was established in 1944, based on experiences with

financial markets and followed by economic globalization. These changes resulted in increased risks in financial markets and is the basis for a tremendous increase in financialization. Another side of the same coin is the expanded opportunities for speculation.[17] Technical aspects seem also of some importance – e.g., progresses in probability theory and developments in information and computer technology.[18] This development has by many been seen as good – as utilizing the advantages of markets. The discussion over financialization in nature conservation reflects a further expansion of this thinking into new policy arenas. Those supporting this trend argue that it can help expand financial resources for this end. It is also argued that markets will increase efficiency in conservation and create ways where protection and use can be sustainably combined.

While the basic logic behind derivatives and to some extent also securitization seems to be hedging against risks, the establishment of such products lends themselves also to speculation. Through e.g., various forms of leverage, it is possible for financial agents to create opportunities for arbitrage. This may, however, create its own risks. It may increase volatility.[19] It may also result in what is termed systemic risks that under certain conditions may result in economic crises.[20] The recent financial crisis is a notable illustration. We will return to this.

3.2 Pricing, price incentives and trade

In the debate over markets or market-like solutions to ensure nature protection/sustainable use, the issue of commodification and pricing of nature values has been a key issue.[21] To understand what financialization means, we find it necessary to clarify different forms and functions of pricing: i) pricing through monetary valuation, ii) through payments, and iii) through trade.

Monetary valuation of nature values/ES is used as a way to price unpriced goods or services. While some ES are traded in markets – e.g., food, timber – most are not, as they are public goods. Hence, some argue that the reason why many ES are under pressure from e.g., economic development is that they lack a price.[22] To include these values in public and private decision-making, measuring their values in monetary terms is seen as crucial. The arguments are partly 'fundamental', partly 'pragmatic'. The 'fundamental' position holds that rational allocation of resources demands a comparison of costs and benefits, implying measurement by a common denominator - money. Moreover, the calculation should be based on individual preferences through people's choices or willingness to pay (WTP). Where markets and hence prices do not exist, one should estimate them indirectly through e.g., house prices, travel costs, or directly through simulated markets – e.g., WTP studies. The 'pragmatic' argument is based on the observation that 'money talks strong'. Monetary assessments are seen as added arguments to biological and ethical ones.

Monetary valuation is also criticized. The critique spans a wide variety of issues from questioning the utilitarian basis for monetary assessments/cost-benefit analysis, the implied emphasis on individual as opposed to social preferences, the problems related to collapsing a multi-dimensional set of values into one scale, and the loss of information involved in that process.[vi] [23]

The compensation or incentive aspect of pricing regards payments to owners/users of e.g., land that delivers ES. The idea is that land-owners/users – if paid to deliver e.g., protection of land to ensure species habitats – can shift to this 'production' instead of cutting trees and selling timber. PES is a key example of this kind of pricing. Incentive payments do not require monetary valuation

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the 1929 financial crisis. It established the International Monetary Fund and the International Bank for Reconstruction and Development – later part of the World Bank – with an aim to ensure financial stability and provide financial flows.

[vi] See Appendix 2 for a description of the various forms of monetary valuation, and the arguments pro et contra.

of the ES. One may decide that the amount of protected forests or forests under some form of sustainable use in an area needs to be enlarged by a certain percentage. Then e.g., the state, municipality or private actors may ensure that by defining a high enough payment – i.e., covering the opportunity costs for the land owners/users.[vii]

The third form of pricing follows from trade – i.e., the price is created through transactions in markets. This demands that the ES is transformed into a tradable commodity. This raise a series of issues regarding transforming public goods and services into private ones. The issues are both technical – e.g., is it possible delimit goods and services of this kind – and ethical – e.g., transforming nature into tradable commodities.[24]

Regarding pricing, the basis for real and simulated markets is principally the same – willingness (and ability) to pay. In practice there are, however, quite some difference. Pricing through markets captures more realistically individual income constraints and the effects on prices on other goods. On the other hand, decisions over the allocation of nature values are left to markets. In the case of monetary valuation only – if accepted as information to be included in public decision processes – the final choice is still based on political judgment.

Financialization demands payments – i.e., there must be pricing in the second or third form as explained above. The 'underlying' must be linked to some kind of cash flow upon which the securitization can take place/the derivative can be built.

3.3 Financialization and the creation of resources for ES

We have already mentioned the increased emphasis on expanding the financial resources for protection not least of biodiversity. In that respect, there has been a hope that markets could be created to deliver new resources. While great values are involved, this is, however, not a simple solution due to the public goods characteristics of biodiversity/many ES.

To start with the simplest case – derivatives – they do not offer any new resources to protection/sustainable use. They may help hedging against some of the (enlarged) environmental risk we are facing, though. Derivatives may also be used to hedge against risks involved when developing other kinds of financial products to support e.g., biodiversity protection. It may, however, also result in speculation that could have negative effects on the protection of biodiversity/ES delivery.

Biodiversity loss and climate change are processes that will increase environmental risks. Derivatives may be constructed to hedge against these. Notably, a weather derivatives market was created in 1997.[25] As Mandel et al. note, "natural disasters correlate poorly with the performance of stocks and traditional bonds".[26] Hence, these derivatives should be interesting for those wanting to diversify portfolios. Certainly, as long as such disasters are a rather marginal phenomenon, protection through financial means may be helpful. If environmental risks become 'systemic' – i.e. reach a level where they have global impact/influence the functioning of the economy – this is clearly less so. Hedging can to some extent handle variation, but not lasting negative trends. The market for derivatives in environmental risks could, however, be seen as an indicator of how bad the status of the environment is.

[vii] For a further discussion about motivations for protecting land and reactions to payments, see Chapter 5 and Appendices 3.4 and 3.5.

While derivatives do not create any new resources, instruments like green bonds, forest bonds and the like[27] could be used to expand the financial basis for biodiversity protection, activities to combat climate change etc. Bonds are debt and can be issued_by public or private agents. If one wants to restore or expand a forest, necessary finances for planting could be raised by issuing bonds. As the buyer is entitled an interest and the full value back upon maturity, the motivation or logic is as for any other debt. The investment must create the necessary return. This certainly limits its capacities as an investment in protection. There may be private benefits from this – e.g., one may sell timber and still deliver public goods like biodiversity protection. One may earn income through ecotourism. There may also be investors that accept reduced returns – while the experience with PES indicates that this is not a strong source (cf. Section 2.3). Nevertheless, the system seems to demand that the state commits itself to pay parts of the returns or set up regulations that ensure the necessary cash flows.[28] One may, however, wonder why the state could not create the resources directly through regulation instead of going the way via supporting a market for bonds and next having to ensure sufficient private profits. As argued by S. Sullivan, the motivation seems rather to come from a financial sector searching for new income opportunities through 'public-private partnerships'.[29] Finally, debt based financing in the form of bonds also raises the issue of ownership to the land if the issuers of bonds are not able to fulfill the obligations in the end.

Forest bonds as described above are not the result of financialization. Such bonds could easily end up as part of such products, though. There are two elements of importance here. That of leveraging and that of bundling. Leveraging is a way for investors to increase revenues on equity/own capital. Box 3.1 is a simple illustration of how one – through utilizing variations in returns on different assets/types of debt (opportunities of arbitrage) – can ensure increased gains for involved traders and equity/shareholders. While this may offer great gains, it may also result in great losses, if assumptions do not hold.

Box 1 Bonds, leveraging and expected returns

Investors typically utilizes the variation in interest rates and returns on various types of debt to maximize returns on equity/own capital. Let us assume that an investment is expected to offer a return of 6 % per year. Basing it on a bank loan (50 %) with an interest rate of 4 %, a bond (40 %) with an interest rate of 5 %, offers an expected return on equity/own investment (10 %) at 20 %. Certainly, if the overall return on the investment is only 4 %, (or lower) the agent faces a large loss. Hence, this kind of leveraging increases financial risks

Bundling of various debts has been seen as a way to reduce the risks created through leveraging. This strategy implies creating securitized products where packages of debts are resold – typically after being tranched. The latter implies that the portfolio of debts are divided up into packages with different levels of risks resold to investors with different risk attitudes. H.S. Shin argues that this increase aggregated risk rather than reducing it. Banks and other financial actors increase their level of debt – through leveraging – to enhance short-term profit. This way they buy each other's assets with borrowed money. Instead of dispersing risks, this concentrate all the risks in the financial system itself.[30] The main point for us here is that products like forests bonds – rather 'innocent' financial products – will like other debts typically be involved in leveraging and securitization. Hence, it should be emphasized that to the extent these instruments become important in biodiversity/ES protection, the success of the strategy becomes linked to the development of financial markets/the capacity to politically regulate hedging procedures.

The developments following the privatization of the English water sector illustrates aspects of this at the micro level. Here water companies use repackaging and leveraging to increase revenues in ways principally similar to the structure described in Box 3.1. Allen and Pryke show

how this creates a redistribution of gains towards shareholders and intermediaries.[31] Subsequently, they observe weak investments in the water services system – indicating that some ‘stripping’ of assets is going on to ensure short-term profits. Finally, authors point at changes in the political process around water services as a consequence of privatization and securitization. In that respect, two issues seem important. First, the financialization is outside the regulation and falls hence outside the ‘political spotlight’. Second, the kind of judgments made regarding the service – which are fundamentally social and political – are obscured by being forced under the logic of trade.[32]

The above example illustrates how financialization may have quite profound effects on the real economy, not only through the positive effects of hedging against (some) risks, but also through redistribution and possible creation of new (systemic) risks and increased volatility.[33] The activities of the financial sector itself needs to be compensated. Hence, financialization is creating its own (transaction) costs. Intermediaries are typically strategically positioned and able to utilize different information asymmetries at their advantage as also illustrated by the above water case. Finally, complexity of various securitized products and derivatives seems to obscure the relationships to the real value of the ‘underlying’ – as was so clearly illustrated by the recent financial crisis. In the end costs were, moreover, to a large extent shifted to taxpayers through various bailouts.

Financial products like bonds and securitized debts is a demand on the future. Using such instruments for environmental protection is, hence, characterized by an inherent dilemma. Economic growth makes the gains on these investments more secure. At the same time, such growth is itself a challenge to the environments we want to protect/use sustainably. This becomes especially notable as derivatives link various ‘underlying assets or values’ – typically across sectors. This way the protection of nature becomes linked to the general development of the economy.

It is finally important to note that using public financing – certainly ultimately based on resources from economic activity – avoids securitization and derivatives’ creation both because the state has the capacity to ‘hedge’ against risks itself and because there is no reason or opportunity for speculation. However, economic growth makes it easier for states to raise the necessary resources needed. That ‘paradox’ is similar to that of financialization. We see, however, little if any gain in going the way through e.g., bonds and derivatives if it is so that the state has still to ultimately guarantee for the financial flows. It needs anyway to create funds through e.g., taxes.

3.4 Experiences with derivatives in the carbon markets

Experiences from the carbon markets provide multiple examples relevant to financialization and trade in ES. We will here briefly explain a few of the dynamics of trade in derivatives and securitization in the EU Emission Trading System (ETS), with emphasis on CERs (Certified Emission Reductions), i.e., carbon credits from the CDM.

Unlike many other commodities, emission allowances, when issued, have no transportation, environmental, storage or insurance risks associated with their possession.[34] However, there is a clear distinction between CERs that are not issued (often referred to as primary), and those that are, as the former carry a high delivery risk. This risk depends on project type, geographical location, financial, environmental and operational aspects, regulatory procedures, market factors etc.

An analysis of expected CERs up to 2011 showed that 29 percent were never issued due to project failures,[viii] [35] illustrating the unpredictable delivery of CERs.

To manage the risks of buying carbon credits, specific know-how is crucial. Transactions are therefore normally handled by intermediaries and specialized organizations. CDM services firms, i.e., specialists, brokers, consultants, niche players, legal advisors, information providers and traders represents the biggest clusters of actors in arenas such as the CDM Bazaar.[36] Integrated firms and traders not only purchase and sell, but also provide financial intermediation and sometimes technological solutions. Some assist project developers through the entire value chain, while others focus on the end-trade. They come primarily from the Annex 1 countries, and work with global carbon asset development, raising capital and offering carbon services to project developers in return for future CER sale cash flow or forward deals for primary CERs. While brokers buy and sell for clients with compliance obligations[ix] through exchanges or over-the-counter (OTC), traders buy or sell directly, e.g. for investment banks, usually to make a profit based on short-term trends in the market, Companies have to open a dedicated trading desk to trade. These trades have facilitated a significant amount of carbon investment funds; also speculation oriented ones.[37]

Financial intermediation involves facilitating market participants, managing risks associated with trading and providing liquidity.[38] The trading companies and brokers (e.g. large banks like Barclays) trade on behalf of clients for financial diversification and risk mitigation[39], but a distinction may be made between those who trade for compliance and those who trade for private gains. Credits are often offered as diversified portfolios, where different projects carrying different degrees of) risks are bundled into tranches. Such structuring activities is driven by market participants who clearly understand project and buyer risk profiles. These service providers are necessary; the amount of regulatory complexities means that the risk of entering into trade is too high without detailed know-how of the market. Hence, securitization and trading with derivatives in emission markets is mainly limited to professional and sophisticated investors.[40]

Trades can be categorized into spot, options, forward and future, where forwards, futures and options are types of derivatives. Spot trading occurs after CER issuance, and sell at a high price. Forward trade implies a sale of credits for future delivery and payment. This means a discounted price compared to spot, and the buyer assumes the risk of delivery. Futures is a sophisticated arrangement of trading and a derivative contract for a fixed future price, which hedges price risk and often include penalties for non-delivery. Options are a contractual opportunity to buy CERs at a certain price and time. Futures were first introduced in the EU ETS in 2005, and they soon represented the majority of trades. In 2009, 72 % of all EUA[x] transactions and 89,4 % of CERs traded were futures.[41] CER futures and options came in 2008.[42] In addition, the banking prohibition between the first and second phase of the EU ETS meant that different categories of derivatives developed.[43] [xi] Existing clearing infrastructure for other OTC derivative markets is usually used also for carbon trades.[44] Most of the trades involving CERs are OTC, more so than with EUAs. This means a lot of data about trading arrangements are unavailable.

Futures traders may be hedgers (who wishes to reduce the risk of price changes and delivery) or speculators trading derivative contracts, and it is difficult to separate these two categories in the carbon markets. The opportunity to hedge invites a wide range of investors, seeking arbitrage

[viii] Furthermore, 12 percent were delayed to the approval process and 27 percent delayed due to issuance

[ix] CERs are valid for compliance within the European trading system up to 13.4% on average

[x] The European Union Allowance, i.e. EU ETS quotas, the tradable unit under the EU ETS.

[xi] One can separate between those that expired in the same phase, and those that matured into the next.

opportunities[45]. Arbitrage activities in the carbon market require expert knowledge, but these activities developed rapidly in the market.[46] The fast turnover of futures and characteristics of the market indicate that trading is used for speculative purposes, providing risk-free profit for arbitragers.[47] Market agents play especially close attention to the CER-EUA spread, due to arbitrage opportunities that arises from buying (cheaper) CERs and sell EUAs for compliance[xii].[48] Price volatility represents risk and results in hedging and futures contracts for risk management, to avoid unfavorable future spot prices and secure future delivery. Futures contracts for CO_2 emissions are also bought by financial actors in other sectors like energy, as carbon markets are interdependent with other markets.

Similar to developments in the financial derivative market, innovators have attempted to design new products to price the future[49] and the growing complexity especially in value chains has implications for transparency and may hide uncertainties and hazards. Specific actors and experts within their particular field may utilize their power to gain leverage and profit. Even states have used market power in order to improve deals for firms within their borders.[50] On the other hand, restricting derivative trading may reduce liquidity available for project developers, and hedging can reduce the cost of over- or under-compliance. However, the speculative traders by non-compliance entities poses risks to stability and may exacerbate price variability and lead to market manipulation.[51] These actors have different motivations from compliance traders, but an investigation of the impact of them is lacking.[52]

The bundling of credits may be seen as disconnecting the derivative and the underlying asset, complicating efficient carbon permit pricing.[53] It is reminiscent to the securitization strategies of house mortgages behind the financial crisis and represents a complexity that separates the traded product from the quality of the underlying asset – i.e., the actual emission reduction[54]. An assumption behind the carbon derivatives has been that futures trade can help *prefigure* the price, however, this "overlooks the extraordinary complexity in trading arrangements" and futures may even influence and drive expectations of future prices – effectively removing the relation between marginal cost of abatement and the traded credit, also in terms of time and place.[55] Experiences from e.g., electricity markets illustrate how markets can end up in multiple layers with complex institutional architecture, creating considerable transaction costs and impacts on price of emission reductions (ibid.). Furthermore, an analysis of the market show that there is a *negative variance risk-premium,* resulting in a demand for high compensation for the risk of volatility spikes.[56] In other words, actors pay large premiums to hedge against the many risks present in carbon markets.

While the CDM has undoubtedly raised a large sum of money for projects in developing countries, there is ample research showing several serious issues. The CDM was never about penalizing emission increases, and therefore resembles a subsidy that may create perverse incentives to increase emissions short term.[57] It has been pointed out that CDM is problematic in terms of the integrity of the offsets, i.e., low additionality; the need for increasingly sophisticated approval procedures; asymmetric information; failure to generate sustainable development outcomes[xiii]; and an opaque and bureaucratic structure.[58] It seems as if technology transfer has also been low.[59] Uncertainty in regulatory processes has undermined the functioning of the carbon market as well as additionality, since less additional projects become more attractive investment

[xii] While purely financial players can buy and sell, they cannot surrender credits for compliance into registries.

[xiii] All CDM projects should contribute towards sustainable development, defined by a set of criteria decided by individual host countries. While one sub-criterion could be the protection of biodiversity, research indicates that contribution towards SD has been limited beyond e.g. employment and supply of renewable energy.

objects due to risk aspects.[60] The market has experienced several security issues, like VAT fraud, phishing and reuse of already surrendered EUAs.[61] Such issues have been continuously addressed. Finally, several studies question the substantial transaction costs involved CDM[62].

3.5 Implications for financialization in biodiversity values

As far as we have been able to establish, there are yet no derivatives where biodiversity values are included as an 'underlying'. Proposals do, however, exist. The most noted one is put forward by Mandel et al. These authors argue that "Two major shortcomings of the US Endangered Species Act have led to inefficient use of conservation dollars: (1) it only provides conservation protection to distressed or rapidly declining species, and (2) it does not take full advantage of the market to reduce costs in conservation". [63] Based on this they propose that "The government, or a suitably funded NGO, issues derivatives based on the likelihood that the species will need protection in the form of land purchases, changes in land use, and/or rehabilitation".[64] The one investing in this derivative 'wins' if they ensure that a set threshold/cap is not passed. If the threshold is passed, the investor must take preemptive action. If one experiences unpredicted decline, investors forfeit their investment. Again, it seems to be a necessity that the state pays the return if it goes well.

The main argument behind the proposal regards increased cost-effectiveness – seen as an inherent quality of markets. It is also assumed that the derivative makes it possible to act long before a threshold is crossed. Finally, they note that offsets can be used to move the protection to land that is less costly to protect. They discuss also the possible effect of short-term volatility on the underlying asset and conclude that this does not seem to be a problem.

The argument that markets deliver cost-efficient solutions regarding nature values/ES is contested. We have already shown how important public actors are in PES and that this partly is due to reduced transaction costs when using public command instead of trade. We discuss this issue further in Section 4.3. It is moreover hard to see how the arguments regarding early action and offsets – to the extent that they hold – are exclusive to derivatives/a market solution. At a more fundamental level, Sullivan argues that "But it seems perverse to transform the value of species survival into a price whose rise or fall is entangled with bets on their susceptibility to irreversible loss, underscored by a calculus whereby species value rises with rarity, or greater risk of extinction".[65]

Box 2 Markets, knowledge and values

The issue has been raised whether markets/payments may change the values held among actors involved. Certainly, bringing in trade implies emphasizing the logic of gain and increased focus at 'what is tradable'. Two issues seem key:

- Markets are in general observed to imply erosion of reciprocal/community relationships and the traditional values related to these – e.g., Bowles (1998), Polanyi (1944). This is not least observed in relation to introducing paid labor. There seems to be no clear evidence that this is happening as an effect of e.g., PES. This may be because such changes take time and PES is yet to young a policy. A specific aspect of this regards the 'crowding out' environmental norms as a result of payments – e.g., Vatn (2010); Muradian et al. (2013). This issue will be discussed in Section 5.3
- Changes in knowledges and practices is the second aspect. Inclusion into markets changes people's relation to nature as changes in production shifts the perspective of what is now 'most valuable'. With this, existing practices and accompanying knowledges tend to be lost – see e.g., Godoy et al. (2005); Gómez-Baggethun et al. (2010b).

Hence, the question of financialization is as much about what kind of motivations for action are 'brought into play' and what role and responsibility different actors – public and private – get given the type of policies put in place. Hence, the fundamental issue regards the relationship between environmental protection/sustainable use and democratic processes. An illustration of the issues involved is found in the proposed index-linked carbon bonds to be issued by governments.[66] The idea here is to hedge against the risk that governments themselves do not meet their emission reduction obligations regarding climate gases. The idea behind this bond is that governments are financially punished if they do not reach targets – i.e., they have to pay higher interest rates on the bonds.

A. Tickell argues that while the public can to some extent control the markets for derivatives through developing and enforcing rules, he notes that turning to this form of policy also represents a shift in power relations towards strengthening the position of traders.[67] It moreover includes a shift in logic as well as in language where price is given priority over other ways of capturing and thinking about nature values. This has – quite understandably – created a heated debate, as a question has also been raised about the potential shift in knowledges and values – see Box 2 for a brief exposition.

4 Socio-economic effects of different instruments

In this chapter, we turn to socio-economic effects of different economic instruments. The chapter is divided in three. First, we present the results of a review on distributional effects of payments/markets in ES. Second, we present some experiences from Costa Rica regarding the potential of using specific criteria for protecting the poor in PES. Finally, we offer insights regarding the level of transaction costs as linked to various economic instruments.

Are all the socio-economic interests at the table?
Illustration Javier Sáez
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4.1 Distributional aspects

Regarding distributional effects, we look at three issues:

- Who is paid?
- What is the level of payment?
- What are the format of the payments?

This part of the analysis is delimited to research on PES and CDM – as we consider these systems to be of the greatest value to policies regarding biodiversity. In the latter case, only forest oriented projects linked to afforestation and reforestation are included, termed CDM_{AR}. Most of the papers utilized are (multiple) case studies, while a few are reviews. A quite consistent picture evolves from this material. However, the precision by which we can draw conclusions is somewhat limited as most analyses are qualitative or use different methods, which makes quantitative comparisons difficult. This reflects the characteristics of the topics studied. We finally note some disagreement regarding especially the issue whether payments cover costs.

4.1.1 Who is paid?

With respect to 'who is paid', the overall picture is quite consistent. First off all, receivers of payments are owners or users of land including concessionaries (in the case of plantations).[68] Most receivers of payments have a formalized property right and having this right seems to be a great advantage for being eligible to participate in PES and CDM_{AR} projects. Since community 'ownership' is dominantly informal – i.e., customary rights – private ownership dominates in PES and CDM_{AR}. Communally owned land has not been well represented in PES schemes.[69] This is often due to lack of formalization/lack of accepted borders. Cases where communities are paid are, however, observed,[70] and there is a trend towards increased inclusion of land under common property. Prioritizing smallholders, making landholders who can document possession, although not full legal tenure, is also a feature of some PES programs.[71] Therefore, use-rights may suffice, but overall it is a disadvantage for participation to not have formalized rights.

The poorer segment of landowners or -users is underrepresented in PES/CDM_{AR}. This seems largely explained by three factors:

- The poor rural population are either landless or they live in communities holding customary rights only. According to the above, this tends to result in these people being excluded from participating in PES/CDM_{AR} projects[72]
- The costs of participation varies in a way that systematically disfavors the poor.
    - They typically have no more land than what is needed for survival and cannot afford setting aside land for conservation/their opportunity costs are high[73]
    - Transaction costs per 'unit of protection'/ton of carbon is relatively higher for trades in small volumes[74]
    - Participation in some projects demands high start-up costs, which are difficult to cover for the poor[75]
- Finally, participation in PES/CDM_{AR} demands a level of knowledge/access to information that the poor often do not have.[76]

While it has been argued that PES could be a win-win – ensuring both ES delivery/nature protection and poverty eradication – the above implies that difficult trade-offs are involved. Alex-Garcia et al. are illustrating the dynamics very well regarding Mexico's PES program (PSAH).[77] The poor often live in areas with low deforestation and are relatively costly to involve. They show, however, that by targeting communal land, trade-offs can be substantially lessened. Similar observations and arguments are offered by others.[78] One important aspect is that this reduces transaction costs.

Women are underrepresented in PES programs. They often lack property rights. Commonly owned land is moreover often under a patriarchal system of decision-making.[79] A notable counter-example is the Bolsa Floresta program in Brazil where payments go exclusively to the female household head.[80] The PES program in Costa Rica (PSA) has also taken measures to increase the rate of women included.[81]

Another vulnerable group is the landless, surviving on renting land. As PES may increase the value of (marginal) land, the landless may be unable to pay what is demanded and are put in an even more difficult situation.[82] It is also significant that where money is paid to communities, there is the risk of 'elite capture', emphasizing the need for ensuring transparency. Jindal et al. note that this is especially a problem where property rights are unclear.[83]

In the case of plantations receiving payments for carbon sequestration, these typically go to the concessionaries.[84] Moreover, there are a series of cases where establishing plantations have resulted in excluding local communities from access to the land against low or no compensation.[85]

We observe that the type of buyer/intermediary and aims of programs are important for distributional effects of PES/CDM_{AR}. The following examples illustrate this:

- Bosselmann and Lund[86] document a study of three intermediaries in the Costa Rican PES program (PSA). They note that the public intermediary (county agricultural center) and a producer cooperative had a higher inclusion of the poor than a trust fund. They note three main reasons for this: 1) the previous networks of the involved intermediaries; 2) the framing of the role of the intermediaries as stipulated by the national PES policy framework; 3) the values of the intermediary.
- McAffe and Shapiro[87] similarly documents developments within the Mexican PES (PSAH) program from the 1990s and onwards. The role of social criteria has been conflicting especially between the World Bank as a funder and the local communities/the

Mexican state. The former has wanted to emphasize environmental efficiency, while the latter put stronger emphasis on the social/developmental aspects. The rules changed over time due to variation in the influence of the different parties. However, Pagiola and Platais[88] emphasize that the World Bank has moved away from what they term standalone PES projects.

- Corbera et al.[89] document a carbon forestry project in the state of Chiapas, Mexico, run by Fondo Bioclimático. At the outset, the fund received 'additional non-carbon funding'. When this funding disappeared in the late 1990s, there was a shift from a combined focus on the environment and community well-being towards carbon contracts with a focus on sequestration only.
- Krause and Loft[90] conducted a study of the Ecuadorian Socio Bosque program, a program with a strong development/social profile. Their analysis shows the importance of this focus of the program, but the rules are still not ensuring full participation among the poorer segments of society.

From the above, one can conclude that to ensure participation among the poor is demanding and that it is necessary to institute specific mechanisms to facilitate their participation in PES/CDM_{AR} type projects. This conclusion has quite general support.[91] However, no 'bullet safe strategy' exists for successful inclusion. Sensitivity to local contexts seems especially important.

4.1.2 What is the level of payment?

Regarding the level of payments, the standard question is if costs for 'sellers' or 'providers' are covered. The literature focus almost exclusively on opportunity and transaction costs. It is generally assumed that if these costs are covered, PES and CDM_{AR} could result in a win-win – i.e., ensure both increased delivery of ES and reduced poverty. We find this argument flawed; if cost are covered, people are 'just' equally well off. As the payments go through a set of intermediaries, we expect that they would be in a stronger position to capture rents.[92] We therefore conclude that payments in general need to exceed opportunity and transactions costs for sellers/providers to offer the necessary basis for a 'win-win'.[xiv]

Regarding the level of payments, we observe that several authors state that payments are lower, in some cases much lower than opportunity costs.[93] Note that sellers will encounter some transaction costs that come on top of the opportunity costs. Wunder and Alban[94] argue against this conclusion. Their reasoning is mainly based on the argument that trades are voluntary and sellers would not sell if they lose. They also point at the fact there may be other potential gains to sellers than the payment, e.g., increased tenure security, local gains from conservation, strengthened relationships between buyers and sellers – e.g., between up-stream and down-streams dwellers in the case of water protection.[95] These issues are important. We need, however, to mention that while

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[xiv] Using the so-called rent driven model of land use, Alix-Garcia et al. (2014:25-26) do a theoretical analysis producing the following conclusions implying that the poorer segment of a middle income group may capture rents (point 3):

"1. Environmental effectiveness will be higher where the returns to agriculture/pasture production are higher, i.e., where geographic deforestation risk is higher.
2. If the correlation between agricultural/pasture rents and poverty is negative and the program is targeted to high risk areas, then middle income landowners will enroll the largest amount of land, the poorest will be excluded, and program effectiveness will be higher among wealthier landholders.
3. In this scenario, there will be greater wealth impacts in areas where deforestation risk is lowest, but within properties enrolled, relatively poorer beneficiaries will gain more wealth."

increased tenure security is a good thing for those acquiring it, formalization may reinforce existing inequalities, even result in exclusion as observed in several African cases.[96]

The argument that voluntary trades per definition must result in a better (or at least as good situation) for both parties, is also questionable. There are rather strong arguments against assuming that people involved actually maximize profits.[97] In a recent study by Janet Fisher of a PES program in Uganda, she interviewed participants about their basis for participating: "Only 11% of participants reported calculating the profitability of the carbon strategy compared to other land uses. A much higher percentage (43%) reported that although they hadn't calculated anything, they were motivated by the future benefits of the project, often citing timber. The remainder (46%) said they had made no calculation, just planted".[98]

The Fisher study reports another issue often recognized in the literature – the fact that while payments often cease after a few years, it is assumed that protection continues. It is, moreover, typically unclear for the participants what the terms of the project is – e.g., its length. Hence, 'sellers' may not realize the full costs or they may not plan to abide by the rules in the longer run. This is a problem for the durability of projects.

While PES projects typically do not cover (all) costs for sellers, it might not be so bad that the poor do not participate. They simply avoid losses, indicating again that the 'win-win idealism' so often encountered is based on weak premises. At the same time, the poorest of the poor – the landless – cannot use 'non-participation' as a way to avoid losses. If they lose, that is an indirect effect of landowners in the area engaging in PES/CDM_{AR}.

4.1.3 What are the formats of payments?

Looking finally at our third question – the format of payments – we observe a dominance of cash payments, while there are also some in-kind transfers.[99] More specifically, we observe:

- In the case of private buyers – typically called 'users' – there is a greater tendency to pay according to what is delivered, while public payments are typically flat rate – maybe differentiated according to type of land-use.[100] Wunder et al.[101] argue that the former shows that 'user-based' payments are more efficient than public programs. This is hard to say. The latter may actually be as efficient since the flat rate payment reduces transaction costs. Public programs typically 'pays' for more than the environmental effect and there are indications that flat rate is more acceptable among 'providers'.[102] We note, though, that this conclusion may be quite context dependent.
- In the case of payments to communities, there are several internal distributional issues:
  - Firstly, we have issues regarding how much is paid to individuals and how much is retained to finance community projects related to infrastructure, education, health care etc. We observe a variety of ways that decisions are made and solutions chosen. While the intermediary seems to play a substantial role, the internal trust in the communities seems to substantially influence how much is allocated to community projects[103]
  - Secondly, we have issues regarding the role of opportunity costs as a basis for defining who gets how much. While these costs clearly varies a lot among members of a community, we have found no case where individual payments are made dependent on these costs. The normal system is a flat payment per household or household member. Vatn et al.[104] document a case from Tanzania where this actually seems to have had an internal positive effect on distribution since the richer

charcoal makers were compensated similarly to those using the forest mainly for collecting fuel wood to own consumption.
    - Finally, we need to emphasize the issue of 'elite capture' and corrupt practices, as there are several examples of leaders enriching themselves.[105]
- Several authors emphasize that just paying individuals does not help much for development nor the goals of protection. Changing practices and creating new income opportunities is a profound process that demands collective action and training.[106] Hence, a wider engagement than 'just paying' seems warranted.

4.2 How may specific criteria for protecting the poor work?

In this section, we discuss Costa Rica's use of specific criteria for targeting its PES program to poor landholders based on work by Porras and colleagues.[107] While the PES Programme has never been designed as a pro-poor program, it is nevertheless bound by social objectives. At the Constitution level, the State must seek to promote welfare and balanced wealth distribution (art. 50). The Forest Law legally created PES (art 3) and defined the mission of FONAFIFO (the institution that manages PES) to be in the benefit of small and medium producers (art 46). Because funding must be annually approved by the National Assembly, the program is portrayed as a mechanism to promote rural development. Lastly, the country's intention to enter REDD+ requires it to follow a strict protocol in terms of poverty and justice.[108]

In practice, program managers have tested a series of measures to promote participation of relatively vulnerable landholders and improve the social outcomes of the program, rather than as "pro-poor" targeting. In Appendix 4.1 we describe these features in relation to the different stages of a PES contract planning and evaluation cycle (i) eligibility: size and tenure (ii) targeting PES pre-applications (iii) choice of payment modalities (iv) contract terms (including transaction costs, payment modalities, length) (v) terms of renewal (vi) and impact evaluation.[109]

Until now, there are no impact evaluation studies looking specifically at social issues at the national level. But looking into contract distribution and allocation between 1997 and 2012, the study by Porras et al.[110] provides a glimpse on the time impact of the social criteria introduced by the program managers. Very small landholders that depend on land for their livelihoods (e.g. through agriculture) are excluded from PES except through agroforestry. Maximum contract size limits the agglomeration of contracts in single physical owners, but not for corporations which can own land anonymously through subsidiaries or family networks.[111]

The eligibility requirement to own land is in itself an indicator of wealth. Most studies in Costa Rica find that PES participants are wealthier than non-participants although some poorer regions show large heterogeneity with large property owners having no formal education.[112] Small holders may pay for legal assistance to properly register boundary conflicts by borrowing on future PES payments. PES may therefore have the positive social impact of indirectly supporting land tenure regularization.

Between 1997 and 2012, FONAFIFO distributed approximately US$340 million as PES. The greatest part –and increasing - of these funds went to legal entities (i.e. corporations 49 per cent), followed by individuals (31 per cent), indigenous groups (13 per cent) and cooperatives (7 per cent). See Figure A4.1 in appendix.

With respect to priority criteria assigned to small properties, a recent study by Porras et al.[113] looked at the distribution of PES contracts relative to property value. They found that smaller properties have higher per hectare prices, and relatively pricier areas are more likely to be fragmented into smaller plots. Attributing priority in the application process to properties of less than 50 hectares will not necessarily convey a measure of the landowners' relative vulnerability (see Table A4.1 in appendix for examples of heterogeneity of prices in 50 hectare? properties).

4.3 Transaction costs related to different economic instruments

Transaction costs (TCs) can be defined as the costs of interaction (cf. Section 2.1) – typically, costs of information gathering, making agreements between parties[xv] and controlling that agreements are fulfilled. Moreover, interactions need a basis form where to interact – i.e., creating rules for systems like cap-and-trade, PES, biodiversity offsets. These costs are part of TCs as we use the concept. Finally, while transactions are typically thought of as trade, we use a wider definition here including also transactions through e.g., command.

TCs vary with the institutional context, the format of the transaction, what is transacted/how specific the service is, how often a transaction is undertaken and the size of it. In the case of ES, rather high levels are observed – e.g., Wunder et al.[114] conclude that for PES, the level of TCs tend to lie in the order of 30-100 % compared to the payment. The authors find that set-up costs are typically higher than running costs. This reflects that many of these payments are rather idiosyncratic. In the case of CDM, Michaelowa and Jotzo[115] indicate TCs in the order of 0.3-1 Euro per tCO_2 for large projects (20.000-200.000 tCO_2 per year) to 10 Euros per tCO_2 small ones (2.000-20. 000 tCO_2 per year).

In trying to clarify what explains the variation in costs, we will start by looking at the institutional context/type of transaction. Markets/trade are thought to reduce TCs compared to e.g., command. While this is so for ordinary commodities, the situation is often different for public goods like ES where command may be more efficient. We have already seen this in the case of PES (section 2.3). In the case of water quality, a typical example is a public water utility adding an extra fee to the water bill raising the money necessary for paying upstream farmers to change land use. TCs may be very low in such a case. Environmental taxes will typically have lower TCs than cap-and-trade systems as one avoids the cost of establishing and running the market. In the case of biodiversity offsets, there is yet no firm basis on which to conclude. Some public systems have been criticized for being overly bureaucratic. While the public must anyway play a strong role as regulator of these systems, we think it is possible that TCs of biodiversity offsets involving trade – i.e., habitat banks – are lower than for pure public systems.

Regarding what is traded, linking payments to an already existing commodity may reduce TCs substantially – e.g., input as opposed to emission taxes. Rørstad et al.[116] document a fertilizer tax where the TCs are just 0.1 % of the level of taxes gathered. This effect is linked to the complexity of the good, as increased complexity demands a much more elaborate set of measurements that may have to be done differently for each single case. Certainly, here we encounter an important trade-off. TCs can be reduced substantially by measuring simple proxies like land under certain management. Using proxies comes, however, often against a large reduction in precision.[117]

The volume traded is maybe the most decisive factor for the size of TCs as a large fraction of these costs is fixed – cf. the data for CDM offered above. In the cases they have looked at, Michaelowa and Jotzo[118] note that size has much larger impact on the level of TCs than the type of project. These were mainly different types of energy projects (carbon). Furthermore, where a methodology for estimating emission reductions was not in place, this increased TCs substantially.

[xv] In the case of legal regulations, the state has the power to command the conditions

Coggan et al.[119] document similar findings for development offsets, while noting that asset specificity (biodiversity) plays a very important role. The same is the case with the policy framework. It is the relatively high level of fixed costs that 'drives' the above conclusions. Regarding policy framework/ institutional context, fixed costs may vary substantially from case to case. The one extreme is an already established system – market or public – where a minor addition is the only thing needed – cf. the water fee and fertilizer tax mentioned above. In other cases, systems must be built form 'scratch'. In the case of PES, we have seen earlier that one may have to start even at the level of clarifying property rights.

The high per unit TCs following from small quantities is a great obstacle against involving e.g., smallholders in the case of forest projects. As already emphasized, various forms of collective agreements – like basing agreements on communities instead of individual households – have been documented to have substantial effects on the level of TCs.[120]

5. Unintended effects on biodiversity of different economic instruments

Unintended effects of a policy mess on the forest frontier
Illustration Javier Sáez
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In this section we discuss the theories behind and available evidence of unintended conservation effects. We divide the discussion into indirect effects at the actor and landscape/system level. At actor level indirect effects include (5.1) shifting of motivational structure in landusers; and (5.2) effects on actors through social networks. At landscape level they include (5.3) indirect effects on biodiversity conservation priorities of targeting conservation instruments at ecosystem services provision and (5.4) spillover or slippage effects through indirect changes in returns to landuse and scarcity. Combinations of indirect property level and landscape level effects occur through (5.5) interactions with the policy mix that governs landuse.

In this chapter we focus at PES as defined in Appendix 1. Using one type of economic instruments as a reference facilitates a compact presentation of the multi-dimensional causes of 'unexpected' effects. From the point of view of the seller or provider of biodiversity conservation at the property level, incentive effects should be similar whether the payment for restoration of habitat originates from PES or from sale of biodiversity offsets (BDO), or likewise whether payments for avoided deforestation originate from PES or the sale of tradable development rights (TDR). However, PES, BDO and TDR may differ markedly in their aggregate effectiveness depending on the definition at landscape level of the conservation objective or development cap, and the extent of the market for trading. Landscape level effects of BDO and TDR have been discussed in the previous report[121].

Furthermore, we have focused our analysis on identification of unexpected impacts of PES. We have not conducted a systematic comparison of these impacts with instruments based on legal rules or information (see Table 1), although we make note of comparisons in several places.

5.1 Unintended effects of economic instruments due to motivational crowding

Economic incentives for conservation and sustainable use are intended to change private landusers motivations. This section discusses empirical evidence of unintended effects of economic instruments on biodiversity conservation due to complex interactions with intrinsic motivations. The focus is on motivational effects of incentives on actions that lead to landuse and biodiversity change. Defining what constitutes 'unintended' or 'unexpected' is difficult because it is a subject specific concept. However, simple micro-economic theory would expect landusers' motivations to be entirely determined by the net private benefits of landuse change prospects and the size of positive and negative incentives relative to their private net returns (see the definition of the rational economic agent in Chapter 2). Unexpected effects relative to this theory - ignoring many years of behavioral economics research - would be any kind of agent behavior not determined by net benefits of prospects.

Unexpected environmental effects may occur when the extrinsic motivator of an economic instrument interacts with or changes intrinsic motivations for biodiversity conservation. Several characteristics of economic instruments may lead to changes in intrinsic motivation, but the discussion here focuses specifically on the evidence about monetary transactions. For instance, 'crowding-in' effects mean that the monetary transfer strengthens the biodiversity conservation incentives provided by intrinsic motivations, while 'crowding-out' effects mean that the monetary transfer reduces those intrinsic motivations. This section is based on a review by Rode and colleagues[122] and additional studies.

Motivational aspects of economic instruments for biodiversity conservation remains an under-researched area[123]. Rode and colleagues conduct a search for empirical studies addressing instruments for biodiversity protection which tested for motivation crowding effects. Of a total sixteen papers from Mexico, Nicaragua, Colombia, Peru, Bolivia, Uganda, Namibia, South Africa, Madagascar, Cambodia, Northern Australia and Ethiopia, that fit these criteria, ten studies reported crowding out effects. Three of these studies present statistically significant results[124]. A choice experiment in Tanzania by Kerr and colleagues[125] found that small positive incentives lead to a lower participation in tree planting than for a baseline group that did not receive compensation. For crowding-in effects, only one experimental study presented statistically significant effects[126]. In the latter study, a proposed fine on local community overfishing which was never actually implemented, nevertheless reduced extraction for several periods after it had been mentioned. A general methodological weakness is that few studies report on the baseline state of intrinsic motivations to conserve biodiversity.

Hiedanpää and Bromley[127] identify Finland's Nature Value trading as an example of crowding-in as the PES programme enabled new mental habits concerning forest biodiversity protection as an economic activity. The commitment to safeguard biodiversity became part of a renewed culture of forestry. Primmer and colleagues[128] report on further factors determining Finish forest owner's propensity to participate in the Nature Values Trading programme. In their study forest owners with high intrinsic motives towards biodiversity protection consciously stayed out of the programme. They suggest that 'staying-out' as a motivation was even strengthened in the non-participant group.

Hiedanpää and Bromley[129] look beyond motivational crowding to characterise PES as an 'inducing transaction' for breaking environmentally harmful habits. They recognize that beyond the 'habit-breaking' incentive of PES to avoid deforestation or start restoration, the harder challenge is 'habit-making' of sustainable landuses after the incentive has ceased. In this light, a study from Colombia showed that silvopastoral conservation practices had greater permanent effect the longer the PES incentive had been in place[130]. Authors emphasize the importance of technical extension services accompanying the economic incentive.

Are findings from the literature review transferable to other countries? Rode and colleagues find the evidence inconclusive about the conditions under which incentives can undermine or reinforce intrinsic motivations for biodiversity conservation. Baseline instrinsic motivations in the prior unpaid voluntary are expected to vary from place to place. Further discussion of the psychological mechanisms explaining crowding-out and crowding-in effects – for both voluntary and coercive instruments – can be found in Appendix 3.4. We have not researched whether there are systematic differences across cultures in these psychological mechanisms. Motivational crowding is expected to depend on some basic instrument and respondent characteristics:

- Coercive versus voluntary economic instruments
- Participation status (participant, non-participant, ex-participant)

Also, research on framing effects on choice and values[131] would lead us to expect that the incentive characteristics and landuse change prospects will affect the psychological mechanisms for motivation crowding:

- Size of payment relative to the private net benefits of the action
- Perceived risk of the landuse change prospects
- Landuse change context including
    - *action versus no action* regarding landuse change
    - avoiding a *loss* (of forest cover) *versus* obtaining a *gain* (in forest cover)
    - *compensation* of foregone net private benefits of no action, *versus payment* of net public benefits of a landuse change

What economic instrument design lessons can be learned from this rapid review? Motivational crowding-out effects seem to be more diverse than crowding-in effects. With no site specific information or adaptation, economic incentives are likely to be less effective than predicted by a simple comparison with private net benefits of landuse change. Motivational crowding effects apply to coercive regulation-based instruments as well and should not be interpreted as an exclusive critique of economic incentives. Technical extension, adaptation and learning opportunities are motivations for voluntary participation in and of themselves, but also likely to increase the effectiveness of PES.

5.2 Unintended effects of social networks on agent behavior

The unexpected effects of different types decision-making behavior in social networks that does not conform to expectations of the rational economic agent's decision-making is an emerging topic in the literature on conservation effectiveness of economic instruments. Some further justification for discussing unintended effects of social networks and their types is given in Appendix 3.5. Neighborhood effects in the PES literature include mechanisms by which economic instruments have an impact on the spatial neighborhood of a property through market mechanisms affecting substitution, scarcity and labor market, e.g. (this is addressed in section 4.3). Studies reviewed in this section draw on data from Scotland, the Solomon Islands, Uganda and Costa Rica.

Here we focus on evidence of social networks influencing participation in PES, which addresses how local 'small-world' social networks affect PES participation. This research argues that measures of social connectivity should complement prioritizing conservation action alongside conservation features and opportunity costs[132]. However, few studies have utilized social network analysis in the wider field of systematic conservation planning initiatives. A rapid literature search[xvi] revealed a few published papers with empirical results on the role of social networks on the effectiveness of PES and economic instruments. There is a much larger literature on

[xvi] We searched using Google and Science Direct for the terms (social networks OR neighbourhood effects) AND (PES OR "payments for ecosystem services" OR "payments for environmental services")

'neighbourhood effects' in adoption of successful farm practices which does not address economic instruments for conservation explicitly, e.g.[133].

Provision of ecosystem services depends on the spatial configuration of interventions in the landscape and be extension the spatial scale and network configuration of multiple farms. A study by van der Horst[134] estimates the extent and relative importance of local land owner networks on entry into ecosystem services payment schemes in Scotland. The author finds evidence of spatio-temporal clustering in the uptake of ecosystem service payments, with micro-clustering and neighbourhood effects being stronger in small and remote communities.

Evidence that social networks are important for PES participation is also provided in the literature on factors determining programme participation. Arriagada and colleagues[135] find in Sarapiqui, Costa Rica that 66% of non-participants state 'lack of information' as the main cause, followed by the system being 'too complicated' (15%). Economic motivations of payment too low or fees being too high was not a dominant reason for non-participation (11%). Literature on social networks describes how copying behavior is more likely under conditions of high uncertainty and complexity and facilitated by social networks[136]. We can speculate whether uncertainty and complexity of landuse decisions may be the reason behind the observation that many do not calculate their costs and benefits in landuse decisions involving PES as observed by Fisher (2012) in PES program in Uganda (discussed further in section 4.1.2).

Robalino and Pfaff[137] find evidence that deforestation on neighboring land can significantly increase the probability of deforestation. They describe "strategic complementarity" as a possible pathway for neighborhood spillovers that relates to social networks. Strategic complementarity can occur when farmers acting as a group can increase bargaining power for buying inputs and selling production from cleared land. Tourism facilities copying each other in maintaining forest cover is another example of strategic complementarity. Person-to-person learning in the neighbourhood is also cited as a possible cause.

On the other hand, Michel[138] found a majority of respondents unaware of PES contracts within a distance of 0.5 km from their residence, suggesting that information sharing networks in the vicinity were almost non-existent. Contact with family and friends constituted a significant source of information concerning PES, but the effect was not stronger than contact with intermediaries and media.

Daniels and colleagues[139] and Sierra and Russman[140] provide evidence for the importance of social networks around intermediaries role in providing information access to PES. Landowners participating in pre-PES incentives were disproportionately represented in early PES cohorts in Costa Rica. Garbach and colleagues[141] find that PES participants have a significantly larger number of social sources of information than non-participants, and the number of social sources exceeded institutional sources of information in all groups.

Our review of factors determining enrolment in voluntary conservation schemes suggests that social networks have been studied to a limited extent. This may be because much of the literature on PES effectiveness has focused either on farm or farmer characteristics. Research on social networks suggest they may be an important source of motivation, as well as directly providing 'recipes' for landuse management that are easy to copy in complex situations. The spatial extent of social networks and neighborhood effects relative to the spatial extent of ecosystem services in PES design is also an under researched field.

What economic instrument design lessons can be learned from this rapid review of social network research? In contexts with strong community organization, network effects are likely. In addition, if landuse management decisions involve complex trade-offs between different interests, and the outcomes or landuse change are uncertain, copying behavior may be more likely than rational net benefit calculation. In such situations economic incentives are less likely to have their intended effect.

5.3 Unintended biodiversity impacts at landscape level of economic instruments targeting ecosystem service provision

5.3.1 Background and assumptions about unintended conservation effects

The effectiveness of economic instruments to conserve biodiversity and/or ecosystem services and attain sustainable levels of use of natural resources relies on the definition of the conservation goals. Maestre and colleagues consider for instance, that a biodiversity policy is effective if it produces *a priori* defined conservation benefits. However, because the process of defining the conservation benefits encompasses many actors, views and values[142] conservation objectives are often defined in a broad sense to avoid conflict[143]. In addition, there is considerable uncertainty about which are appropriate levels of use to reduce the current, unsustainable level of pressures on natural resources[144] and about what portions and amounts of biodiversity need to be conserved.

Some important characteristics of the conservation problem must been borne in mind when considering both effectiveness and unintended effects of conservation policy instruments. First, conservation objectives often range over spatial scales, and they therefore involve decision-making at various levels of governance. For example, biodiversity conservation strategies require both the inclusion of individual species, functions or lineages in protected areas, but also sets of interacting species over large areas[145]. Similarly, the quality of pollination services depends not only on the local conditions, but also on the characteristics of the surrounding landscape at various scales[146].

Second, due to the complexity of the conservation problem, overall ecological sustainability goals are operationalized as multiple conservation objectives (e.g. see Aichi targets in Appendix 3.1,[147]) and various indicators of biodiversity conservation value and ecosystem service provision are used. Examples include the restoration of degraded ecosystems[148], the enhancement of multiple ecosystem services provision – including climate mitigation – [149] and the conservation of particular habitat types, taxonomic, functional and phylogenetic diversity[150]. Multiple conservation objectives are needed both because of the difficulty of managing complex ecological systems and because the anthropogenic factors that drive biodiversity loss are many. In these cases, mixes of policy instruments that deal with the various factors of biodiversity loss are needed[151]. In this context, it is important that the aims of the particular economic instrument – and how accurately they are defined – are seen as a backdrop for the analysis of unintended effects.

Thirdly, ecological systems are dynamic and processes manifest at different temporal scales. There is much uncertainty about how long-term dynamics responds to both natural and anthropogenic factors due to the lack of comprehensive ecological data spanning over long enough periods. Unintended effects of economic conservation instruments can occur due to the uncertainty around predicted or intended outcomes of the conservation actions. In this context, instruments that enable tracking of impacts, learning and adaptation at the various levels of decision-making are likely to be more robust and resilient than instruments than do not facilitate these processes.

In the following, we focus on the context conditions that can determine unintended impacts in terms of biodiversity conservation outcomes. Data are drawn from a number of meta-analyses (Appendix 3.2) with examples from Norway, California, Nicaragua, Costa Rica and Brazil.

5.3.2 Bio-physical conditions that can result in unintended effects on biodiversity and ecosystem service outcomes

Overall conservation goals are operationalized as various, often independently established, conservation objectives accompanied by a series of indicators (e.g. occurrence of populations, species, habitat types and ecosystem/land-cover properties linked to the provision of ecosystem services). Consequently, the degree to which conservation features and the provision of ecosystem services overlap in space will critically determine the level of co-benefits and trade-offs that will take place when economic instruments are implemented in a particular landscape, the 'policyscape'[152]

Studies that have assessed the degree of co-occurrence of conservation features and ecosystem services indicate moderate overlap and demonstrate the existence of trade-offs when implementing policy instruments. This situation can potentially lead to unintended negative effects on certain conservation features when single or a sub-set of objectives are targeted. For instance, Chan and colleagues explored the trade-offs and opportunities for aligning conservation goals for biodiversity with six ecosystem services (carbon storage, flood control, forage production, outdoor recreation, crop pollination, and water provision) in the Central Coast ecoregion of California, United States[153]. They found that targeting ecosystem services directly could meet several ecosystem services and biodiversity goals, but that targeted biodiversity protection was necessary in addition since biodiversity losses of 44% were expected relative to targeting biodiversity alone. Schröter and colleagues and Locatelli and colleagues arrive at similar conclusions regarding the partial spatial overlap among different ecosystem services and biodiversity conservation priority areas in Norway and Costa Rica, respectively[154].

Which ecosystem services are considered seems to be important in determining the degree of spatial congruence. Locatelli and colleagues found little correspondence between areas of high carbon sequestration capacity and priority areas for conservation. Also, the studies by Chan and colleagues[155] and Schröter and colleagues[156] show that while priority areas for biodiversity conservation had high capacity to mitigate carbon emissions, areas with high carbon sequestration and storage capacity had low overlap with conservation priority areas. At the global scale, Venter and colleagues[157] also conclude that there is a trade-off between cost-effective climate mitigation measures that aim at preventing deforestation (carbon storage, not CO_2 sequestration) and biodiversity conservation: "if REDD focuses solely on cost-effectively reducing carbon emissions, its benefits for biodiversity are low, protecting only slightly more vertebrate species than if funds were allocated at random among forest-losing countries" and that "if the same REDD funds were targeted to protect biodiversity, almost four times the number of species would be protected". Some of these trade-offs may be attributed to the criteria used to identify biodiversity conservation priorities[158]. Taxonomic diversity (number of species, diversity of species) has most commonly been used to represent conservation objectives, but other facets of biodiversity such as functional diversity correspond more closely with levels of ecosystem service provision[159] (Table A4.3.2 – Appendix 3.2).

Trade-offs are also likely to occur among different features of biodiversity that need to be protected. For instance, the spatial distribution of taxonomic diversity corresponds poorly with phylogenetic diversity, an aspect of biodiversity that reflects the evolutionary history of species and bio-geographical distribution patterns[160]. The spatial incongruence leads to taxonomic diversity and the evolutionary history of terrestrial vertebrates being unequally protected, even in Europe with the world's most extensive protected areas network. In the tropics, plant and animal taxa generally respond differently to environmental parameters[161].

5.3.3 Context conditions that may affect the occurrence of unintended effects on biodiversity and ecosystem service outcomes.

The degree of overlap and of potential spatial trade-offs between conservation objectives of biodiversity and ecosystem service is context-dependent. Some broad dimensions of such context are:

Forest transition stage: What to protect and where may depend on the forest transition stage. In an early stage with high levels of forest cover and low rates of forest conversion, the spatial correspondence is defined by naturally occurring patterns of biodiversity features (taxonomic, phylogentic, functional diversity[162]). In later stages with a stable landuse mosaic, the resulting correlations between biodiversity and ecosystem services has been created by repeated landuse changes. Decisions about whether to direct instruments to non-use areas (protected areas) or economic instruments to partial use areas (multi-functional landscapes) involves trade-offs about ecosystem services and biodiversity conservation priorities[163]. Fine-grain, multi-functional agro-pastoral landscapes have the potential to integrate elements of biodiversity conservation with regulating services such as sediment retention[164], maintenance of soil fertility[165], pest control[166] and provisioning services (food and fiber production). Although these landscapes do not maintain all biodiversity[167], they are important to conserve certain biodiversity features[168] and complement the role of the protected area system, which is often insufficient to protect biodiversity and which has little impact on habitat destruction and degradation in unprotected territory[169].

Landuse change (LUC) context: Two broad LUC contexts – ‘avoided deforestation’ and ‘restoration’ – are relevant for priority setting of forest conservation instruments and for the potential occurrence of trade-offs between conservation and ecosystem service provision. In contrast to the small scale, fine grain agricultural mosaics described above, landscapes under intensive agricultural production harbor low biodiversity, and the ecosystem service provided is almost exclusively food production. Restoration to recover other ecosystem services such as erosion control, sediment and eutrophication control of water courses, soil formation, habitat and food resources for fauna, is largely impaired by the high levels of forest fragmentation[170]. This can reach thresholds beyond which biodiversity structures (e.g. forest tree species) and functions (e.g. climate amelioration for seedling establishment, seed dispersal[171] are lost, or where recovery is extremely costly[172]. In these cases, economic instruments may only be effective in protecting biodiversity in remaining forest patches with low opportunity costs, but would not cover other conservation needs, including the restoration of important ecosystem services[173].

Public/private benefit types: Several examples presented in the previous sections are indicative that part of the spatial correspondence between biodiversity conservation and ecosystem services depends on whether ecosystem services provided result in private or public benefits. For the purposes of a simplified conceptual model, private forest benefits flow to the on-site landuser and are typically provisioning and cultural ecosystem services. Public benefits flow to all other off-site actors, and are typically cultural, regulating and supporting ecosystem services, including biodiversity. In the example of the Central Coast ecoregion of California, Chan and colleagues[174] found that excluding the two agriculture-focused services - crop pollination and forage production – eliminated all negative correlations among ecosystem services and biodiversity in their analysis. Also the cases in São Paolo State[175] and in Telemark County, Norway[176] indicate strong trade-offs between provisioning services of private benefits versus public benefits.

What lessons can be learned from this rapid review of biodiversity and ecosystem function linkages regarding conservation instrument design? First of all findings from biodiversity and ecosystem function research apply to spatial targeting of both coercive liability-based instruments as well as voluntary incentive based instruments. Decisions about whether to direct regulation instruments to non-use areas (protected areas) or economic instruments to partial use areas (multi-functional landscapes) involves trade-offs about ecosystem services and biodiversity conservation priorities. Multiple conservation objectives are needed both because of the difficulty of managing complex ecological systems and because the anthropogenic factors that drive biodiversity loss are many. Agricultural mosaic landscapes, although they cannot maintain all biodiversity, are important to conserve certain biodiversity features and complement the role of protected areas, which are often insufficient to protect biodiversity and which have little impact on habitat destruction and degradation in unprotected territory. In landscape mosaics, 'policyscapes' that deal with the various spatial factors of biodiversity loss will consist of mixes of instruments that are spatially targeted and complementary across landscapes. A potential weakness of voluntary conservation instruments (with or without monetary incentives) implemented as the only instrument, and particularly when land-tenure is dominated by small-holdings, is the small scale and fragmented implementation. (Economic) instruments that enable tracking of impacts, learning and adaptation at the various levels of decision-making are likely to be more robust and resilient than instruments than do not facilitate these processes. Pilot testing of voluntary conservation instruments and monitoring of effects needs to take place over more than a few properties, on participating and neighboring properties, and over longer time periods in order to understand landscape scale effects. Scale effects are also specific to landscape mosaics and bioregions, so predictions of (economic) instrument effectiveness based on experiences from other areas, even within the same country, are not necessarily transferable.

5.4 Unintended landscape level effects of landusers' adaptations to landuse and conservation incentives

In this section we review 'unintended' impacts on biodiversity that occur through landusers' rational economic adaptation to the introduction of economic incentives for conservation. We start with the unintended effect called 'adverse self-selection'. We then discuss 'slippage', also known as 'leakage' or 'negative spillovers' in voluntary conservation programmes, focusing particularly on results from studies of PES[177]. A non-systematic literature search suggests that slippage in the context of voluntary forest conservation and PES has received attention in a number of papers during the last decade. While there has been some model simulation work, we were not able to find any empirical studies of slippage with regard to biodiversity offsets[xvii]. Examples are drawn from Norway, Canada, USA, Costa Rica, Brazil and Mexico.

5.4.1 'Adverse' self- selection

'Adverse self-selection' happens because participation as a 'seller' is voluntary and landowners can enroll the least profitable land for conservation in exchange for monetary compensation[178]. This is perhaps the least unexpected of impacts of voluntary conservation incentives because it recognizes the same behavior in landowners that economists have been encouraging authorities to adopt in conservation planning[179]. For example, early years of PES in Costa Rica attracted landholders with the lowest opportunity costs, with subsequent PES cohorts both in protection and regeneration increasingly found on agriculturally productive land[180]. Enrolling land facing higher threat could raise payments' impact on deforestation[181]. In some landscapes it has been shown that effectiveness of targeting of biodiversity can be improved while lowering costs[182]. However,

[xvii] We used reference lists conducted a search on Science Direct and Google for the terms (Payment for ecosystem service OR biodiversity offsets) AND (slippage OR spill-over OR leakage OR indirect effects).

increasing payments lowers the total area that can be enrolled on a fixed conservation budget typical of many developing countries[183].

Diminishing marginal conservation target achievement to increasing conservation expenditure is a general characteristic of a 'conservation-production possibility frontier' observed across boreal and tropical forest landscapes[184]. Self-selection to cheap land is 'adverse' in as much as authorities do not expect that biodiversity value and agricultural landuse capacity are often positively correlated, meaning there are few 'win-win' opportunities in the landscape mosaic that has experience a forest transition (see section 4.3 for further discussion). Outcomes may be unexpected because authorities fail to recognize that landusers will enroll land from the opposite end of the conservation-production possibility frontier to what planners would like. Conservation auctions recognise this asymmetry - it is neither adverse nor unexpected - and use the auction procedure to attract offers of high conservation value land at the lowest possible non-coerced cost to authorities[185].

5.4.2 Slippage, leakage, spatial spillovers

'Slippage', 'leakage' and 'spillover effects' refer to impacts through market price effects on the landuse off-site from where conservation incentives are paid. Robalino and Pfaff[186] discuss the concepts of 'strategic substitutability in conservation / clearing' and 'strategic complementarity in conservation/ clearing' (Table 3). Complementarity and substitutability may occur together in a given landscape reinforcing or cancelling one another out. Robalino and Pfaff find evidence in Costa Rica of lower strategic substitutability closer to the capital, towns and main roads - increased production of agricultural goods near large markets is not likely to lower prices enough to reduce deforestation rates. They find evidence of higher strategic complementarity close to national parks, suggesting the role that protected area tourism may play in reinforcing conservation in buffer areas (without additional need for PES).

Table 3. Spillover effects – substitutability and complementarity in forest clearing and conservation

Examples	**Substitution**	**Complementarity**
Forest clearing	Local prices may fall if others deforest for agricultural production, reducing the incentives for further clearing . **(strategic substitutability in clearing).**	Local profitability of deforestation may rise if others who clear for production arrange transport to market that features economies of scale in transport for other producers **(strategic complementarity in clearing).**
Forest conservation	If enough others landusers have conserved and have created a tourism destination, this may raise the returns to forest clearing to create a new hotel **(strategic substitutability in conservation).**	On the other hand, farmers have been observed to band together to maintain contiguous blocks of forest for tourism, where the commitment to forest by one raises the returns from maintaining forest for others **(strategic complementarity in conservation).**

Source: Based on Robalino and Pfaff (2013:427-428)

Alix-Garcia and colleagues[187] evaluate two forms of 'slippage' in Mexico's PSAH programme which are related to Robalino and Pfaff[188] strategic substitutability'. 'Substitution slippage' occurs where conservation payments to a specific plot increase deforestation on other property belonging to the same program recipient. When a farmer enrolls some productive land into conservation, the marginal profit of cropped land increases relative to non-cropped land, without price effects necessarily playing a role[189]. 'Scarcity slippage' occurs where conservation payments with high levels of program participation increase deforestation due to increased scarcity of land, of agricultural production resulting in higher crop prices in local markets. Increased agricultural rents can also have an indirect effect on the poor by decreasing real wages where agricultural markets are local[190] (income effects of economic instruments are discussed in greater detail in Chapter 4). Alix-García and colleagues find evidence of substitution slippage in poor municipalities, suggesting a role for credit constraints (see Chapter 4).

Alix-García and colleagues also find indications of scarcity slippage with more land enrolled in an area leading to higher deforestation in buffers zones, conditional on the density of the road network. However, the results are not conclusive as high enrolment in an area may be correlated with deforestation on unobserved spatially correlated variables. Slippage has also been observed in the US conservation reserve programme, although only 'substitution slippage' can be clearly identified[191]. Alix-Garrcia and colleagues conclude that 'scarcity slippage' is more likely to occur in countries where rigidities in credit, labour and land markets are more pronounced (see also Chapter 4 for a discussion of distributional impacts under such conditions).

5.5 Unintended effects of economic instrument interactions in policy mixes

5.5.1 Policy mix interactions as a cause of unexpected effects

This section discusses how the effectiveness of economic instruments (for conservation and sustainable use) may be co-determined by a mix of other sector policy instruments. It provides some tools for detailed assessment of 'governance and capacity needs'[192]. We draw on empirical research mainly from the POLICYMIX project[xviii] , based on case studies from Norway, Finland, Portugal, Germany, Costa Rica and São Paulo and Mato Grosso States, Brazil[193]. In Appendix 3.6 we discuss a number of conceptual frameworks that could be used to identify unexpected policy mix effects. Indirect policy mix effects may compound the four types of unexpected effects discussed above. They occur both at property and landscape scale. An important notice at the outset is that policy instrument interactions are not unique to economic instruments.

Ring and Schröter-Schlaack[194] define a policy mix as "a combination of policy instruments which has evolved to influence the quantity and quality of biodiversity conservation and ecosystem service provision in public and private sectors." The OECD[195] mentions overlap of instruments in a policy mix as a potential source of policy inefficiency. Ring and Schröter-Schlaack[196] argue that the coincidence of instruments for biodiversity conservation may have a number of different functional roles under different circumstances, including conflictting, redundant, sequencing dependency, but also complementary and synergistic effects. Andrés and colleagues[197] identify a gap in research on unintended, unwanted and avoidable indirect effects of biodiversity policies, what they call five types of "rebound effects". These effects include (i) spatial spillovers also referred to a displacement or leakage, (ii) incongruence between protection of different types of biodiversity, (iii) unintended ecosystem function responses, (iv) trade-offs between biodiversity and ecosystem services, and (v) shifting or cascading of negative impacts from biodiversity to other environmental impacts. It is notable that of all the examples Andrés and colleagues used to

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[xviii] http://policymix.nina.no/

illustrate the five rebound effects almost all are examples of coercive, non-voluntary, regulatory instruments such as public protected areas, marine reserves, biological corridor creation, forest concession allocation, harvest extraction limits, and red list species conservation schemes. The only exception cited is the US conservation reserve program, where farmers are compensated for retired land from agricultural production, but where 20% cropland area retired is replaced elsewhere known as 'slippage'[198]. The authors identify a need for research on "rebound effects" of economic instruments in particular. But a question remains whether we should expect (based on theory) particular rebound effects for economic instruments.

To answer this question a vocabulary for economic instruments' interactions with the broader landuse policy mix is useful. Ring and Barton[199] discuss a number of different 'geometries of interaction' between and functional roles of instruments (see Appendix 3.6.1). Instrument interactions of the kind discussed are not particular for economic instruments, but they are often 'unexpected' because effects are indirect. For example, agricultural land scarcity can be increased by combinations of public protected areas and voluntary conservation payments in buffer zones acting together, reinforcing the 'scarcity slippage' effects discussed in section 5.4 (an example of 'interaction through socio-ecological systems' discussed in Appendix 3.6.1).

5.5.2 Instrument interactions at landscape scale

The analysis of interactions between economic instruments and other landuse policies is multi-dimensional. Several studies in the POLICYMIX project carried out qualitative two-way evaluations of single economic instruments (PES, tradable development rights, ecological fiscal transfers) with single conservation policy instruments[200]. A general limitation for quantitative studies of instrument interactions using before-after-control-impact evaluation methodologies is the need for large enough samples of participating properties that are exposed to the different two-way combinations of instruments[201]. To identify the incremental effect of PES, each participant must be matched with a group of non-participant farms with similar biophysical land use capacity, market access characteristics and proximities to other conservation instruments in force (Appendix 3.6.2). The PES programs in questions also needs to have been in place for a sufficient number of years to have observable effects on land cover. Few economic instruments for voluntary conservation in place today meet the requirements for conducting quantitative evaluation of instrument impact interactions.

In their study of PES-National Park interactions in Costa Rica, Robalino and colleagues'[202] preliminary results show that parks and 'protection PES' are perfect policy substitutes in the period 2000-2005 – properties receiving PES within national park boundaries have no incremental effect on avoided deforestation. The estimated effect of payments when those are implemented outside any protection is around 2.5% over the 5 year period (i.e. additional avoided deforestation of 0.5 parcels per year for every 100 enrolled parcels compared to the baseline deforestation rate on 100 matched non-participant parcels). However, the effect of implementing payments inside a buffer zone around national parks decreases to 1.4%. This implies that proximity to national parks reduces the effects of PES payments by 1.1%. Buffer areas without payments reduced deforestation by around 1.2%. When the sample of PES participants is too small or recent for quantitative impact evaluation, GIS modeling can be used to evaluate the extent to which voluntary conservation and protected areas target the same landuse characteristics, providing indications of possible policy interactions or policy 'gaps' in the landscape. For example, Barton and colleagues[203] found that voluntary forest conservation in Norway was poorly represented in high forestry value, high nature index forests[204], whereas these areas were better represented by public protected areas.

5.5.3 Instrument interactions at property scale

The framework for characterizing instrument interactions proposed by Ring and Barton[205] is generic. While GIS based 'policyscape analysis' used by Barton and colleagues[206] identifies the landuse characteristics where instruments overlap, it does not address the 'mechanisms' at property level which determine how PES effectiveness is strengthened or weakened by other instruments. Pagiola and colleagues[207] review the literature on determinants of PES participation and propose a number of PES program and household characteristics that determine eligibility, desire and ability to participate. Their framework illustrates from a conceptual point of view the high context specificity of PES participation. Porras and colleagues[208] review studies of PES in Costa Rica during the last 20 years and show the large diversity of landscape, farm and farmer characteristics that co-determine participation and effectiveness. Variable signs and significance vary within the same country, depending on the region. In Appendix 3.6.3 we broaden this analysis by providing examples of how the effectiveness of other policy instruments also co-determine PES effectiveness; such as PES eligibility being determined by cadastral mapping of tenure security; motivations to participate being determined by forest landuse regulations lowering opportunity costs of conservation; or the ability to participate being determined by access to credit.

The framework in Appendix 3.6.3 suggests at what point in a sequence of steps other instruments may co-determine the decision to participate in PES by affecting household characteristics. Another possible explanation for interaction may be found in the particular rules that make up PES and the other instruments in question (Appendix 3.6.4). Barton and colleagues[209] conduct a comparative analysis of the rules that make up voluntary forest conservation instruments in Norway, Finland and Costa Rica. Inspired by Ostrom[210] they argue that a source of interaction is the possible conflicting interpretations by the landowner, intermediaries and different institutions of specific PES rules. These may 'interfere' with similar rules of other instruments applied coincidentally to the same property. Examples include redundancy or conflict in forest management requirements between the PES contracts and broader forest regulations, property title requirements for PES eligibility which are not fulfilled by land titling institutions, credit requirements of PES which are not matched by rural credit institutions.

5.5.4 Instrument interactions in the policy cycle – path dependency

In this final section we briefly turn to unexpected instrument interactions over longer time scales of the policy cycle[211]. Appendix 3.6.4 suggests that there may be longer term conflicts between instruments at different time steps in the policy cycle. These conflicts may be unexpected at any particular point in time due to the uncertainty regarding external demand shocks and contractions on agricultural landuse that characterize forest transitions and long policy cycles. Experimental and adaptive policy design - for example using time limited voluntary conservation contracts[212] – can be important in dealing with longer term uncertainty in the implementation of economic instruments for conservation and sustainable use. Barton and colleagues[213] use an agent-based model to simulate how PES spatial targeting depends on previous targeting of public protected areas and PES contract lengths across multiple stages of a forest transition.

In a policy cycle, there are trade-offs between budget decisions and conservation policy objectives, which may or may not be foreseen. For example, the future mix of policies must balance resource use and policy outputs across a multiple year policy cycle. Börner and colleagues[214] evaluated the trade-offs across the Amazon between the cost-effectiveness command-and-control versus PES incentives using a GIS model of opportunity costs to forestry of REDD, cost of inspection, enforcement likelihood and expected fines. Command-and-control was the most cost-effective approach from the regulators point of view, but with high social opportunity cost. Future PES schemes are expected to reduce social costs by compensating for conservation, but are expected to increase budgetary outlays and possibly decrease REDD effectiveness.

5.6 Conclusions on unintended effects

What lessons can be learned regarding design of economic instruments from our review of unexpected impacts of economic instruments? In this chapter we have discussed the possibility that ecosystem function and biodiversity inter-linkages, motivational crowding, social networks and copying behavior, slippage or leakage, and policy interaction effects may be the source of unintended positive and negative effects of economic instruments for conservation and sustainable landuse. As such we have provided a checklist of issues that may affect policy assessment[215] regarding e.g. least-cost policy, safeguards and governance and capacity needs.

Unintended effects at property and landscape level mean that there are fewer *'win-win'* conservation opportunities than some market-based instrument literature would suggest. Diminishing marginal conservation target achievement with increasing conservation expenditure is a general characteristic of a 'conservation-production possibility frontier' observed across both boreal and tropical forest landscapes. This is particularly the case when high biodiversity conservation value is accompanied with high opportunity costs to agriculture. Furthermore, self-selection by landowners of conservation 'efforts' on cheap land is adverse for conservation when forest biodiversity value and agricultural landuse capacity are positively correlated. 'Scarcity *slippage'* – increased local agricultural prices and incentives for forest conversion due to scarcity of agricultural land - is more likely to occur in countries where rigidities in credit, labour and land markets are more pronounced. However, slippage is not unique to economic instruments, but is relevant wherever conservation takes land out of agricultural production for a local market. *Motivational crowding* effects apply both to coercive regulation-based instruments as well as economic incentives. Motivational crowding-out effects seem to be more diverse than crowding-in effects. However, the empirical evidence for crowding-out is still scant. Technical extension, adaptation and learning opportunities are motivations for voluntary participation by themselves, but are also likely to increase the effectiveness of PES. In contexts with strong community organization, *social network effects* are likely. In addition, if landuse management decisions involve complex trade-offs between different interests, and the outcomes or landuse change are uncertain, copying behavior may be more likely than private net benefit calculation suggests. In such situations economic incentives are less likely to have their intended effect.

Findings from *biodiversity and ecosystem function* research apply to spatial targeting of both coercive liability-based instruments, as well as voluntary incentive based instruments. Decisions about whether to direct regulation instruments to non-use areas (protected areas) or economic instruments to partial use areas (multi-functional landscapes) involves trade-offs about ecosystem services and biodiversity conservation priorities. Multiple conservation objectives are needed both because of the difficulty of managing complex ecological systems and because the anthropogenic factors that drive biodiversity loss are many. Economic – and other - instruments that enable tracking of impacts, learning and adaptation at the various levels of decision-making are likely to be more robust and resilient than instruments than do not facilitate these processes. Pilot testing of voluntary conservation instruments and monitoring of effects needs to take place over more than a few properties, on participating and neighboring properties, and over longer time periods in order to understand landscape scale effects. *Scale effects* are also specific to landscape mosaics and bioregions, so predictions of (economic) instrument effectiveness based on experiences from other areas, even within the same country, are not necessarily transferable.

Although agricultural mosaic landscapes do not maintain all biodiversity, they are important to conserve certain biodiversity features and complement the role of the protected area system, which are often insufficient to protect biodiversity and which have little impact on habitat destruction and degradation in unprotected territory. A potential weakness of voluntary conservation instruments as the only instrument in a landscape mosaic is small scale and fragmented implementation. '*Policyscapes*' consisting of mixes of instruments that are spatially targeted and complementary across fragmented landscapes mosaics are designed to deal with the various spatial factors of biodiversity loss. In principle[216] there is a complementary role for payment for ecosystem services in a landscape mosaic when net private benefits of habitat restoration or conservation are negative, but exceeded by public net benefits. However, unintended indirect effects make such calculations of public net benefits difficult to carry out. Unexpected effects at property level also suggests that calculation of private net benefits – even by the landuser themselves – is a complex task. Unintended effects of PES highlight the need for some redundancy in the *policy mix*, as well as information instruments, and adaptive flexibility that allow learning and make landuse prospects clearer.

Finally, policy instrument interactions are not unique to economic instruments. A better understanding of policy mix interactions should not only be useful in eliminating conflicting or unnecessarily redundant instruments, but also in identifying instrument synergies or situations where policy instrument redundancy is important for conservation risk management.

6. Conclusion

Payments for nature values are observed to take a variety of forms. The largest volumes are found in what we have termed liability based systems – e.g., cap-and-trade systems – especially those for carbon. It, is, however notable, that in these cases, the protection of nature lies in the cap. The market operates to reduce costs for abiding by the cap.[xix] In the case of non-liability based systems – i.e., systems where 'buyers' or 'users' take on the responsibility for environ-mental protection – public agents are strongly dominating as 'buyers' and only a very minor part takes the form of (complete) market trades.

Prêt-a-manger versus locally adapted policy mix recipes?
Illustration Javier Sáez
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There is an increased interest in expanding the role of private actors and markets in conservation and sustainable use of ES. In relation to that, the issue of financialization has been emphasized, referring to the danger that 'financial logics' may become dominating and be harmful for the aims set for ES delivery. Typical forms of financialization are derivatives and securitization. Regarding the former we conclude that they will not bring any new resources to ES delivery. They are instruments developed to hedge against risks in financial transactions. They can also hedge against natural risks – e.g., effects of climate change and biodiversity loss. They may handle variations in e.g., weather patterns, but their capacity as a measure against negative trends is weak. It is notable that financial instruments like derivatives lend themselves to speculation and that they may cause risk as much as they may hedge against it.

Forest bonds are an example of financial products where new resources may be made available from the private sector – e.g., reforestation or afforestation projects. They demand returns on the investments, and that can only be ensured through emphasizing private goods components – e.g., timber – or making states responsible for paying for the public goods components. We generally conclude that with the motivations presently characterizing the private sector, the prospects for raising large amounts of private resources for public goods components of ES trough new financial instruments is very weak. Rather there is a danger for 'state capture' – i.e., turning the role of states/public agents into that of securing private profits from ES protection. We note that the main interest in 'nature derivatives' and 'green bonds' come from a financial sector that are looking from new income opportunities and to earn income form the trading itself (TCs). We also note that while hedging may be important, this need is an effect of using the market. To the

[xix] While not discussed in this report, we note that to the extent trading in a cap-and-trade system reduces overall costs, resources are freed up that could be directed towards further protection. Ensuring that to happen would imply public action to direct the gains towards such an aim, though. We also note that cap-and-trade systems for carbon and systems linked to these like the CDM could be used to ensure biodiversity protection through defining cross-compliance responsibilities.

extent that resources in the end have anyway to come from the public purse, one can avoid costly hedging and speculation through using fund-based approaches.

One objective of this report was to document socio-economic impacts of different economic instruments. The analyses were limited to PES and CDM_{AR}. Regarding distributional effects, we note that payments mainly go to larger land-owners. This is explained mainly through high transaction costs for low volume trades and the fact that small-holders often need all the land for sustaining themselves. Payments are repeatedly observed to be lower than opportunity costs. Hence, PES and CDM_{AR} has to a minor extent been able to ensure a win-win situation – both reducing deforestation and poverty. Focusing more at the community level as opposed to individual land-owners and including intermediaries that are motivated towards supporting community development, seems to be effective means to include the poor, while safeguards are important to avoid elite capture. It will also help to reduce per unit TCs that are observed to be very dependent on the volume of delivered ES. Regarding non-liability systems, it is observed that public bodies have the capacity to reduce TCs substantially as compared to private actors. In the case of liability based systems, the situation is different. These are dominated by private actors. It is unclear, though, to what extent transaction costs play a decisive role in that respect.

In the last part of the report, we discuss potential sources of unintended effects on biodiversity conservation of economic instruments through ecosystem function and biodiversity inter-linkages, motivational crowding, social networks, slippage, and policy interactions. Unintended effects at property and landscape level mean that there may be fewer 'win-win' conservation opportunities than some market-based instrument literature would suggest. Land-owners self-select conservation 'efforts' on cheap land – this is adverse for conservation where natural habitat biodiversity and agricultural land use capacity are positively correlated. 'Scarcity slippage' in the form of increased local agricultural prices and incentives for forest conversion due to scarcity of agricultural land is more likely to occur in countries where rigidities in credit, labor and land markets are more pronounced. Voluntary conservation motivations may be crowded out by economic incentives for conservation, but may not be recovered once incentives stop. In contexts with strong community organization, social network effects are likely, with copying behavior substituting net benefit rationales where land use decisions are complex and uncertain. Other instruments in a policy mix may interact unexpectedly with economic instruments. Almost by definition, empirical evidence for these unintended effects is scant - effects foreseen are more likely to be monitored. We present a number of cases where unintended effects have been observed, but they are often local and context specific.

This report emphasizes the role of actor's motivations and forms of interaction for the effect of policy instruments on ES delivery, costs and distribution. While we note that many important mechanisms are context dependent, a quite consistent overall picture has materialized. Markets may be helpful to reduce costs of publicly established caps, but they are less helpful the more complex the ES is and the more demanding it is to ensure additionality. Moreover, it is difficult to raise funding from private actors for biodiversity/ES without using the public power to command – using e.g., taxes. Hence, in cases where it is difficult to 'rule via caps', there is much less to gain from markets. Overall, the importance of public engagement stands out as crucial. One should not be surprised as we talk largely about public goods.

Endnotes

Summary

[i] Vatn et al. (2011)

Chapter 1

[2] COP 10 (2010)
[3] Vatn et al. (2011)

Chapter 2

[4] see Vatn (2011);
[5] e.g., Cooter (2000)
[6] e.g., Schotter (1994)
[7] Perrot-Maître (2006)
[8] Milder et al. (2010)
[9] World Bank (2011)
[10] According to Nathaniel and Jenkins (2012)
[11] ibid.
[12] World Bank (2012)
[13] see Hahn et al. 2014)
[14] Madsen et al. (2010)

Chapter 3

[15] see also Hahn et al. (2014); Farooqui and Schultz (2012); Sullivan (2013)
[16] Kolb and Overdahl (2010)
[17] Pryke and Allen (2000); Giron and Chapoy (2013)
[18] see also Arnoldi (2004) and Sullivan (2013) on these issues
[19] e.g., McNally and Levi (2011)
[20] Tickell (2000); Hellwig (2009); Büscher (2010)
[21] e.g., TEEB (2010), Gómez-Baggethun et al. (2010a); Spash (2011)
[22] TEEB (2010)
[23] see e.g., Vatn and Bromley (1994); O'Neill et al. (2008)
[24] e.g., Gómez-Baggethun et al. (2010a)
[25] Hess et al. (2002)
[26] Mandel et al. (2010:45)
[27] Mandel et al. (2010a); Cranford et al. (2011)
[28] Cranford et al. (2011)
[29] Sullivan (2013)
[30] Shin (2009)
[31] Allen and Pryke (2013)
[32] see also Graeber (2001) and Sullivan (2013)) on this
[33] e.g., Strange (1998); McNally (2011)
[34] Daskalakis et al. (2012)
[35] Cormier et al. (2013)
[36] Schneider (2010)
[37] see Lohmann (2010)
[38] see Chevallier (2011)
[39] Schneider (2010)
[40] Hill et al. (2008)
[41] Mizrach (2012)
[42] Dalaskias et al. (2012).
[43] Daskalakis et al. (2009)
[44] Mizrach (2012)
[45] Chester and Rosewarne (2011)
[46] Mansanet-Batallera et al. (2011)
[47] ibid.; Colla et al. (2012); Feng et al. (2012)
[48] Chevallier (2010); (2012)
[49] Mol (2012)
[50] Andrew (2008)
[51] Taschinia et al. (2013)
[52] ibid.
[53] Chester and Rosewarne (2011)
[54] Mol (2012), see also Knox-Hayes (2013)
[55] Chester and Rosewarne (2011:23), see also Mol (2012)
[56] Chevallier (2013)
[57] see Burniaux et al. (2009)
[58] see e.g. Hultaman et al. (2012)
[59] Das (2011)
[60] ibid.
[61] IETA (2013)
[62] see Burniaux et al. (2009); Cacho et al. (2013)
[63] Mandel et al. (2010:44)
[64] ibid. pp. 46
[65] Sullivan (2013:208)
[66] London Accord (2009)
[67] Tickell (2000)

Chapter 4

[68] e.g., Porras et al. (2008); Kosoy et al. (2007); Wunder et al. (2008); Jindal et al. (2008); McAffe and Shapiro (2010); Bosselmann and Lund (2013); Krause and Loft (2013).
[69] e.g., Porras et al. (2008).
[70] see Corbrera et al. (2007a); Alix-Garcia et al. (2012); Ingram et al. (2014)
[71] Porras et al. (2013)
[72] see also Unruh (2008)
[73] see Grieg-Gran and Bishop (2004); Corbera et al. (2007b); Westermann (2007); Wunder and Alban (2008) and Wunder et al. (2008)
[74] e.g., Michaelowa and Jotzo (2005); Jindal et al. (2008), Bosselmann and Lund (2013)
[75] Mahanty et al. (2013)

[76] Bosselmann and Lund (2013); Mahanty et al. (2013)
[77] Alex-Garcia et al. (2014)
[78] e.g., Porras et al. (2008) and Mahanty et al. (2013)
[79] e.g., Corbera et al. (2007a)
[80] Viana (2008)
[81] Porras et al. (2013)
[82] see e.g., Sullivan (2010); Nakakaawa et al (2011); Mahanty et al. (2013)
[83] Jindal et al. (2008)
[84] e.g., Jindal et al. (2008)
[85] e.g., Eraker (2000); Cossalter and Pye-Smith (2003); Lang and Byakola (2006); Benjaminsen et al. (2008); Jindal et al. (2008)
[86] Bosselmann and Lund (2013)
[87] McAffe and Shapiro (2010)
[88] Pagiola and Platais (2007)
[89] Corbera et al (2007a)
[90] Krause and Loft (2013)
[91] e.g., Porras et al. (2008; 2013) and Wertz-Kanounnikoff et al. (2008)
[92] e.g., Wunder (2007); Kosoy et al. (2008); Corbera et al. (2009)
[93] e.g., Porras et al. (2008); Kosoy et al. (2008); Gross-Camp et al. (2012); Mahanty (2013)
[94] Wunder (2007) and Wunder and Alban (2008)
[95] see also Corbera et al. (2007b); Porras et al. (2008); Muradian et al. (2008)
[96] Ensminger (1996); Sjaastad and Cousins (2008)
[97] e.g., Frank (1987); Pattanayak et al. (2010)
[98] Fisher (2012:50)
[99] see Porras et al. (2008); Wunder et al. (2008); Jindal et al. (2008)
[100] see Porras et al. (2008; 2013); Wunder et al. (2008)
[101] ibid.
[102] e.g., Gross-Camp et al. (2012); Vatn et al. (2013)
[103] e.g., Vatn et al. (2013); Nabanoga et al. (2013)
[104] Vatn et al. (2013)
[105] e.g., Jindal et al (2008)
[106] e.g., Porras et al. (2008); Vatn et al. (2013); Hiedanpää and Bromley (2014)
[107] Porras (2010); Porras et al. (2013I; Porras et al. (2014)
[108] see Porras et al. (2012)
[109] Porras et al. (2013)
[110] ibid.
[111] ibid.
[112] Miranda et al. 2003; Zbinden and Lee (2005)
[113] Porras et al. (2014)
[114] Wunder et al. (2008)
[115] Michaelowa and Jotzo (2005)
[116] Rørstad et al. (2007)
[117] Vatn (1998)
[118] Michaelowa and Jotzo (2005)
[119] Coggan et al. (2013)
[120] e.g., Corbera et al. (2007b); Mahanty et al. (2013); McAffe and Shapiro (2010)

Chapter 5

[121] Vatn et al. (2011)
[122] Vatn (2010)
[123] Vatn (2010)
[124] Cardenas et al. (2000); Garcia-Amado et al. (2013); Kerr et al. (2012)
[125] Kerr et al. (2012)
[126] Rodriguez-Sickert et al. (2008)
[127] Hiedanpää and Bromley (2014)
[128] Primmer et al. (2014)
[129] Hiedanpää and Bromley (2014)
[130] Zapata et al. (forthcoming)
[131] Kahneman and Tversky (2000)
[132] Mills et al. (2014); van der Horst (2011)
[133] Sutherland et al. (2012)
[134] van der Horst (2011)
[135] Arriagada et al. (2009)
[136] Ormerod (2012)
[137] Robalino and Pfaff (2012)
[138] Michel (2012)
[139] Daniels et al. (2010)
[140] Sierra and Russman (2006)
[141] Garbach et al. (2012)
[142] Mace et al. (2012); Maestre Andrés et al. (2012)
[143] Rusch et al. (2013)
[144] MA (2005); Rockström et al. (2009)
[145] Devictor et al. (2010)
[146] Ahrné et al. (2009); Ricketts et al. (2008); Steffan-Dewenter (2002)
[147] Maestre Andrés et al. (2012)
[148] DeClerck et al. (2010)
[149] Locatelli et al. (2014)
[150] Devictor et al. (2010); Zupan et al. (2014)
[151] Maestre Andrés et al. (2012)
[152] Barton et al. (2013a)
[153] Chan et al. (2006)
[154] Locatelli et al. (2014); Schröter et al. (forthcoming)
[155] Chan et al. (2006)
[156] Schröter et al. (forthcoming)
[157] Venter et al. (2009)
[158] Maestre Andrés et al. (2012)
[159] De Bello et al. (2010); Díaz et al. (2007); Lavorel et al. (2011); Lavorel et al. (2013); Mulder et al. (2013)
[160] Devictor et al. (2010); Zupan et al. (2014)
[161] DeClerck et al. (2010)
[162] Devictor et al. (2010); Zupan et al. (2014)
[163] Schröter et al. (forthcoming)
[164] Niemeyer et al. (2014)
[165] Casals et al. (2014)
[166] Tscharntke et al. (2011)
[167] Cassano et al. (2009); van Breugel et al. (2013)
[168] Daily et al. (2003); Harvey et al. (2006); Schroth and Harvey (2007)

[169] DeClerck et al. (2010)
[170] Herrera and Garcia (2010); Honnay et al. (2005)
[171] Van Breugel et al. (2013)
[172] Bernasconi (2013)
[173] Bernasconi (2013)
[174] Chan et al. (2006)
[175] Bernasconi (2013)
[176] Schröter et al. (forthcoming)
[177] Robalino and Pfaff (2012); Robalino (2007); Alix-Garcia et al. (2012); Wu (2000); Wu et al. (2001); Lichtenberg (2004); Lichtenberg and Smith-Ramirez (2011); Fraser and Waschik (2005); Murray et al. (2004); Chomitz (2002)
[178] Hiedanpää and Bromley (2014)
[179] Naidoo et al. (2006)
[180] Daniels et al. (2010); Robalino and Pfaff (2013)
[181] Robalino and Pfaff (2013)
[182] Barton et al. (2009)
[183] Porras et al. (2013)
[184] Barton et al. (2009); Hauer et al. (2009); Joppa and Pfaff (2009)
[185] Ferraro (2008)
[186] Robalino and Pfaff (2012)
[187] Alix-Garcia et al. (2012)
[188] Robalino and Pfaff (2012)
[189] Wu (2005)
[190] Robalino (2007); Uchida et al. (2009)
[191] Wu (2000); Wu (2005)
[192] OECD (2013)
[193] Barton et al. (2014e)
[194] Ring and Schröter-Schlaack (2011b)
[195] OECD (2007)
[196] Ring and Schröter-Schlaack (2011b)
[197] Andrés et al. (2012)
[198] Wu (2000)
[199] Ring and Barton (forthcoming)
[200] Barton et al. (2012); Primmer et al. (2013); May et al. (2012); Santos et al. (2012); Schröter-Schlaack et al. (2013); Chacón-Cascante et al. (2012); Romeiro et al. (2012)
[201] Robalino et al. (2014)
[202] Robalino et al. (2014)
[203] Barton et al. (2013)
[204] Nybø et al. (2012)
[205] Ring and Barton (forthcoming)
[206] Barton et al. (2013a)
[207] Pagiola et al. (2005)
[208] Porras et al. (2013)
[209] Barton et al. (2014b)
[210] Ostrom (2005)
[211] Brewer and DeLeon (1983); Kivimaa and Mickwitz (2006)
[212] Pagiola (2008)
[213] Barton et al. (2014a)
[214] Börner et al. (2011)
[215] OECD (2013)
[216] Pannell (2008)

References

Ahrné, K., J. Bengtsson and T. Elmqvist, 2009. Bumble bees (Bombus spp) along a gradient of increasing urbanization. Plos One 4.

Alix-Garcia, J.M., K.R.E Sims and P. Yañes-Pangans, 2014. Only One Tree from Each Seed? Environmental Effectiveness and Poverty Alleviation in Mexico's Payments for Ecosystem Services Program. Unpublished paper, Department of Agricultural and Applied Economics, University of Wisconsin, Madison, USA.

Alix-Garcia, J.M., E.N. Shapiro and K.R.E. Sims, 2012. Forest Conservation and Slippage: Evidence from Mexico's National Payments for Ecosystem Services Programs. Land Economics, 88:613–638.

Allen, J. and M. Pryke, 2013. Financialising household water: Thames Water, MEIF, and 'ring-fenced' politics. Cambridge Journal of Regions, Economy and Society, 6:419-439.

Andam, K.S., P.J. Ferraro, A. Pfaff, G.A. Sanchez-Azofeifa and J.A. Robalino, 2008. Measuring the effectiveness of protected area networks in reducing deforestation. P Natl Acad Sci USA, 105:16089-16094.

Andrés, S.M., L.C Mir, J.C.J.M. van den Bergh, I. Ring and P.H. Verburg, 2012. Ineffective biodiversity policy due to five rebound effects. Ecosystem Services, 1:101–110.

Andrew, B., 2008. Market failure, government failure and externalities in climate change mitigation: the case for a carbon tax. Public Administration and Development, 28(5):393-401.

Angelsen, A., 2007. Forest cover in space and time: combining von Thünen and the forest transition. World Bank Policy Research Working Paper 4117. World Bank, Washington DC.

Arnoldi, J., 2004. Derivatives. Virtual values and Real Risks. Theory. Culture & Society, 21(6):23-42

Arriagada, R.A., E. Sills, S.K. Pattanayak and P.J. Ferraro, 2009. Combining Qualitative and Quantitative Methods to Evaluate Participation in Costa Rica's Program of Payments for Environmental Services. Journal of Sustainable Forestry, 28:343–367.

BBOP, 2009. Biodiversity Offset Design Handbook. In: (BBOP), B.a.B.O.P. (Ed.), Washington D.C.

Barton, D.N. and W.L.V. Adamowicz, 2013. Path-dependent policyscapes: a theoretical approach to the evaluation of policymixes for biodiversity conservation, ESEE 2013 Conference: Ecological Economics and Institutional Dynamics. 10th biennal conference of the European Society for Ecological Economics, 18-21 Jun 2013, Lille (France).

Barton, D.N., S. Blumentrath and G. Rusch, 2013a. Policyscape. A Spatially Explicit Evaluation of Voluntary Conservation in a Policy Mix for Biodiversity Conservation in Norway. Society & Natural Resources, 26:1185-1201.

Barton, D.N., S. Blumentrath and G.M. Rusch, 2013b. Policyscape - a spatially explicit evaluation of voluntary conservation in a policymix for biodiversity conservation in Norway. Society and Natural Resources, 26:1185 - 1201.

Barton, D.N., D. Faith, G. Rusch, H. Acevedo, L. Paniagua and M. Castro, 2009. Environmental service payments: Evaluating biodiversity conservation trade-offs and cost efficiency in the Osa Conservation Area, Costa Rica. Journal of Environmental Management, 90:901-911.

Barton, D.N., H. Lindhjem, G.M. Rusch, A. Sverdrup-Thygeson, S. Blumentrath, M.D. Sørheim, H. Svarstad and V. Gundersen, 2012. Assessment of existing and proposed policy instruments for biodiversity conservation in Norway. POLICYMIX Report Issue No 1/2012.

Barton, D.N., C. Klassert and V. Adamowicz, 2014a. Path dependent policyscapes: an agent-based modeling approach to the evaluation of policymixes for biodiversity conservation. In Ring, I., D.N. Barton and G.M. Rusch (eds.), International Conference on Policy Mixes in Environmental and Conservation Policies. UFZ, 25–27 February 2014, Leipzig, Germany.

Barton, D.N., E. Primmer, A. Chacón-Cascante and D. Caixeta Andrade, 2014b. Cross case comparison of payments for ecosystem services in forest (PES). In: Santos, R., P. May, D.N. Barton and I. Ring (eds.), Comparative assessment of policy mixes across case studies - common design factors and transferability of assessment results. POLICYMIX Report, Issue No. 1/2014. Available at http://policymix.nina.no.

Barton, D.N., E. Primmer, A. Chacón-Cascante and D. Caixeta Andrade, 2014c. Cross case comparison of payments for ecosystem services in forest (PES). In Santos, R., P. May, D.N. Barton and I. Ring (eds.), Comparative assessment of policy mixes across case studies - common design factors and transferability of assessment results. POLICYMIX Report, Issue No. 1/2014.

Barton, D.N., I. Ring and G.M. Rusch, 2014d. From panaceas to policy mixes – an overview of the POLICYMIX project. In Barton, D.N., I. Ring and G.M. Rusch (eds.), International Conference on Policy Mixes in Environmental and Conservation Policies, 25–27 February 2014, Leipzig, Germany.

Barton, D.N., I. Ring, G.M. Rusch, R. Brouwer, M. Grieg-Gran, E. Primmer, P. May, R. Santos, H. Lindhjem,C. Schröter-Schlaack, N. Lienhoop, J. Similä, P. Antunes, D. Caixeta Andrade, A. Romerio, A. Chacón-Cascante, F. DeClerck, M. Tingstad and K. Sivertsen 2014e. Guidelines for multi-scale policy mix assessment. POLICYMIX Technical Brief no. 12.

Benjaminsen, T.A., I. Bryceson and F. Maganga, 2008. Climate change in Tanzania: Trends, policies and initiatives. Noragric working paper. Norwegian University of Life Sciences.

Bernasconi, P., 2013. Custo-efetividade ecológica da compensação de reserva legal entre propriedades no estado de São Paulo, Instituto de Economia. Universidade Estadual de Campinas, 113.

Börner, J., S. Wunder, S. Wertz-Kanounnikoff, G. Hyman and N. Nascimento, 2011. REDD sticks and carrots in the Brazilian Amazon. Assessing costs and livelihood implications. Working Paper No. 8, CGIAR Research Program on Climate Change, Agriculture and Food Security (CCAFS).

Bosselmann, A.S. and J.F. Lund, 2013. Do intermediary institutions promote inclusiveness in PES programs? The case of Costa Rica. Geoforum, 49:50-60.

Bowles S., 2008. Policies Designed for Self-Interested Citizens May Undermine "The Moral Sentiments": Evidence from Economic Experiments. Science, 320:1605-09.

Brewer, G.D. and P. DeLeon, 1983. The foundations of policy analysis. Dorsey Press Homewood, Ill.

Burniaux, J. M., J. Chateau, R. Dellink, R. Duval, S. Jamet, and O. En, 2009. The Economics of Climate Change Mitigation: How to Build the Necessary Global Action in a Cost-effective Manner. OECD Economics Department Working Papers No. 701.

Büsher, B., Derivative Nature: interrogating the value of conservation in 'Boundless Southern Africa'. Third World Quarterly, 31(2):259-256.

Cacho, O.J., L. Lipper and J. Moss 2013. Transaction costs of carbon offset projects: A comparative study. Ecological Economics, 88, 232-243.

Cardenas, J.C., J. Stranlund and C. Willis, 2000. Local environmental control and institutional crowding out. World Development 28(10):1719–1733.
Casals, P., M.J. Romero, G.M. Rusch and M. Ibrahim, 2014. Soil organic C and nutrient contents under trees with different functional characteristics in seasonally dry tropical silvopastures. Plant and Soil, 374(1-2):643-659.
Cassano, C.R., G. Schroth, D. Faria, J.H.C. Delabie and L. Bede, 2009. Landscape and farm scale management to enhance biodiversity conservation in the cocoa producing region of southern Bahia, Brazil. Biodiversity and Conservation, 18:577-603.
Chadwick, B. P., 2006. Transaction costs and the clean development mechanism. Natural Resources Forum, 30(4):256-271.
Chacón-Cascante, A., M. Ibrahim, Z. Ramos, F.A.J. DeClerck, R. Vignola and J. Robalino, 2012. Costa Rica: National level assessment of the role of economic instruments in the conservation policymix. Policymix Report Issue 2/2012. http://policymix.nina.no. CATIE.
Chan, K.M.A., M.R. Shaw, D.R. Cameron, E.C. Underwood and G.C. Daily, 2006. Conservation planning for ecosystem services. PLOS Biology, 4:2138-2152.
Chang, R. 1997. Incommensurability, Incomparability, and Practical Reason. Cambridge,
Chester, L. and S. Rosewarne, 2011. What Is the Relationship between Derivative Markets and Carbon Prices? Draft paper presented at the Conference "NatureTM Inc? Questioning the Market Panacea in Environmental Policy and Conservation," The Hague, The Netherlands, 30 June–2 July.
Chevallier, J., 2009. Carbon futures and macroeconomic risk factors: a view from the EU ETS. Energy Economics, 31(4):614-625.
Chevallier, J., 2011. Econometric analysis of carbon markets: the European Union emissions trading scheme and the clean development mechanism. Springer.
Chevallier, J., 2013. Variance risk-premia in CO_2 markets. Economic Modelling, 31:598-605.
Chomitz, K., 2002. Baseline, leakage and measurement issues: how do forestry and energy projects compare? Climate Policy, 2:35-49.
Coggan, A., E. Buitelaar, S. Whitten and J. Bennett, 2013. Factors that influence transaction costs in development offsets: Who bears what and why? Ecological Economics, 88:222-231.
Colla, P., M. Germain, and V. Van Steenberghe, 2012. Environmental policy and speculation on markets for emission permits. Economica, 79(313):152-182.
Cooter, R., 2000. Do good laws make good citizens? An economic analysis of internalized norms. Virginia Law Review, 86(8):1577-1602.
COP 10, 2010. Report to the tenth meeting of the conference of the parties to the convention on biological diversity. UNEP/CBD/COP/10/27, 19 December 2010.
Corbera, E., N. Kosoy and M. Martinez-Tuna, 2007a. Equity implications of marketing ecosystem services in protected areas and rural communities: Case studies from Meso-America. Global Environmental Change, 17:365-380.
Corbera, E., K. Brown and W.N. Adger, 2007b. The Equity and Legitimacy of Markets for Ecosystem Services. Development and Change, 38(4):587-613.
Corbera, E., C.G. Soberianis and K. Brown, 2009. Institutional dimensions of Payments for Ecosystem Services: An analysis of Mexico's carbon forestry programme. Ecological Economics, 68:743-761.
Cormier, A., and V. Bellassen, 2013. The risks of CDM projects: How did only 30% of expected credits come through? Energy Policy, 54:173-183.

Cornwell, W. K., J.H.C. Cornelissen, K. Amatangelo, E. Dorrepaal, V.T. Eviner, O. Godoy, S.E. Hobbie, B. Hoorens, H. Kurokawa, N. Pérez-Harguindeguy, H.M. Quested, L.S. Santiago, D.A. Wardle, I.J. Wright, R. Aerts, S.D. Allison, P. van Bodegom, V. Brovkin, A. Chatain, T.V. Callaghan, S. Díaz, E. Garnier, D.E. Gurvich, E. Kazakou, J.A.l. Klein, J. Read , P. Reich, N.A. Soudzilovskaia, M.V. Vaieretti and M. Westoby, 2008. Plant species traits are the predominant control on litter decomposition rates within biomes worldwide. Ecology Letters, 11:1065-1071.

Cossalter, C. and C. Pye-Smith, 2003. Fast-Wood Forestry. Myths and Realities. Cifor, Bogor, Indonesia.

Cranford, M., I.R. Henderson, A.W. Mitchell, S. Kidney, and D. Kanak, 2011. Unlocking Forest Bonds: A High-Level Workshop on Innovative Finance for Tropical Forests. Workshop Report. WWF Forest & Climate Initiative, Global Canopy Programme and Climate Bonds Initiative

Daily, G.C., G. Ceballos, J. Pacheco, G. Suzan and A. Sanchez-Azofeifa, 2003. Countryside biogeography of neotropical mammals: Conservation opportunities in agricultural landscapes of Costa Rica. Conservation Biology, 17:1814-1826.

Daniels, A.E., K. Bagstad, V. Esposito, A. Moulaert and C.M. Rodriguez, 2010. Understanding the impacts of Costa Rica's PES: Are we asking the right questions. Ecological Economics.

Das, K., 2011. Technology Transfer under the Clean Development Mechanism: an empirical study of 1000 CDM projects. Working Paper 014, The Governance of Clean Development Working Paper Series. School of International Development, University of East Anglia, UK

Daskalakis, G., D. Psychoyios and R. N. Markellos, 2009. Modeling CO_2 emission allowance prices and derivatives: Evidence from the European trading scheme. Journal of Banking and Finance, 33:1230-1241.

Daskalakis, G., G. Ibikunle, and I. Diaz-Rainey, 2011. The CO_2 Trading Market in Europe: A Financial Perspective. In Dorsman, A., W. Westerman, M.B. Karan and Ö. Arslan (eds.) Financial Aspects in Energy. Berlin Heidelberg, Springer, 51-67.

Daily, G., (ed.), 1997. Nature's Services. Social Dependence on Natural Ecosystems. Washington DC, Island Press.

de Bello, F., S. Lavorel, S. Diaz, R. Harrington, J.H.C. Cornelissen, R.D. Bardgett, M.P. Berg, P. Cipriotti, C.K. Feld, D. Hering, P.M. da Silva, S.G. Potts, L. Sandin, J.P. Sousa, J. Storkey, D.A. Wardle and P.A. Harrison, 2010. Towards an assessment of multiple ecosystem processes and services via functional traits. Biodiversity and Conservation, 19:2873-2893.

DeClerck, F.A.J., R. Chazdon, K.D. Holl, J.C. Milder, B. Finegan, A. Martinez-Salinas, P. Imbach, L. Canet and Z. Ramos, 2010. Biodiversity conservation in human-modified landscapes of Mesoamerica: Past, present and future. Biological Conservation, 143:2301-2313.

Devictor, V., D. Mouillot, C. Meynard, F. Jiguet, W. Thuiller and N. Mouquet, 2010. Spatial mismatch and congruence between taxonomic, phylogenetic and functional diversity: the need for integrative conservation strategies in a changing world. Ecology Letters, 13:1030-1040.

Diaz, S., S. Lavorel, F. de Bello, F. Quétier, K. Grigulis and T.M. Robson, 2007. Incorporating plant functional diversity effects in ecosystem service assessments. PNAS, 104:20684-20689.

Dryzek J.S. 2000. Deliberative Democracy and Beyond: Liberals, Critics, Contestations. Oxford, Oxford University Press.

Ensminger, J., 1996. Culture and property rights. In Hanna, S., C. Folke, and K.G. Mäler (eds.): Rights to Nature: Ecological, Economic, Cultural, and Political Principles of Institutions for the Environment. Washington, Island Press, 179-204.
Eraker, H., 2000. Keiserens nye trær - om norske treplantasjer, CO_2-kvoter og nykolonialisme i Uganda. NorWatch Rapport 5. Framtiden i våre hender.
Flanagan, K., E. Uyarra and M. Laranja, 2010. The 'policy mix' for innovation: rethinking innovation policy in a multi-level, multi-actor context, Manchester, Manchester Business School Working Paper.
Farooqui, M.F. and M. Schultz, 2012. Co-chairs' Summary of Dialogue Seminar on Scaling up Biodiversity Finance, Quito 6-9 March 2012. Montreal: Secretariat of the Convention on Biological Diversity. http://www.dialogueseminars.net/quito/quito_home.html
Feng, Z. H., Y. Wei and K. Wang, 2012. Estimating risk for the carbon market via extreme value theory: An empirical analysis of the EU ETS. Applied Energy 99(2012): 97-108.
Ferraro, P.J., 2008. Asymmetric information and contract design for payments for environmental services. Ecological Economics, 65:810-821.
Fisher, J., 2012. No pay, no care? A case study exploring motivations for participation in payments for ecosystem services in Uganda. Oryx, 46(1):45–54
Frank, R.H., 1987. If Homo Economicus could choose his own utility function, would he want one with a conscience? The American Economic Review, 77:593–604.
Fraser, I. and R. Waschik, 2005. Agricultural Land Retirement and Slippage: Lessons from an Australian Case Study. Land Economics, 82:206–226.
Gamfeldt, L., T. Snäll, R. Bagchi, M. Jonsson, L. Gustavsson and P. Kjellander, L. Gamfeldt, M. C. Ruiz-Jaen, M. Fröberg, J.Stendahl, C. D. Philipson, G. Mikusiński, E. Andersson, B. Westerlund, H. Andrén, F. Moberg, J. Moen, J. Bengtsson, 2013. Higher levels of multiple ecosystem services are found in forests with more tree species. Nature Communications. Nature Communications 4, Article number: 1340 doi:10.1038/ncomms2328
Garbach, K., M. Lubell and F.A.J. DeClerck, 2012. Payment for Ecosystem Services: The roles of positive incentives and information sharing in stimulating adoption of silvopastoral conservation practices. Agricultural Ecosystem Environment, 156:27-36.
García-Amado, L., M. Ruiz Perez and B. Barcía, 2013. Motivation for conservation: Assessing integrated conservation and development projects and payments for environmental services in La Sepultura Biosphere Reserve, Chiapas, Mexico. Ecological Economics, 89:92-100.
Giron, A. and A. Chapoy, 2013. Securitization and financialization. Journal of Post Keynesian Economics, 35(2):171-186.
Gneezy, U. and A. Rustichini, 2000. Pay enough or don't pay at all. Quarterly Journal of Economics, 115:791-810.
Godoy, R., V. Reyes-Garcia, E. Byron, W.R. Leonard and V. Vadez, 2005. The effect of market economies on the well-being of indigenous peoples and on their use of renewable natural resources. Annual Review of Anthropology, 34:121-138
Gómez-Baggethun, E. , R. de Groot, P.L. Lomas and C. Montes, 2010a. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecological Economics, 69(6):1209-1218.
Gómez-Baggethun, E., S. Mingorria, V. Reyes-Garcia, L. Calvet and C. Montes, 2010b. Traditional Ecological Knowledge Trends in the Transition to a Market Economy: Empirical Study in the Doñana Natural Areas. Conservation Biology, 24(3):721-729

Graeber, D., 2011. Toward an Anthropological Theory of value. The False Coin of Our Own Dreams. New York, Palgrave.

Grieg-Gran, M. and J. Bishop, 2004. How Can Markets for Ecosystem Services Benefit the Poor? In Roe, D. (ed.): The Millennium Development Goals and Conservation: Managing Nature's Wealth for Society's Health. International Institute for Environment and Development, London.

Gross-Camp, N.D., A. Martin, S. McGuire, B. Kebede and J. Munyarukaza, 2012. Payments for ecosystem services in an African protected area: exploring issues of legitimacy, fairness, equity and effectiveness. Oryx, 46(1):24-33.

Gunningham, N. and D. Sinclair, 1998. Designing Environmental Policy. In Gunningham, N. and P. Grabosky (eds.), Smart Regulation: Designing Environmental Policy. New York, Oxford University Press, 375-453.

Gunningham, N. and D. Sinclair, 1999. Regulatory Pluralism: Designing Policy Mixes for Environmental Protection. Law & Policy, 21:49-76.

Hahn, T., C. Iuarte-Lima and C. McDermott, 2014. Commodification motives and degrees: the CBD objective may benefit from using price signals as incentives but nor from pricing ecosystem services or fianancialisation. Unpublished paper presented at the Quito Dialogue Seminar, April 9-12, 2014.

Hanley, N., J. Shogren and B. White, 2013. Introduction to Environmental Economics (2nd ed.). Oxford, Oxford University Press.

Harvey, C.A., A. Medina, D. Merlo Sanchez, S. Vilchez, B. Hernandez, J.C. Saenz, J.M. Maes, F. Casanoves and F.L. Sinclair, 2006. Patterns of animal diversity in different forms of tree cover in agricultural landscapes. Ecological Applications, 16:1986-1999.

Hauer, G., V. Adamowicz, F. Schmiegelow, M. Weber, S. Cumming and R. Jagodzinski, 2009. Tradeoffs between forestry resource and conservation values under alternate forest policy regimes: A spatial analysis of the western Canadian boreal plains. Ecological Modelling, 221:2590-2603.

Hellwig, M.F., 2009. Systemic risk in the financial sector: An analysis of the subprime-mortgage financial crisis. De Economist, 157:129–207

Herrera, J.M. and D. Garcia, 2010. Effects of Forest Fragmentation on Seed Dispersal and Seedling Establishment in Ornithochorous Trees. Conservation Biology, 24:1089-1098.

Hess, U., K. Richter and A. Stoppa, 2002. Weather risk management for agriculture and agribusiness in developing countries.In Dischel R.S. (ed.): Climate risk and the weather market, financial risk management with weather hedges. London, UK: Risk Books.

Hiedanpää, J. and D.W. Bromley, 2014. Payments for ecosystem services: durable habits, dubious nudges, and doubtful efficacy. Journal of Institutional Economics, May:1-21.

Hill, J., T. Jennings, and E. Vanezi, 2008. The emissions trading market: risks and challenges, London: Financial Services Authority.

Hodgson, G. M., 2007. The Revival of Veblenian Institutional Economics. Journal of Economic Issues, XLI(2):325-340.

Holland, A., 1997. Substitutability, Or Why Strong Sustainability Is Weak and Absurdly Strong Sustainability is Not Absurd. In J. Foster (ed.): Valuing Nature? Economics, Ethics and Environment. London, Routledge, 119–34.

Honnay, O., H. Jacquemyn, B. Bossuyt and M. Hermy, 2005. Forest fragmentation effects on patch occupancy and population viability of herbaceous plant species. New Phytologist, 166:723-736.

Hultman, N. E., S. Pulver, L. Guimarães, R. Deshmukh and J. Kane, 2012. Carbon market risks and rewards: Firm perceptions of CDM investment decisions in Brazil and India. Energy Policy, 40:90-102.

Ingram, J.C., D. Wilkie, T. Clements, R.B. McNab, F. Nelson, E.H. Baur, H.T. Sachedina, D. D. Peterson and C.A.H. Foley, 2014. Evidence of Payments for Ecosystem Services as a mechanism for supporting biodiversity conservation and rural livelihood. Ecosystem Services, 7:10-21.

International Emissions Trading Association (IETA), 2013. Greenhouse Gas Market 2013 Report: Looking to the Future of Carbon Markets. Ed. Mansell, Anthony. Viewed 10. May 2014. http://www.ieta.org/index.php?option=com_content&view=article&id=779:ghgmarket2013&catid=26:reports&Itemid=93

Jindal, R., B., Swallow and J. Kerr, 2008. Forestry-based carbon sequestration projects in Africa: Potential benefits and challenges. Natural Resources Forum, 32:116-130.

Joppa, L.N. and A. Pfaff, 2009. High and Far: Biases in the Location of Protected Areas. Plos One 4.

Joppa, L.N. and A. Pfaff, 2010. Reassessing the forest impacts of protection The challenge of nonrandom location and a corrective method. Annals of the New York Academy of Sciences, 1185:135-149.

Kahneman, D. and A. Tversky (eds.), 2000. Choices, Values, and Frames, 2000. Cambridge, Cambridge University Press.

Kerr, J., M. Vardhan and R. Jindal, 2012. Prosocial Behavior and Incentives: Evidence from Field Experiments in Rural Mexico and Tanzania. Ecological Economics, 73:220-227.

Kivimaa, P. and P. Mickwitz, 2006. The challenge of greening technologies - Environmental policy integration in Finnish technology policies. Res Policy, 35:729-744.

Knox-Hayes, J., 2013. The spatial and temporal dynamics of value in financialization: Analysis of the infrastructure of carbon markets. Geoforum, 50:117-128.

Kolb, J.A., and R.W. Overdahl (eds.), 2009. Financial Derivatives. Pricing and Risk Management. New Jersey, John Wiley & Sons Inc.

Kosoy N., M. Martinez-Tuna, R. Muradian and J. Martinez-Alier, 2007. Payments for environmental services in watersheds: Insights from a comparative study of three cases in Central America. Ecological Economics, 61: 446-455.

Kosoy, N., E. Corbera and K. Brown, 2008. Participation in payments for ecosystem services: Case studies from the Lacandon rainforest, Mexico. Geoforum, 39:2073–2083

Krause, T. and L. Loft, 2013. Benefit Distribution and Equity in Equador's Socio Bosque Program. Society and Natural Resources, 26:1170-1184.

Lang, C. and T. Byakola, 2006. A funny place to store carbon: UWAFACE Foundation's tree planting project in Mount Elgon National Park, Uganda. World Rainforest Movement, Montevideo, Uruguay

Lavorel, S., K. Grigulis, P. Lamarque, M.P. Colace, D. Garden, J. Girel, G. Pellet and R. Douzet, 2011. Using plant functional traits to understand the landscape distribution of multiple ecosystem services. Journal of Ecology, 99:135-147.

Lavorel, S., J. Storkey, R.D. Bardgett, F. de Bello, M.P. Berg, X. Le Roux, M. Moretti, C. Mulder, R.J. Pakeman, S. Diaz and R. Harrington, 2013. A novel framework for linking functional diversity of plants with other trophic levels for the quantification of ecosystem services. Journal of Vegetation Science, 24(5):942-948.

Lichtenberg, E., 2004. Are Green Payments Good for the Environment? Agricultural and Resource Economics Review, 33:138-147.
Lichtenberg, E. and R. Smith-Ramirez, 2011. Slippage in Conservation Cost Sharing. American Journal of Agricultural Economics, 93(1):113-129.
Locatelli, B., P. Imbach and S. Wunder, 2014. Synergies and trade-offs between ecosystem services in Costa Rica. Environmental Conservation, 41:27-36.
Lohmann, L., 2010. Uncertainty markets and carbon markets: Variations on Polanyian themes. New Political Economy, 15(2):225-254.
London Accord, 2009. Index linked carbon bonds. http://www.london-accord.co.uk/wiki/index.php/Index-Linked_Carbon_Bonds
MEA (Millenium Ecosystem Assessment), 2005. Ecosystems and Human Well-being: Synthesis, UN, Washington, DC.
Mace, G.M., K. Norris and A.H. Fitter, 2012. Biodiversity and ecosystem services: a multilayered relationship. Trends in Ecology & Evolution, 27:19-26.
Madsen, B., N. Carroll and K. Moore Brands, 2010. State of Biodiversity Markets Report: Offset and Compensation Programs Worldwide. Ecosystem Marketplace, Washington DC. http://www.thegef.org/gef/sites/thegef.org/files/publication/sbdmr.pdf
Maestre A., S., L. Calvet Mir, J. C. J. M. van den Bergh, I. Ring and P.H. Verburg, 2012. Ineffective biodiversity policy due to five rebound effects. Ecosystem Services, 101-110.
Mahanty, S., H. Suich and L. Tacconi, 2013. Access and benefits in payments for environmental services and implications for REDD+: Lessons from seven PES schemes. Land Use Policy, 31:38-47.
Mandel, J.T., C.J. Donlan and J. Armstrong, 2010. A derivative approach to endangered species conservation. Frontiers in Ecology and the Environment, 8(1):44-48
Mansanet-Bataller, M.,J. Chevallier, M. Hervé-Mignucci, and E. Alberola, 2011. EUA and sCER phase II price drivers: Unveiling the reasons for the existence of the EUA–sCER spread. Energy Policy, 39(3):1056-1069.
Martinez-Alier, J., 2002. The Environmentalism of the Poor. Cheltenham, Edward Elgar.
Matulis, B.S., 2012. The narrowing gap between vision and execution: Neoliberalization of PES in Costa Rica. Geoforum (2012), http://dx.doi.org/10.1016/j.geoforum.2012.09.001.
May, P.H., J. Andrade, J.L. Vivan, K. Kaechele, M.F. Gebara and R. Abad, 2012. Assessment of the role of economic and regulatory instruments in the conservation policymix for the Brazilian Amazon – a coarse grain analysis. Policymix Report No. 5/2012.
McAffe, K. and E.N. Shapiro, 2010. Payments for Ecosystem Services in Mexico: Nature, Neoliberalism, Social Movements, and the State. Annals of the Association of American Geographers, 100(3):579-599.
McNally, R and M. Levi, 2011. A Crude Predicament**.** The Era of Volatile Oil Prices. Foreign Affairs, July/August, 7/1.
Mendenhall, C.D., C.H. Sekercioglu, F.O. Brenes, P.R. Ehrlich and G.C. Daily, 2011. Predictive model for sustaining biodiversity in tropical countryside. P Natl Acad Sci USA, 108:16313-16316.
Michaelowa, A., 2013. Linking the CDM with domestic carbon markets. Climate Policy, 14(3):353-371.
Michaelowa, A. and F. Jotzo, 2005. Transaction costs, institutional rigidities and the size of the clean development mechanism. Energy Policy, 33:511-523.

Michel, J.V., 2012. Neighbourhood-effects of payments for environmental services. Case study in the Sarapiqui Region,Heredia Province, Costa Rica, Department of International Environment and Development Studies, Master Thesis, University of Life Sciences (UMB), 104.

Milcu, A., C. Roscher, A. Gessler, D. Bachmann, A. Gockele, M. Guderle, D. Landais, C. Piel, C. Escape, S. Devidal, O. Ravel, N. Buchmann, G. Gleixner, A. Hildebrandt and J. Roy, 2014. Functional diversity of leaf nitrogen concentrations drives grassland carbon fluxes. Ecology Letters, 17:435-444.

Mills, M., J. G. Álvarez-Romero, K. Vance-Borland, P. Cohen and R. L. Pressey, 2014. Linking regional planning and local action: Towards using social network analysis in systematic conservation planning. Biological Conservation, 169:6-13.

Miranda, J., G.M. Rusch, P. Casals, F. DeClerck, M. Ibrahim, F. Casanoves and F. Jiménez, 2013. Transferencia de lluvia y captura de nutrientes: efecto de las características y rasgos funcionales de los árboles. Revista Agroforestería de las Américas.

Milder, J.C., S.J. Scherr and C. Bracer, 2010. Trends and Future Potential of Payment for Ecosystem Services to Alleviate Rural Poverty in Developing Countries. Ecology and Society, 15(2):4. URL: http://www.ecologyandsociety.org/vol15/iss2/art4/

Miranda, M., I. Porras and M.L. Moreno, 2003. The social impacts of payments for environmental services in Costa Rica. A quantitative field survey and analysis of the Virilla watershed. International Institute for Environment and Development (IIED).

Mizrach, B., 2012. Integration of the global carbon markets. Energy Economics, 34(1):335-349.

Mol, A. P., 2012. Carbon flows, financial markets and climate change mitigation. Environmental Development, 1(1):10-24.

Mooney, H. and P. Ehrlich, 1997. Ecosystem Services. A Fragmentary History. In Daily, G., (ed.), 1997: Nature's Services. Social Dependence on Natural Ecosystems. Washington DC, Island Press, 11-20.

Montague-Drake, R.M., D.B Lindenmayer and R.B. Cunningham, 2009. Factors affecting site occupancy by woodland bird species of conservation concern. Biological Conservation, 142:2896-2903.

Mulder, C., F.S. Ahrestani, M.B. Bahn, D.A. Bohan, M. Bonkowski, B.S. Griffiths, R.A. Guicharnaud, J. Kattge, P.H. Krogh, S. Lavorel, O.T. Lewis, G. Mancinelli, S. Naeem, J. Peñuelas, H. Poorter, P. Reich, L. Rossi, G.M. Rusch, J. Sardans and I.J. Wright, 2013. Connecting the green and brown worlds: Allometric and stoichiometric predictability of above- and below-ground networks, In Woodward, G. (Ed.), Advances in Ecological Research (49):69-175.

Muradian, R., M. Martinez-Tuna, N. Kosoy, M. Perez and J. Martinez-Alier, 2008. Institutions and the performance of payments for water-related environmental services. Lessons from Latin America. Working paper. Development Research Institute, Tilburg University.

Muradian, R., E. Corbera, U. Pascual, N. Kosoy and P.H. May, 2010. Reconciling theory and practice: An alternative conceptual framework for understanding payments for environmental services. Ecological Economics, 69:1202-1208.

Muradian, R., M. Arsel, L. Pellegrini, F. Adaman, B. Aguilar, B. Agarwal, E. Corbera, D. Ezzine de Blas, J. Farley, G. Froger, E. Garcia-Frapolli, E. Gómez-Baggethun, J. Gowdy, N. Kosoy, J.F. Le Coq, P. Leroy, P. May, P. Méral, P. Mibielli, R. Norgaard, B. Ozkaynak, U. Pascual, W. Pengue, M. Perez, D. Pesche, R. Pirard, J. Ramos-Martin, L. Rival, F. Saenz, G. Van Hecken, A. Vatn, B. Vira and K. Urama, 2013. Payments for ecosystem services and the fatal attraction of win-win solutions. Conservation letters, 6(4):274-279.

Murray, B.C., B.A. McCarl and H. Lee, 2004. Estimating Leakage from Forest Carbon Sequestration Programs. Land Economics, 80:109-124.
Nakakaawa, C.A., P.O. Vedeld and J.B. Aune, 2011. Spatial and temporal land use and carbon stock changes in Uganda: implications for a future REDD strategy. Mitigation and Adaptation Strategy and Global Change, 16:25–62.
Nanbanoga, G. N and J. Namaalwa 2013. What payment systems for REDD+ do local people favor? Experiences from Uganda. Presentation at the conference 'Options for National REDD+ Architectures', UMB, May 29-31, 2013.
Naidoo, R., A. Balmford, P.J. Ferraro, S. Polasky, T.H. Ricketts and M. Rouget, 2006. Integrating economic costs into conservation planning. Trends in Ecology & Evolution, 21:681-687.
Niemeyer, R.J., A.K. Fremier, R. Heinse, W. Chavez and F.A.J. DeClerck, 2014. Woody Vegetation Increases Saturated Hydraulic Conductivity in Dry Tropical Nicaragua. Vadose Zone Journal, 13.
Nybø, S., G. Certain and O. Skarpaas, 2012. The Norwegian Nature Index – state and trends of biodiversity in Norway. Norsk Geografisk Tidsskrift - Norwegian Journal of Geography, 5:241-249.
OECD, 2007. Instrument Mixes for Environmental Policy. OECD, Paris.
OECD, 2013. Scaling-up Financial Mechanisms for Biodiversity. OECD Publication. http://dx.doi.org/10.1787/9789264193833-en
O'Neill, J., 1993. Ecology, Policy and Politics. Human Well-Being and the Natural World. London, Routledge.
O'Neill J., A. Holland and A. Leist, 2008. Environmental Values. London, Routledge.
Ormerod, P., 2012. Positive Linking. How Networks Can Revolutionise the World. Croydon, Faber and Faber Ltd.
Ospina, S., G.M. Rusch, D.A. Pezo, F. Casanoves and F.L. Sinclair, 2012. More stable productivity of semi natural grasslands than sown pastures in a seasonally dry climate. Plos One 7, e35555. DOI: 10.1371/journal.pone.0035555
Ostrom, E., 2005. Understanding Institutional Diversity. Princeton, NJ, Princeton University Press.
Pagiola, S., 2008. Payments for environmental services in Costa Rica. Ecological Economics, 65:712-724.
Pagiola, S., and G. Platais, 2007. Payments for Environmental Services: From Theory to Practice. World Bank, Washington.
Pagiola, S., A. Arcenas and G. Platais, 2005. Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. World Development, 33:237-253.
Pannell, D.J., 2008. Public Benefits, Private Benefits, and Policy Mechanism Choice for Land-Use Change for Environmental Benefits. Land Economics, 84:225-240.
Pattanayak, S.K., S. Wunder and P.J. Ferraro, 2010 Show me the money: do payments supply environmental services in developing countries? Review of Environmental Economics and Policy, 42:254–274.
Pejchar, L., R.M. Pringle, J. Ranganathan, J.R. Zook, G. Duran, F. Oviedo and G.C. Daily, 2008. Birds as agents of seed dispersal in a human-dominated landscape in southern Costa Rica. Biological Conservation, 141:536-544.
Perrot-Maître, D., 2006. The Vittel payments for ecosystem services: a "perfect" PES case? International Institute for Environment and Development, London, UK

Pfaff, A. and A. Robalino, 2012. Protecting forests, biodiversity, and the climate: predicting policy impact to improve policy choice. Oxford Review of Economic Policy, 28:164–179.

Pfaff, A., J. Robalino, G.A. Sanchez-Azofeifa, K.S. Andam and P.J. Ferraro, 2009. Park location affects forest protection: Land characteristics cause differences in park impacts across Costa Rica. The B.E. Journal of Economic Analysis & Policy, 9(2):5.

Polanyi K. 1944 [1957]. The Great Transformation. The political and economic origins of our time. Boston, Beacon Press.

Porras, I., M. Grieg-Gran and N. Neves, 2008. All that glitters. A review of payments for watershed services in developing countries. Report, International Institute for Environment and Development, London.

Porras, I., (ed.) 2010. Tracking the social impacts of payments for environmental services in Costa Rica. In International Institute for Environment and Development, London.

Porras, I., D.N. Barton, M. Miranda and A. Chacón-Cascante, 2013. Learning from 20 years of Payments for Ecosystem Services in Costa Rica. International Institute for Environment and Development, London.

Porras, I., A. Chacon-Cascante, D.N. Barton and D. Tobar, 2014, forthcoming. Ecosystems for sale: Land prices and Payments for Ecosystem Services in Costa Rica. International Institute for Environment and Development.

Primmer, E., R. Paloniemi, J. Similä, P. Leskinen, P. Punttila, A. Tainio, S. Sironen and N. Leikola, 2013. Finland: Assessment of existing and proposed policy instruments for biodiversity conservation at national level. POLICYMIX Report 2/2013.

Primmer, E., R. Paloniemi, J. Similä and A. Tainio, 2014. Forest owner perceptions of institutions and voluntary contracting for biodiversity conservation: Not crowding out but staying out. Ecological Economics, 103:1-10.

Pryke, M. and J. Allen, 2000. Monetized time-space: derivatives – money's 'new imaginary'? Economy and Society, 29(2):264-284.

RCG, R.C.G.L., 2008. Actualización Plataforma de Valores de Terrenos por Zonas Homogéneas. San José: Ministerio de Hacienda, Dirección General de Tributación. División Organo de Normalización Técnica, Gobierno de Costa Rica.

Reid, J.L., C.D. Mendenhall, J. Abel Rosales, R.A. Zahawi and K.D. Holl, 2014. Landscape Context Mediates Avian Habitat Choice in Tropical Forest Restoration. Plos One 9.

Ricketts, T.H., J. Regetz, I. Steffan-Dewenter, S. A. Cunningham, C. Kremen, A. Bogdanski, B. Gemmill-Herren, S. S. Greenleaf, A. M. Klein, M. M. Mayfield, L. A. Morandin, A. Ochieng and B. F. Viana, 2008. Landscape effects on crop pollination services: are there general patterns? Ecology Letters, 11:499-515.

Ring, I. and D.N. Barton, forthcoming. Economic instruments in policy mixes for biodiversity conservation and ecosystem governance. In Martinez-Alier J. and R. Muradian (Ed.), Handbook of Ecological Economics. Cheltenham, Edward Elgar.

Ring, I. and C. Schröter-Schlaack, 2011a. Instrument Mixes for Biodiversity Policies. POLICY-MIX Report, Issue No. 2/2011. UFZ - Helmholtz Centre for Environnmental Research, Leipzig. http://policymix.nina.no, p. 208.

Ring, I. and C. Schröter-Schlaack, 2011b. Justifying and assessing policy mixes for biodiversity and ecosystem governance. In Ring, I. and C. Schröter-Schlaack (Ed.), Instrument Mixes for Biodiversity Policies. POLICYMIX Report, Issue No. 2/2011. UFZ - Helmholtz Centre for Environnmental Research, Leipzig, 14-35.

Robalino, J., A. Pfaff, G.A. Sánchez-Azofeifa, F. Alpízar, C. León and C.M. Rodríguez, 2008. Deforestation impacts of environmental services payments. Costa Rica's PSA Program 2000-2005. Environment for Development Discussion Paper Series, August.

Robalino, J., C. Sandoval-Alvarado, A. Pfaff, D.N. Barton and A. Chacón-Cascante, 2014. Substitutability and complementarity of forest conservation policies. In Ring, I., D.N. Barton and G.M. Rusch (eds.), Policy Mixes in Environmental and Conservation Policies. UFZ, Leipzig, Germany, 25–27 February 2014.

Robalino, J.A., 2007. Land conservation policies and income distribution: who bears the burden of our environmental efforts? Environ Dev Econ, 12:521-533.

Robalino, J.A. and A. Pfaff, 2012. Contagious development: Neighbor interactions in deforestation. J Dev Econ, 97:427-436.

Robalino, J.A. and A. Pfaff, 2013. Ecopayments and Deforestation in Costa Rica: A Nationwide Analysis of PSA's Initial Years. Land Economics, 89:432-448.

Rockstrøm, J., W. Steffen, K. Noone, Å. Persson, F.S.I. Chapin, E. Lambin, T. Lenton, M. Scheffer, C. Folke, H.J. Schellhuber, B. Nykvist, C.A. de Wit, T. Hughes, S. van der Leeuw, H. Rodhe, S. Sörlin, P.K. Snyder, R. Costanza, U. Svedin, M. Falkenmark, L. Karlberg, R.W. Corell, V.J. Fabery, J. Hansen, B. Walker, D. Liverman, K. Richardson, P. Crutzen and J. Foley, 2009. Planetary boundaries: Exploring the safe operating space for humanity. Ecology and Society, 14(32).

Rode, J., E. Gómez-Baggethun and T. Krause, 2013. Economic incentives for biodiversity conservation: what is the evidence for motivation crowding?, UFZ Discussion Papers 19/2013. December 2013. Helmholtz-Zentrum für Umwelforschung GmbH - UFZ.

Rodríguez-Sickert, C., R.A. Guzmán and J.C. Cardenas, 2008. Institutions influence preferences: Evidence from a common pool resource experiment. Journal of Economic Behaviour and Organization, 67:215–227.

Romeiro, A.R., P. Bernasconi, B.P. Puga, D. Caixeta Andrade and R.P. Sobrinho, 2012. Assessment of existing and proposed policy instruments for biodiversity conservation in São Paulo -Brazil: a coarse grain analysis. POLICYMIX Report Issue No 3/2012.

Rusch, G.M., D.N. Barton and P. Bernasconi, 2013. Best practice guidelines for assessing effectiveness of instruments on biodiversity conservation and ecosystem services provision. POLICYMIX Technical Brief No. 10. In POLICYMIX (Ed.).

Rørstad, P.K., A. Vatn and V. Kvakkestad, 2007. Why do transaction costs of agricultural policies vary? Agricultural Economics, 36:1-11.

Sagoff, M., 1988. The Economy of the Earth: Philosophy, Law and Environment. Cambridge, Cambridge University Press.

Santos, R., P. Antunes, P. Clemente and T. Ribas, 2012. Assessment of the role of economic instruments in the Portuguese conservation policymix – a national coarse grain analysis. POLICYMIX Report Issue No 6/2012.

Schneider, M., H. Hendrichs and V.H. Hoffmann, 2010. Navigating the global carbon market An analysis of the CDM's value chain and prevalent business models. Energy Policy, 38:277–287

Schotter, A.R., 1994. Microeconomics. A Modern Approach. New York, HarperCollins College Publishers.

Schröter-Schlaack, C., I. Ring, S. Möckel, C. Schulz-Zunkel, N. Lienhoop, R. Klenke and T. Lenk, 2013. Assessment of existing and proposed policy instruments for biodiversity conservation in Germany: The role of ecological fiscal transfers, POLICYMIX Report No. 1/2013. UFZ-Helmholtz Centre for Environmental Research, Leipzig.

Schröter, M., G.M. Rusch, D.N. Barton, S. Blumentrath and B. Nordén, forthcoming. Ecosystem services and opportunity cost levels shift spatial priorities for forest biodiversity conservation.

Schroth, G. and C.A. Harvey, 2007. Biodiversity conservation in cocoa production landscapes: an overview. Biodiversity and Conservation, 16:2237-2244.

Shin, H.S., 2009. Securitization and financial stability. The Economic Journal, 119:309–332.

Sierra, R. and E. Russman, 2006. On the efficiency of the environmental service payments: a forest conservation assessment in the Osa Peninsula, Costa Rica. Ecological Economics, 59:131-141.

Simon, H.A., 1956. Rational Choice and the Structure of the Environment. Psychological Review, 63:129-138.

Sjaastad, E. and B. Cousins, 2009. Formalisation of Land Rights in the South: An Overview. Land Use Policy, 26(1):1-9.

Soma, K. and A. Vatn, 2010. Is there anything like a citizen? A descriptive analysis of instituting a citizen's role to represent social values at the municipal level. Environmental Policy and Governance, 20:30–43.

Spash C.L., 2000. Multiple Value Expression in Contingent Valuation: Economics and Ethics. Environmental Science Technology, 34:1433-1438.

Spash C.L., 2009. The new environmental pragmatists, pluralism and sustainability. Environmental Values, 18(3):253-256.

Spash C.L., 2011. Terrible Economics, Ecosystems and Banking. Environmental Values, 20(2):141–145.

Strange S., 1998. What theory? The theory in Mad Money. CSGR Working Paper. http://www2.warwick.ac.uk/fac/soc/csgr/research/workingpapers/1998/wp1898.pdf.

Steffan-Dewenter, I., U. Munzenberg, C. Burger, C. Thies and T. Tscharntke, 2002. Scale-dependent effects of landscape context on three pollinator guilds. Ecology Letters, 83:1421-1432.

Steffen, W., A. Sanderson, P.D., Tyson, J. Jäger, P.A. Matson, B. Moore III, F. Oldfield, K. Richardson, H.J. Schnellnhuber, B.L. Turner and R.J. Wasson, 2005. Global Change and the Earth System. A Planet Under Pressure. Berlin, Springer Verlag. 2nd printing.

Sullivan, S., 2010 The environmentality of 'Earth Incorporated': on contemporary primitive accumulation and the financialisation of environmental conservation. Paper presented at the conference 'An Environmental History of Neoliberalism', Lund University, 6-8 May 2010

Sullivan, S., 2013. Banking Nature? The Spectacular Financialisation of Environmental Conservation. Antipode, 45(1):198-217.

Sutherland, L.A., D. Gabriel, L. Hathaway-Jenkins, U. Pascual, U. Schmutz, D. Rigby, R. Godwin, S.M. Sait, R. Sakrabani, W.E. Kunin, T.G. Benton and S. Stagl, 2012. The 'Neighbourhood Effect': A multidisciplinary assessment of the case for farmer co-ordination in agri-environmental programmes. Land Use Policy, 29:502-512.

Taschinia, L., M. Chesney, and M. Wang, 2013. Experimental Comparison between Markets on Dynamic Permit Trading and Investment in Irreversible Abatement with and without Non-Regulated Companies (August 13, 2013). Centre for Climate Change Economics and Policy Working Paper No. 51.

Tickell, A., 2000. Dangerous derivatives: controlling and creating risk in international money.

Geoforum, 31:87-99.
TEEB, 2010. Mainstreaming the Economics of Nature: A Synthesis of the approach, conclusions and Recommendations of TEEB. Available at http://www.teebweb.org/TEEBSynthesis-Report/tabid/29410/Default.aspx.
Tscharntke, T., Y. Clough, S.A. Bhagwat, D. Buchori, H. Faust, D. Hertel, D. Holscher, J. Juhrbandt, M. Kessler, I. Perfecto, S. Scherber, G. Schroth, E. Veldkamp and T.C. Wanger, 2011. Multifunctional shade-tree management in tropical agroforestry landscapes - a review. Journal of Applied Ecology, 48:619-629.
Tversky, A., P. Slovic and D. Kahneman, 1990. The Causes of Preference Reversal. The American Economic Review, 80(1):204- 217.
Uchida, E., S. Rozelle and J.T. Xu, 2009. Conservation Payments, Liquidity Constraints, and Off-Farm Labor: Impact of the Grain-for-Green Program on Rural Households in China. American Journal of Agricultural Economics, 91:70-86.
Unruh, J. D., 2008. Carbon sequestration in Africa: The land tenure problem. Global Environmental Change, 18(4):700-707
Urwin, K. and A. Jordan, 2008. Does public policy support or undermine climate change adaptation? Exploring policy interplay across different scales of governance. Global Environmental Change, 18:180-191.
van Breugel, M., J.S Hall, D. Craven, M. Bailon, A. Hernandez, M. Abbene and P. van Breugel, 2013. Succession of ephemeral secondary forests and their limited role for the conservation of floristic diversity in a human-modified tropical landscape. Plos One 8.
van der Horst, D., 2011. Adoption of payments for ecosystem services: An application of the Hägerstrand model. Applied Geography, 31:668-676.
Vatn, A., 1998. Input vs. Emission Taxes. Environmental Taxes in a Mass Balance and Transactions Cost Perspective. Land Economics, 74(4):514-525.
Vatn A., 2000. The Environment as a Commodity. Environmental Values, 9:493-509.
Vatn A. 2005. Institutions and the Environment. Cheltenham, Edward Elgar.
Vatn A., 2009. An Institutional Analysis of Methods for Environmental Appraisal. Ecological Economics, 68:2207-2215.
Vatn, A., 2010. An institutional analysis of payments for environmental services. Ecological Economics, 69:1245-1252.
Vatn, A., 2011. Environmental Governance – A Conceptualization. In Kjosavik, D. and P. Vedeld (eds.): The Political Economy of Environment and Development in a Globalized World. Exploring the Frontiers. Trondheim, Tapir Academic Press, 131-152.
Vatn, A. and D.W. Bromley, 1994. Choices without Prices without Apologies. Journal for Environmental Economics and Management, 26:129-148.
Vatn, A., D.N. Barton, H. Lindhjem, S. Movik, I. Ring and R. Santos, 2011. Can markets protect biodiversity. An evaluation of financial mechanisms. Noragric Report No. 60. Department of International Environment and Development Studies, Noragric, Norwegian University of Life Sciences, UMB. (Also published as Norad Report 19/2011).
Vatn, A., G. Kajembe, R. Leiva-Montoya, E. Mosi, M. Nantongo and D.A Santos Silayo, 2013. Instituting REDD+. An analysis of the processes and outcomes of instituting REDD+ in two pilot areas – RDS Rio Negro (Brazil) and Kilosa (Tanzania). London, International Institute for Environment and Development.
Venter, O., W.F. Laurence, T. Iwamura, K.A. Wilson, R.A. Fuller and H.P. Possingham, 2009. Harnessing Carbon Payments to Protect Biodiversity. Science, 326:1368-1368

Viana, V.M., 2008. Bolsa Floresta (Forest Conservation Allowance): An innovative mechanism to promote health in traditional communities in the Amazon. Estudos Avancados, 22(64):143-153.

Vila, M., J. Vayreda, L. Comas, J. Josep Ibanez, T. Mata and B. Obon, 2007. Species richness and wood production: a positive association in Mediterranean forests. Ecology Letters, 10:241-250.

Wertz-Kanounnikoff, S., M. Kongphan-Apirak and S. Wunder, 2008. Reducing forest emissions in the Amazon Basin. A review of drivers of land-use change and how payments for environmental services (PES) schemes can affect them. Working paper no 40. Bogor: CIFOR.

Westermann, O., 2007. Poverty, Access and Payments for Watershed Hydrological Services. A social feasibility study with case in Tiquipaya Watershed Bolivia. PhD dissertation. Department of Environmental, Social and Spatial Changes, Roskilde University Centre, Denmark.

World Bank, 2011. State and Trend of the Carbon Market 2011. Washington DC

World Bank, 2012. State and Trend of the Carbon Market 2012. Washington DC.

Wu, J., 2000. Slippage Effects of the Conservation Reserve Programs. American Journal of Agricultural Economics, 82:979-992.

Wu, J., 2005. Slippage Effects of the Conservation Reserve Programs: Reply. American Journal of Agricultural Economics, 87:251-254.

Wu, J., D. Zilberman and B.A. Babcock, 2001. Environmental and Distributional Impacts of Conservation Targeting Strategies. Journal of Environmental Economics and Management, 41:333-350.

Wunder, S., 2005. Payments for environmental services: Some nuts and bolts. CIFOR Occasional Paper No. 42. Center for International Forestry Research, Bogor, Indonesia.

Wunder, S., 2007. The Efficiency of Payments for Environmental Services in Tropical Conservation. Conservation Biology, 21(1):48–58.

Wunder S. and M. Alban, 2008. Decentralized payments for environmental services: The cases of Pimampiro and PROFAFOR in Ecuador. Ecological Economics, 65:685-698.

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

Young, O.R., 2002. The Institutional Dimensions of Environmental Change. Fit, Interplay, and Scale. Cambridge MA, MIT Press.

Zbinden, S. and D.R. Lee, 2005. Paying for Environmental Services: An Analysis Of Participation in Costa Rica's PSA Program. World Development, 33:255–272.

Zapata, C. , J. Robalino, M. Ibrahim, D.N. Barton, A. Solarte and D. Tobar, forthcoming. Impact of a mix of payments for environmental services and technical extension on the conservation and silvopastoral practices in Quindio, Columbia.

Zupan, L., M. Cabeza, L. Maiorano, C. Roquet, V. Devictor, S. Lavergne, D. Mouillot, N. Mouquet, J. Renaud and W. Thuiller, 2014 Spatial mismatch of phylogenetic diversity across three vertebrate groups and protected areas in Europe. Diversity and Distributions, 20(6):674-685.

APPENDICES

Appendix 1: Core concepts

Arbitrage

Arbitrage implies utilizing or taking advantage of a price difference between two or more markets. It can take different forms, but implies basically the purchase of securities – a debt, an equity or a derivative - in one market for resale in another market in order to profit from a discrepancy in prices. There is typically a distinction made between risk and risk-free profits from arbitrage.

Biodiversity offsets

Biodiversity offsets are "designed to compensate for significant residual adverse biodiversity impacts arising from project development and persisting after appropriate prevention and mitigation measures have been implemented. The goal of biodiversity offsets is to achieve no net loss "(BBOP, 2009:6).

Biodiversity offsets are environmental liabilities of development. A developer's initial project plans may entail environmental liability being subject to a mitigation plan following an Environmental Impact Assessment. The mitigation hierarchy regards biodiversity offsets as a measure 'of last resort' after having taken prior mitigation steps to 1. avoid; 2. minimize; and 3. mitigate impacts on-site. Offsets are meant to compensate for the residual on-site impact after these measures have been implemented.

Biodiversity offsets can be formulated as a liability to compensate without the option to trade it – i.e., a publicly administered system. Biodiversity offsets with habitat banking involved trade. There is substantial debate about the effect of biodiversity offsets – especially if no net loss is or even can be obtained and how to ensure that steps 1 and 2 above are taken seriously.

Brokers

A broker is a person or firm, which primarily assists clients or investors in buying and selling, for example by facilitating the trade, acting as an agent and a commission. They also represent sellers in the market, helping determine market value, the best options, finding buyers etc. Brokers can trade on their own or for a securities or brokerage firm, but tend to be in direct contact with clients and trading based on their wishes and needs. Brokers are not only found in the private sector; also NGOs facilitate trades and assist project developers in markets for ecosystem services.

Bonds

A bond is a debt security between an issuer and a holder. It is used to raise capital by borrowing. Bonds can be issued by agents like governments, municipalities and corporations. The issuer is obliged to pay the holders a (pre-defined) interest and to repay the debt at a predefined date – the date of maturity. A bond is typically negotiable, which means that the ownership of the instrument can be transferred in the secondary market. Bonds and stocks are both securities – see below. The main difference is that while stockholders are investors – they have an equity stake in the company –bondholders are lenders – i.e., they have a creditor stake. This means that bondholders have priority and will be repaid before stockholders if the issuer goes bankrupt. Special types of bonds of interest for this report are ‘green’ bonds and ‘forest’ bonds. These are oriented towards raising capital for environmentally oriented projects, the former typically oriented towards climate projects and the latter towards reforestation etc.

Cap-and-trade

Cap-and-trade is a policy tool/regulatory system where a mandatory cap is put on e.g. emissions and there is some flexibility in how involved actors can comply towards this cap. The purpose is to reduce emissions, pollution or other unwanted actions. Companies or other entities can then choose to sell (or trade) unused allowances, or innovate or invest in equipment to meet their allocated limit. The cap ensures an economic incentive to pollute less.

Certified Emission Reductions (CERs)

The *CDM* (see below) allows emission-reduction projects in developing countries to earn certified emission reduction (CER) credits, each equivalent to one ton of CO_2. These CERs can be traded and sold, and used by industrialized countries to a meet a part of their emission reduction targets under the Kyoto Protocol (UNFCCC).

The Clean Development Mechanism (CDM)

The CDM is one of three flexibility mechanism under the Kyoto protocol, defined in Article 12 of the Protocol. It allows a country with an emission-reduction or emission-limitation commitment under the Kyoto Protocol (Annex B Party) to implement an emission-reduction project in developing countries. The CDM has enabled an estimated USD 315 billion in capital investment into more than 7500 projects, and resulted in 1,4 billion issued carbon credits from developing countries. A CDM project must provide emission reductions that are additional to what would otherwise have occurred, i.e. the criteria of a*dditionality.*

Derivatives

A derivative is a financial contract, a security, which *derives* its value from one or more underlying. The latter could be an asset like grain, timber etc., but also an interest rate or an index. The value of the derivative is determined by fluctuations in the underlying asset, for example stocks or bonds. Derivatives are used to hedge risk, but can also be used for speculation. common derivatives include forwards, futures, swaps and options.

- *Forwards* are customized contracts (usually between two parties) to buy or sell an asset at a fixed price in the future. It is usually highly customized and traded over-the-counter.
- *Futures* are second level securities, derived from e.g. stocks or commodities. A futures contract set out the terms for buying something not yet produced at a defined price, primarily to hedge or speculate rather than exchange goods. The futures market is characterized by high liquidity, risk and complexity. The futures market helps determine prices, risk reduction due to pre-set prices.

- In the case of a *swap* two counterparties exchange cash flows of a financial instrument of one party for that of the another – i.e. the two parties agree to exchange one flow of cash against another.
- An *option* is a contract offering a buyer the right, but not the obligation to buy or sell an asset at a set price, at a fixed point of time. Such securities are attractive for those who believes the price of the underlying asset will increase or drop.

Ecosystem services and nature values

The concept 'nature values' is used to describe both anthropocentric and non-anthropocentric/ 'intrinsic' values related to nature. The anthropocentric values regard values that support human well-being. Non-anthropocentric or 'intrinsic' values refer to nature's 'own worth'. Such values seem not to exist independent of human observation, though. It is humans that assign such value to things, species and processes. The point is nevertheless that it is not their 'usefulness' to humans – their basis as constituents of human well-being – that is the source of the value we assign. It is rather that we observe and accept that other parts of nature – especially other species – have a right to exist. We appreciate their integrity. What this demands of us is a contested topic – see e.g., O'Neill et al. (2008)

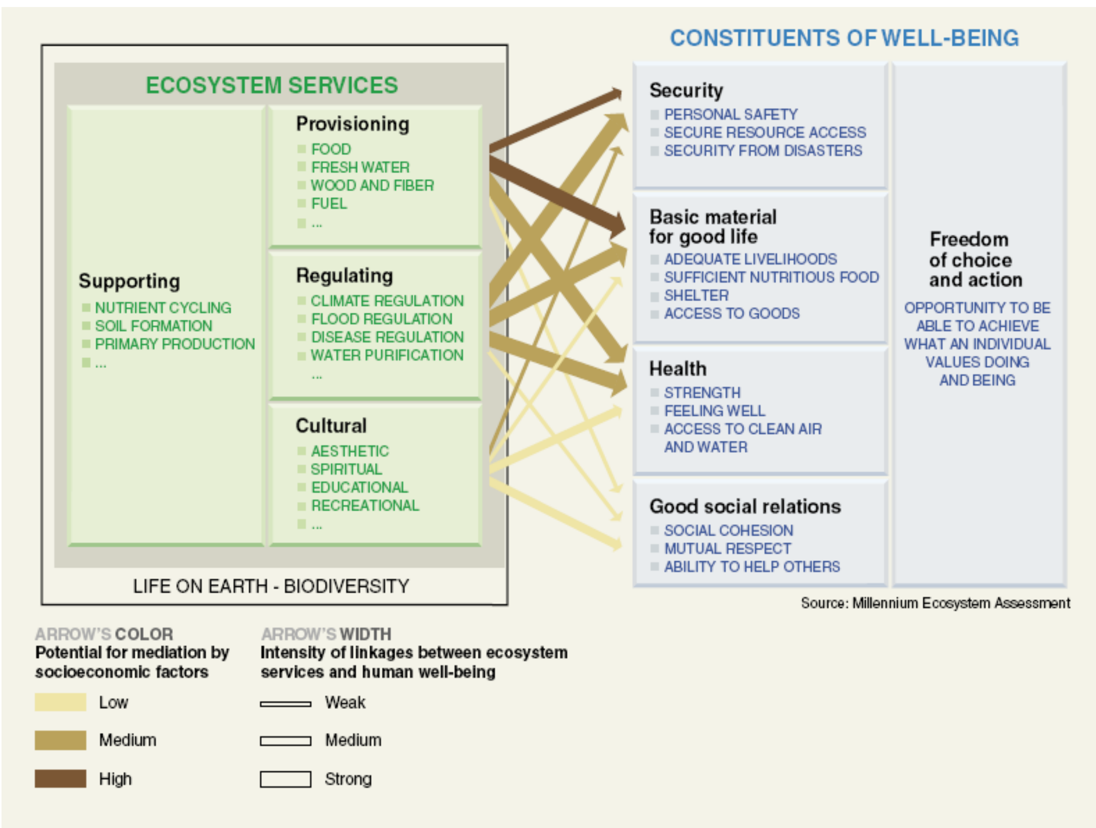

Figure A1.1. The Links between Ecosystem Services and Human Well-being.
Source: MEA (2005)

The concept of ecosystem services (ES) is a sub-category of the nature values concept emphasizing the anthropocentric aspects only. According the Mooney and Ehrlich (1997) the concept was first used in writing in 1970. Its spread and extended use seema vastly enhanced through Daily (1997) and MEA (2005). The latter publication produced a conceptualization of ES and its link to human well-being. Figure A5.1 gives an overview of the MEA synthesis.

There has been a substantial debate about the 'value' of thinking of nature in ES terms – see e.g., Gómez-Baggethun (2010a). This refelects partly the above distinction between 'nature values' and ES and the worry that the ES focus will narrow down our understanding of nature. It reflects also the view that the focus on services will result in a thinking of nature as 'commodities' – as goods and services to be traded and should have a price. While I find this to be an important debate, I think the ES concept is still a valuable term. Care regarding its use, is however warranted.

Equity

Equity commonly refers to ownership in an asset after its debts are paid off. Examples includes stocks or a security representing an ownership interest.

European Union Allowance (EUA)

The tradable unit under the EU ETS. One EUA represents the right to emit 1 ton of CO_2.

European Union Emission Trading System (EU ETS)

The ETS is the largest multi-national, greenhouse gas emissions trading scheme in the world and a key climate change policy tool to reduce greenhouse gas emissions. It covers more than 11,000 power stations and industrial plants in 31 countries. The first compliance phase was from 2005 to 2007, the second from 2008 to 2012 and the third phase runs from 2013 to 2020.

Exchanges

Exchanges are highly organized markets for trade in standardized financial products/securities. Prices and quantities are regularly displayed (see also Over the Counter)

Financialization

Financialization is the turning of tradable commodities' values into financial objects that can themselves be traded. Examples of financialization are creation of derivatives and securitization (see these concepts)

Hedging

To hedge implies an investment to reduce the risk of adverse price movements. A hedger is then someone that sells or buys in the futures market in order to secure the future price of the commodity traded, usually to protect against price risks and volatility. As opposed to hedgers, speculators aim to profit from such price changes that hedgers aim to protect themselves from.

Issued vs. non-issued CERs

All CDM project go through a long and costly registration process, facing multiple risks at the various stages of project development. Once a project is registered, CERs generated from the project may be verified and issued, i.e. confirmed as actual emissions reductions and risk free units in terms of compliance. We observe here a distinction between pCERs and sCERs. While this may not always be the case, pCERs are usually purchased directly from projects at an early phase when it may not have been registered or implemented. What the literature commonly refers to as secondary CERs (sCERs) is therefore assumed to be already issued or contracted and guaranteed credits that carry much lower delivery risks.

Leverage

Leverage – as the concept is used in this report – regards ways to increase gains from investments – typically through using borrowed funds to invest in an asset. If the returns from the asset is above the interests paid on borrowed funds, the return on equity can increase substantially. If this is not the case, the investor may face bankruptcy. Hence, leveraging increases risk and was an important element in the recent financial crisis.

Motivational crowding

'Crowding-in' effects mean that the monetary transfer strengthens the biodiversity conservation incentives provided by intrinsic motivations, while 'crowding-out' effects mean that the monetary transfer reduces those intrinsic motivations.

Over the Counter (OTC)

Over the counter (OTC) is also called off-exchange trading (see Exchanges above). It is done directly between two parties and typically in non-standardized products. Prices are often not published for the public.

Payments for ecosystem services (PES)

Wunder (2005:3) offers the most referred definition of PES. He states that PES is:

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

In practice, however, the money used for payments are often gathered through non-voluntary means – e.g., taxes or user fees. As a consequence, Muradian et al. (2010:1205) see "PES as a transfer of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources."

We note that what distinguishes PES from e.g., payments in a carbon market is the fact that those paying in the case of PES are not forced by a cap. Reducing environmental values through cutting forests or pollution – i.e., cost shifting – is allowed. Hence, those paying do voluntarily take on a responsibility for reducing environmental damages. Note that in the special case of the state, tax payers are really the buyers. They are, however, forced through paying taxes etc. So, the voluntary aspect here is with the political actor – who acts on behalf of its citizens.

Policy mix

A combination of policy instruments which has evolved to influence the quantity and quality of biodiversity conservation and ecosystem service provision in public and private sectors.

Policyscape

A spatially explicit policy mix for biodiversity conservation and sustainable land use.

Primary market

This is where the primary transaction occurs, mainly between project developer and an intermediary or final buyer. The trade is often structured using a contract format called 'Emission Reduction Purchase Agreement' (ERPA), and may be forward, spot or option contracts, or combinations. The contract has important provisions like purchase price, volume and shortfall, legal title to CERs, liabilities etc. CERs generated in the primary market are commonly referred to as pCERs.

Public goods and services

Public goods/services (square IV in Figure A1.2) are characterized by being *non-rival* in use or consumption – i.e., the use/consumption of one actor does not influence the use/consumption of another – and having *high exclusion costs* – i.e., it is very costly if not impossible to turn the good/service into a commodity. Private goods (I) have the opposite characteristics.

		Costs of exclusion (TC)	
		Low	High
Rivalry in use or in consumption	Yes	I	III
	No	II	IV

Figure A1.2 Characterisation of goods and services according to costs of exclusion and rivalry in use or in consumption.

The figure also includes two more categories – club goods (II) and common-pool resources (III). Nature values/ES are dominantly difficult to commoditize as exclusion costs are high. In the main text, we say that they are public goods. That is actually a simplification as there is often rivalry in use/consumption. Hence, many are common-pool. That is why use creates problems for their maintenance. In the present report the focus is on involving nature values/ES in trade. In that regard, it is the issue of exclusion that is the most important, and we have simplified by referring systematically to public goods and services only.

Secondary market

All subsequent transactions following the primary sale, i.e. when the credit is resold in the market place, is encompassed by the secondary market. CERs from the secondary market are commonly referred to as sCERs. In the early phases of the market, the primary market was the main market; however, from approximately 2008 onwards the amount of secondary CERs grew quickly. In 2010, it amounted to 13 % of traded credits in the global carbon market as opposed to only 1 % of primary.

Securities

A security can be defined as an instrument that represents financial value, and is fungible and negotiable, i.e., tradable. Securities are broadly categorized as

- Debt securities (such as bonds, loans and debentures)
- Equity securities (such as stocks)
- Derivatives (such as forwards, futures, options and swaps)

Social network

Random networks have no predetermined structure and are familiar in epidemicology. Small-world networks have structure, which corresponds to the social situation of 'overlapping friends of friends' structures. Scale-free networks have many forms, but are generally recognizable by a few agents acting as 'hubs' with many connections, in contrast to most other agents in the network who are linked to a small number of others.

Securitization

Securitization implies the creation of a financial instrument through combining different (financial) assets and subsequently offering repackaged instruments to investors. These repackaged instruments can be bonds. This process creates liquidity and allows small investors to buy shares into a larger pool of assets. Securitization typically involves tranching (see below).

Traders

While there are many similarities between brokers and traders, there are some important differences. First of all, traders usually work for large investment management firms, buying and selling based on the strategies for asset management in that firm. Brokers, as mentioned above, usually trade for clients, based on their needs and wishes. Traders can trade for themselves or someone else, but unlike e.g. investors who take a long-term perspective on their investments, traders tend to hold assets for a short time and aim to capitalize on short-term trends. Thus, they focus on the market itself rather than commodity fundamentals, in order to take advantage of small price movements.

Tranching

Tranche, French for "slice", implies a slice of pooled securities with different risks, rewards and/or delivery into the same transaction. Each trance has a different degree of risk, so tranches have different capital or liability structures. The tranches are offered to investors, who will be interested in different risk profiles. E.g., pension funds will prefer senior tranches that carry less risk. The process takes advantage of e.g., different interest rates at different points of time, and the securities are sold as bundled products. However, the structure may be complex to the point where the underlying value of the mortgage or asset can no longer be properly estimated. A tranche may be based on certain assumptions about the development in the markets and as such,

Volatility

Volatility is related to risk and uncertainty about potential changes in the value of an asset (security). If volatility is high, there is a risk that prices can change quickly, and dramatically. It can be a statistical measure of the spread of returns for a security or market, or a variable:

1. A statistical measure of the dispersion of returns for a given security or market index. Volatility can either be measured by using the standard deviation or variance between returns from that same security or market index. Commonly, the higher the volatility, the riskier the security.
2. A variable in option pricing formulas showing the extent to which the return of the underlying asset will fluctuate between now and the option's expiration. Volatility, as expressed as a percentage coefficient within option-pricing formulas, arises from daily trading activities. How volatility is measured will affect the value of the coefficient used.

Willingness to pay (WTP)

Willingness to pay is the basic concept in the economic theory of value, which is based on individual preferences for goods and services. It reveals the (relative) importance of one good (or service) against another measured in monetary terms. WTP by all consumers in a market forms the basis for the demand of the good involved. It is notable that since payments also are influenced by income, WTP is influenced by both preferences and ability to pay – see also Appendix 2.

Appendix 2: Monetary valuation and its critique[1]

(by Arild Vatn)

Monetary valuation of nature values has become a highly emphasized and debated topic. While nature delivers a series of services to humanity –that are moreover of tremendous value and importance – it is argued that these values are not taken sufficiently into account because they are not economically valued. Many ecosystem services (ES) are not demonstrated, respectively captured in monetary terms – e.g., TEEB (2010).

There are at least three – partly interlinked – arguments supporting monetary valuation that are of great importance. First, *rational allocation* in a society demands that costs and benefits are compared – i.e., that decisions are made on the basis of cost-benefit analyses (CBA). We should not do anything where costs exceed benefits; that is to waste resources. To make such a comparison possible, monetization is necessary – bringing all values into a comparable form. Second, pricing *makes the values involved visible* in a form that makes them countable in present market oriented societies. Monetizing implies to speak the strongest and most common language of such societies and is necessary to ensure backing by policy makers. Finally, monetary assessments *communicates well to business and finance*. They enable these actors to understand what is at stake and may even make them interested in ES investments. Biodiversity and ES deliveries are under tremendous pressure and we need to expand the financial basis to succeed in protecting them.

Several methods have been developed to assess monetary values – willingness to pay – of ES. It is common to divide between revealed and stated preferences. The former relates to prices as observed in markets – either directly or indirectly. Direct observation means simply prices of ES that are traded. While many are not, assessments may still be made based in information from markets. Here one often refers to 'surrogate markets'. In the case of *hedonic pricing*, monetized values of ES are obtained through observing variations in tradable goods like homes that follow from variations in environmental qualities – e.g., local air quality, landscape values, and noise. In the case of the travel cost method, monetary values of certain ES are obtained by estimating the costs people pay to 'consume them' – e.g., the costs for visiting a national park, a scenic view etc. (Hanley et al. 2013)

Monetary values of many ES are not possible to obtain this way or the quality will be low. To handle this, methods for stated preferences have been developed. These 'simulated markets' may be categorized in two (ibid.). First, we have contingent valuation, where people offer their willingness to pay (WTP) for defined ES. The service is described and a bidding procedure is used to estimate the WTP. Second, monetary values can be assessed through choice experiments. Here respondents are presented with a menu of alternative ES states. The argument for the latter method is that people are more used to choose between alternatives than stating their WTP. Moreover, in choice experiments, preferences for various components of ES can be examined more systematically.

There has been substantial debate about the sensibility of basing decisions regarding nature values on the above kind of monetary assessments. We will here present three of the most important – the information problem, the issue of plural values and the question regarding of what kind of values should form the basis for decision making regarding nature values.

The *information problem* can be divided in two. First, we have the loss of information. Nature is complex and can mainly be understood as a set of interlinked processes – e.g., Steffen et

[1] References are found in the overall Reference list placed before Appendices

al. (2005). Ecosystems or their services are not easy to make into commodities that can be delineated and their value measured in a single term. They are rather characterized by opaqueness, uncertainty, thresholds etc. – all kinds of dynamics that make monetization result in loss of crucial information (Vatn and Bromley 1994). In cases like the evaluation of thresholds, monetary assessments fails fundamentally as decision support. Second, we have the challenge of ensuring that those that do monetary evaluations – whether through markets directly, or more typically through simulated markets like in willingness to pay studies – are well enough informed. Given the former, one may say that one is never 'well enough informed'. Anyway, decisions must be made on the basis of the best of exiting information. That seems to demand processes of deliberation between those doing the valuation and experts. That is possible to make in willingness to pay studies, but is rarely done. In real markets this is a tremendous problem.

The issue of *plural values* is linked to the above since complexity itself implies plural, often non-commensurable values (Vatn 2009). Several issues are of importance here. First, people may not be able to transform a multi-dimensional problem into one measurement scale – e.g., the species characteristics, nutrient cycling, aesthetics, climate and water regulation that are linked to a piece of land. It may be technically consistent to do so, but people are just not trained to think about all involved dimensions in monetary terms – to compare across scales – e.g., Tversky, Slovic and Kahneman (1990). Second, the different value dimensions may be incommensurable. They can simply not be measured in one scale. This characterization of the world is an issue strongly emphasized not least in the philosophy literature – e.g., Chang (1997); O'Neill (1993); O'Neill et al. (2008) – and is also linked to the above argument regarding information loss. A specific aspect here regards the dimension of ethics. The literature emphasizes 'nature's intrinsic value' – e.g., the intrinsic value of a species/its 'inherent right' to exist (Holland 1997). Intrinsic values can hardly be an inherent feature of nature. It must be something that humans assign to it. Nevertheless, this value is not an anthropocentric one. It is not about a service that nature offers to humans. Another aspect of this regards that we have nature in common. This implies that on person's preferences/use may change other people's opportunities. This brings in yet another ethical issue – should people 'be free' to hold whatever preferences regarding common values (Vatn 2000). This is an ethical question that cannot be treated within e.g., the utilitarian framework of CBA.

This takes us to the last issue – what *kind of values* should decisions over nature values be based on. This question makes only sense given an understanding where preferences are accepted to be of different types. It is standard to distinguish between consumer and citizen preferences – e.g., Sagoff (1988) – or between individual and social preferences (Hodgson 2007). The perspective here is that people hold different preferences dependent on what role they have. The role of the consumer emphasizes individual preference – typically understood within the perspective of preference utilitarianism as underlying CBA. It may be argued that preference utilitarianism is a weak basis for environmental decision-making – e.g., O'Neill et al. (2008); Spash (2000; 2009); Vatn (2010). These decisions should rather be based on citizens' deliberation and on social as opposed to individual preferences. A social preference is a preference that individuals hold about what a better state for society not for the individual per se. The literature confirms that the distinction between individual and social preferences is meaningful to people (e.g., Hodgson 2007; Soma and Vatn 2010). It turns our attention towards deliberation (e.g., Dryzek 2000), using dialogue to assess the better arguments concerning the future for humanity and its environments. In relation to that, it is notable that institutional contexts are not neutral regarding value expressions. Shifting decisions from one context – e.g., the deliberative forum –

to another – e.g., CBA/the market – influences how we think about the values involved and (can) express ourselves (O'Neill et al. 2008; Vatn 2010).

The above does not imply that monetary values should be abandoned from all kinds of nature values. Especially among the sub-group ecosystem services, there are several values that have been made into commodities – e.g., food and timber – and where markets and pricing clearly has a role to play. It is, however, notable that in the case of food production, values beyond the commodity like milk, cheese, grain etc., are protected through separate policy measures. This regards e.g., food safety, food security, landscape amenities. According to MEA (2005) ES may be divided in provisioning, regulating, supporting and cultural services. While it is easier to commoditize provisioning services like food than let us say supporting ones like nutrient cycling, we note that land use policies illustrate well how difficult it is to use markets as the only institution to allocation even of these services. This does not imply that payments cannot be used to ensure e.g., the protection or delivery of landscape services as also emphasized in the main text.

Appendix 3: Further documentation regarding unintended effects (Chapter 5)

Appendix 3.1 The Achi targets

(by Graciela Rusch)

Aichi Targets that address explicitly policy instruments and/or conservation goals that are more appropriate to be targeted by economic instruments, and where targeting of these instruments can have unintended effects:

Target 3: By 2020, incentives, including subsidies, harmful to biodiversity are eliminated, phased out or reformed in order to minimize or avoid negative impacts.

Target 5: By 2020, the rate of loss of all natural habitats, including forests, is at least halved and where feasible brought close to zero, and degradation and fragmentation is significantly reduced.

Target 7: By 2020, areas under agriculture, aquaculture and forestry are managed sustainably, ensuring conservation of biodiversity.

Target 11: By 2020, at least 17 per cent of terrestrial and inland water, and 10 per cent of coastal and marine areas, especially areas of particular importance for biodiversity and ecosystem services, are conserved through effectively and equitably managed, ecologically representative and well connected systems of protected areas and other effective area-based conservation measures, and integrated into the wider landscapes and seascapes

Target 13: By 2020, the genetic diversity of cultivated plants and farmed and domesticated animals and of wild relatives, including other socio-economically as well as culturally valuable species, is maintained, and strategies have been developed and implemented for minimizing genetic erosion and safeguarding their genetic diversity.

Target 14: By 2020, ecosystems that provide essential services, including services related to water, and contribute to health, livelihoods and well-being, are restored and safeguarded

Target 15: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification.

Appendix 3.2 Biodiversity and ecosystem functions
(by Graciela Rusch)

Table A3.1: Examples of biodiversity features that correspond with ecosystem function. * Reviewed in de Bello et al. 2010. The degree of correspondence reflects different biological characteristics that are associated with ecological functions, and of the level of current knowledge about the linkages between biodiversity structures and ecological function. The material presented in the table are a few examples of different functions, not a review.

Biodiversity (supporting)	**Ecosystem function 1**	**Ecosystem function2**	**Ecosystem function3**
Leaf properties in terrestrial plants (chemical composition and structure)	Organic matter decomposition, mineralization, nutrient circulation (177 cases *, Cornwell et al. 2008)	Herbivory control – Plant-animal interactions (24 cases*)	Primary productivity, carbon uptake, fluxes (Milcu et al. 2014)
Body size, burrowing activity, life-history characteristics of soil macrofauna	Organic matter decomposition, mineralization, nutrient circulation (55 cases *)	Surface water flow/run-off (12 cases *).	Herbivory control (1 case*, body size)
Canopy density, size, Leaf Area, growth form composition (e.g. woody vs grasses, annuals vs perennial plants)	Rainfall interception, water flow/run-off (6 cases *), Miranda et al. 2013)	Infiltration/maintenance soil humidity (6 cases*)	Evapotranspiration, water cycle regulation (23 cases reviewed in de Bello et al. 2010).
Macrophytes (leaf properties, growth rates, leaf area index)	Nutrient/sediment retention (12 cases *, Niemeyer et al. 2014).		
Fish feeding habits	Multiple effects on nutrient/sediment retention (6 cases *).		
Allometric relationships in trophic chains	Soil fertility (Mulder et al. 2013).		
Taxonomic diversity terrestrial plants	Stabilization of biomass production (Ospina et al. 2012)	Biomass production/Primary productivity (Vila et al. 2007, Gamfeldt et al. 2013)	Soil carbon storage (Gamfeldt et al. 2013)
Fructivorous birds abundance	Seed dispersal of tropical trees (Pejchar et al. 2008, Reid et al. 2014)		

Appendix 3.3 Ecosystem function scale effects of adoption of conservation practices

(by David N. Barton)

To design optimum incentives for landuse management we need to understand the complexities of the biodiversity-ecosystem function relationship. For instance, in a cost-benefit analysis, Griffiths and colleagues (2008) use a biodiversity – pest control function and show three examples of possible relationships of conservation for bio-control strategies and the scale of adoption of that strategy. Their three models provide a useful illustration of (1) expected linear effects of adoption scale (2) unexpected negative effects of adoption scale and (3) unexpected positive effects of adoption scale of biocontrol technology. It can be easily adapted to also illustrate how the benefits of different types of PES instruments can depend both on the scale of adoption of PES and on the scale of adjacent landuses.

Figure A3.1 is adapted from Griffiths et al. (2008). In panel A, PES adoption is assumed to have a linear effect on benefits with no landscape context constraints. We then use their reasoning adapted to two types of PES. In panel B the effects of PES protecting a forest patch derive largely from redistribution of natural enemies to adjacent cropland; for the first fields planted with crops natural enemies are drawn from the adjacent forest habitat boosting pest control, but for subsequent multiple fields the effect diminishes as the predator population from the PES-protected forest patch is insufficient to cover new area. This is a "saturation effect" of landuse in the neighborhood of the PES contract. In scenario C effects on cropland of the initial forest regeneration PES contracts in a largely agricultural landscape are negligible, but then increase with the scale of conversion of cropland to forest through PES regeneration contracts. This occurs if forest regeneration alters natural-enemy richness and abundance and their numbers increase with increasing relative forest habitat area and connectivity to natural forests. This is referred to as a 'transition effect' (Griffiths et al., 2008). The functional form of the 'saturation' and 'transition' effects drawn in Figure A3.1 are hypotheses which will depend on the changing configuration of the landscape mosaic. The neighborhood or spillover effect referred to here are due to ecological function, rather than mechanisms through social network or the market. What policy lessons can be learned for economic incentives? Pilot testing of PES and monitoring of effects needs to take place over more than a few properties and over longer time in order to capture landscape scale effects. The combination of landuse adoption scale and landscape properties is specific to landscape mosaics and bioregions, so predictions of effectiveness are not necessarily transferable, even within the same country.

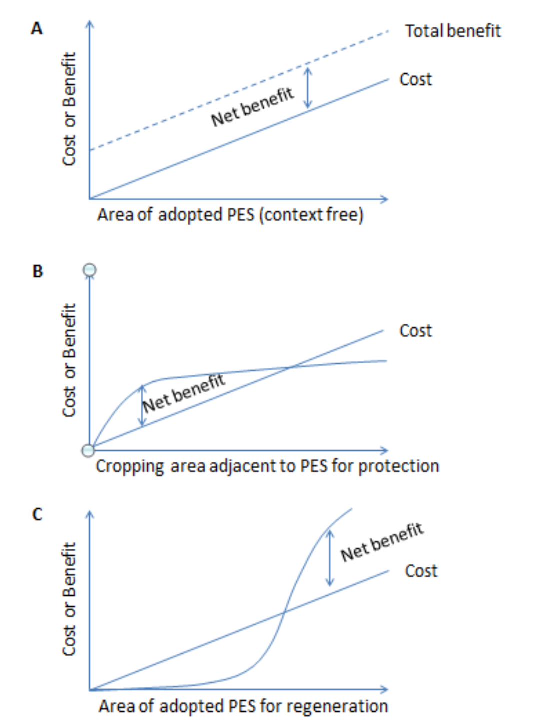

Figure A3.1 Scale- and land-use mosaic dependency of the ecological functions underpinning pest control, and its consequences on the cost-benefit outcome of PES adoption. (A) context and constraint free, (B) with a saturation effect of pest control with increasing crop area, (C) with non-linear increase in pest control due to habitat cohesion with increasing area of restored forest cover. Source: adapted from Griffiths et al. (2008)

Appendix 3.4: Unintended effects of economic instruments due to motivational crowding

(by David N. Barton)

In this appendix we discuss the theory and conceptual framework for unintended effects of economic instruments due to the group of phenomena called 'motivational crowding'. Motivational crowding refers to the hypotheses that economic incentives may "crowd-out" intrinsic motivations (Muradian et al., 2013; Rode et al., 2013), in the particular case of this report land users' moral commitment towards biodiversity conservation.

For the purposes of this report the intended effect of an economic incentive refers to the expected change in landusers' behaviour caused by an incentive that changes the private landusers benefit-cost ratio of landuse change. The positive and negative incentives for landuse change due to different instruments as discussed by (Pannell, 2008) in his public-private benefits framework (PPBF). Positive and negative incentives in this framework refer to voluntary (e.g. PES), coercive (e.g. public protection) as well as informational (e.g. extension services).

The "expected incentive effects" refers to economic instruments involving a monetary transfer. In the PPBF a positive monetary incentive slightly higher than the net private cost of a potential landuse change is expected to lead to actual changed landuse (e.g. regeneration of abandoned pasture). A negative monetary incentive slightly higher than the potential net private benefits of landuse change is expected to discourage landuse change (e.g. forest clearing for monocultures).

Motivation crowding suggests that intrinsic motivations for biodiversity conservation[2] are not necessarily independent from the expected monetary incentive effect described above. Instead they may counteract (" crowd-out") or complement ("crowd-in") instrinsic motivation for biodiversity conservation, depending on the context (Rode et al. 2013). This is a problem when voluntary conservation motivations are crowded out by economic incentives for conservation, but do not recover once incentives stop.

3.4.1 What is motivation crowding?

Rode et al. (2013) conduct a review of psychological mechanisms for motivation crowding and discuss whether the different mechanisms are relevant for positive or negative incentives. We suggest that Rode et al. (2013) hypotheses regarding which psychological mechanisms seem relevant for different incentives, should distinguish between monetary transfers that take place for voluntary versus coercive instruments. Furthermore, we suggest that the different mechanisms of motivation crowding may depend on other contextual factors such as the potential landuse change situation (e.g. degeneration versus recovery of forest) and whether the crowding applies to a non-participant, participant or ex-participant landuser vis-à-vis the incentive. These aspects are presented as a series of questions in Table A3.2 adapted from Rode and colleagues.

[2] For the purpose of simplfying the presentation we define biodiversity conservation as including measures both to avoid biodiversity loss and to restore biodiversity through active management

Table A3.2 Psychological mechanisms and context variables explaining motivation crowding

Effect:	**Psychological mechanism**	**Explanation**	**Coercive or voluntary incentives for which this mechanisms seems relevant**	**Economic rationale for monetary transaction (avoided action, proaction)**	**Relevance for participants, non-participants, ex-participants**
Crowding-out	Control aversion	Individuals with a sense of autonomy and self-determination dislike feeling controlled.	Coercive incentives only?	Avoided action ?	Participants
	Frustration	Individuals are frustrated when they perceive regulations as a signal of distrust that they will do the right thing.	Coervice incentives only?	Avoided action & proaction?	Participants
	Reduce Image Motivation	Others cannot distinguish if one undertakes a social activity voluntarily or by pressure.	Coercive and voluntary incentives	Avoided action & proaction?	Participants
	Frame-shifting	An individual's attention is shifted towards a focus on economic reasoning (short-term).	Non-voluntary and voluntary Depends on type of monetary transaction?	Avoided action & proaction?	Participants
	Release from moral responsibility	Allowing monetary payments to compensate for environmental harm releases individuals from feelings of responsibility and guilt.	Non-voluntary and voluntary Depends on type of monetary transaction?	Avoided action (negative incentives)	Participants
	Changes in values or mindsets	The focus on economic reasoning affects attitudes and mindsets in the longer term.	Non-voluntary and voluntary . Depends on type of monetary transaction?	Avoided action & proaction?	Participants Ex-participants; Non-participants
Crowding-in	Enhanced self-esteem through social recognition	Individuals perceive rewards as supporting and acknowledging their behaviour.	Voluntary. Rewards only?	Avoided action & proaction	Participants
	Prescriptive effect	Individuals receive a normative signal of what constitutes desirable societal action.	Voluntary. Depends on type of monetary transaction?	Avoided action & proaction	Participants Ex participants Non-participants
	Reducing pressure by forcing non-moral individuals to compliance	Intrinsically motivated individuals can more easily act upon their motivation when not facing the bad example or even "exploitation" of non-moral individuals.	Voluntary. Depends on type of monetary transaction?	Proaction only	Non-participants

Source: Abbreviated and adapted from Rode et al. (2013).

3.4.2 Unexpected motivational effects on non-participant, participants, and ex participants

The motivational crowding literature has largely referred to the motivations of people who are directly targeted – voluntarily or by coercion – by an incentive. The idea of motivational crowding can be broadened to cover 'motivational exclusion' and effects on non-participants. Primmer et al. (2014) refer in addition to a 'staying out' effect. In their study this involved landowners who decided to avoid forest conservation compensation altogether.

For completeness, we should also consider a third type of landuser; e.g. the ex-participant of voluntary economic instruments. Longer term motivational crowding effects such as changes in mindsets and prescriptive behavior may continue to act also after the incentive has been removed (Zapata et al., forthcoming).

In Table A3.2 we speculate that motivational crowding mechanisms apply differently across these three types of landusers in the context of incentives for biodiversity conservation in forests. For example, most crowding out effects apply to participants motivations only, while 'changes in values or minds' will apply to ex-participants and may apply to non-participants in the longer term through learning in social networks. For crowding-in, we would expect that enhanced self-esteem comes only to participants, while the 'prescriptive' effect on norms applies equally to participants, ex-participants and non-participants.

3.4.3 Motivational crowding in coercive versus voluntary contexts

We also observe that not all of the psychological mechanisms listed by Rode and colleagues apply exclusively to economic instruments or their monetary transactions rules (Table A3.2). The mechanisms of 'control aversion', 'frustration', and 'reduce image motivation' should be equally relevant for coercive instruments without monetary transactions, such as public protected areas. Coercive rules are also an integral part of economic instruments - e.g. monitoring, enforcement and verification rules laid down in contracts. Furthermore, there is reason to question whether the psychological mechanisms work in the same way when the 'coercion' is accepted as part of a voluntarily contract.

On the other hand 'frame-shifting', 'release from moral responsibility' and 'changes in values or mindsets' refer specifically to effects of a monetary transaction.

Regarding 'crowding-in' effects, they should all refer to voluntary policy instruments by definition. If it was a non-voluntary instrument there would be no non-participants to crowd-in. We think 'enhanced self-esteem through social recognition' would apply only to those monetary transactions perceived as rewards (as distinct from compensation or payments). 'Prescriptive effects' would apply widely to both actions and avoided action contexts, and to participants, ex-participants and non-participants of economic instrument schemes. The prescriptive effect should depend on the economic rationale of monetary transaction (e.g. compensation of actions undertaken versus payment for non-action). 'Reducing pressure by forcing non-moral individuals to compliance' would seem to apply particularly to non-participants whose intrinsic motivations to take biodiversity conserving actions are strengthened by 'staying out' (Primmer et al. 2014).

3.4.4 Economic rationale for the monetary transaction

Furthermore, it seems relevant to ask whether the psychological mechanisms of motivational crowding that take place for monetary transfers in particular, could vary depending on the economic rationale of the transfer:

- liability charge (tax, offset), (negative incentives - avoided action)
- compensation for foregone opportunities, (negative incentives - avoided action)

- compensation for costs of actions undertaken, (positive incentives - proaction)
- a payment for the value of services rendered, (positive and negative incentives)
- or a social reward (positive and negative incentives)

The rationale of the monetary transfer is also conditional on the particular landuse change situation - whether the incentive encourages or discourages landuse change.

3.4.5 Size of payment relative to service supplied/demand

Rode et al (2013) cite Gneezy and Rustichini (2000) regarding the importance of the size of payment for motivation crowding; "pay enough or don't pay at all". In an example from the demand side of ecosystem services, Barton et al. (2009) found in a choice experiment of the willingness to pay additional sewage fees for good ecological status in water bodies in Norway, that households experience negative utility from payments for small improvements relative a status quo with no improvement. In this case we could add, "pay enough to get a significant improvement or don't pay at all."

3.4.6 Landuse change context of incentives and their normative – expected - effects

In Table A3.2 we propose that the mechanisms of motivation crowding are also specific to the landuse change context. In the public private benefit framework (PPBF) Pannell (2008) defines the normative use of positive and negative incentives as being conditional on the ratio of private to public net benefits of a potential or actual landuse change (Figure A3.2).

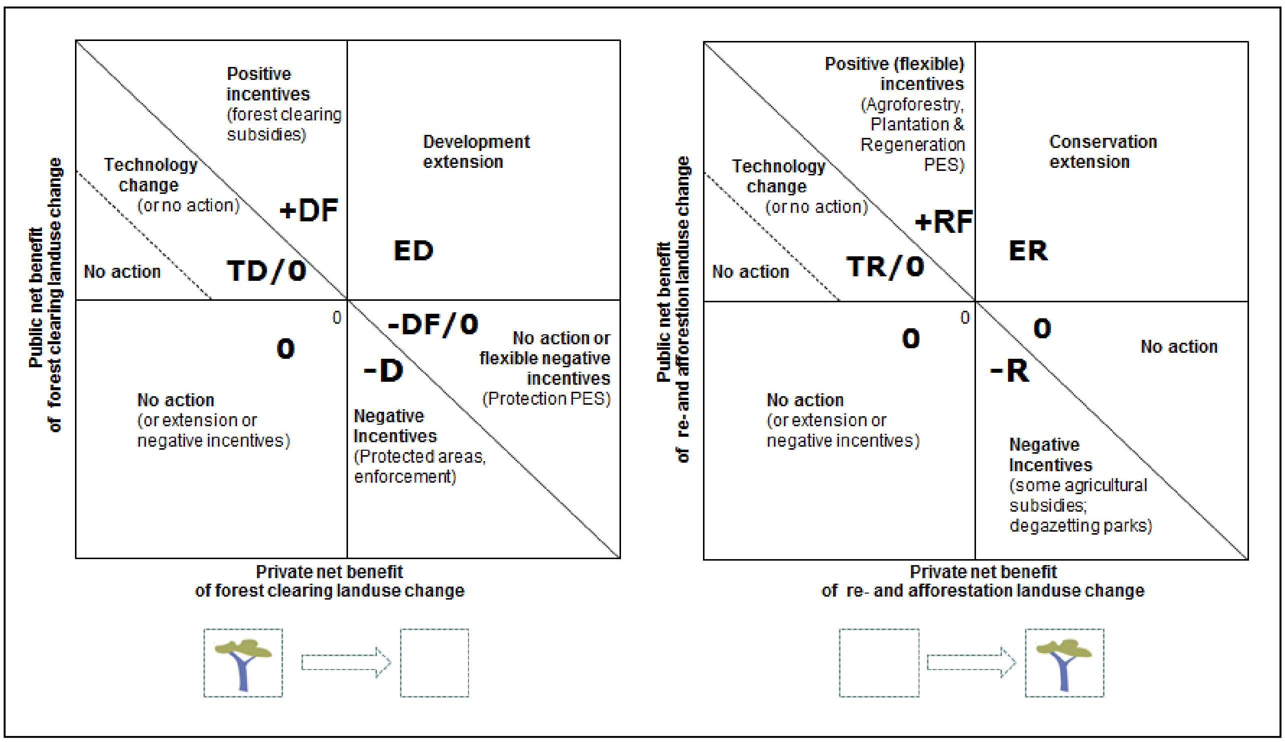

Figure A3.2. The public-private benefits framework applied to forest clearing (degradation) and re- and afforestation (recovery). Source: Barton and Adamowicz (2013) based on Pannell (2008). The expected effect of positive (+) and negative (-) incentives is to encourage or discourage landuse change. Motivational crowding effects will mean that instruments cannot be targeted as suggested by private net benefit ratios of landuse change used in the PPBF framework.

Pannell (2008) proposes how the normative targeting of instruments may be conditional on the landscape characteristics of any particular location. The normative expected incentive effect is to encourage or discourage landuse change depending on the size of the incentive. In figure A3.2 Barton and Adamowicz (2013a) adapted Pannell's(2008) original PPBF to distinguish between landuse change contexts of forest degradation versus forest recovery.

Positive and negative incentives are specific to the forest transition stage a landscape is in. Specific negative and positive incentives have been developed to address deforestation versus reforestation stages. For example, PES to discourage deforestation (negative incentive denoted as -D); PES to encourage reforestation(positive incentive denoted as +R); forest road building subsidies that may both encourage forest clearing (positive incentive encouraging deforestation denoted as +D) and taxes on plantation wood or agricultural subsidies that also discourage reforestation (negative incentive discouraging reforestation denoted as –R).

The economic rationale of the monetary transaction in the instruments shown in the PPBF varies according to context. For example, different prescriptive signals regarding norms may be sent depending on the different economic rationales of the instrument

- *action versus no action* regarding landuse change
- avoiding a *loss* (of forest cover) *versus* obtaining a *gain* (in forest cover)
- *compensation* of foregone net private benefits of no action (horizontal axis), *versus payment* of net public benefits of a landuse change (vertical axis)

We think that the context dependence of motivational crowding in relation to different landuse change settings, the chosen instrument and its economic rationale merit further research.

Appendix 3.5 Unintended effects through social networks

(by David N. Barton)

"Where networks are important, generalized statements about policy effectiveness across different policy domains lack validity because of the different natures of the different types of network which predominate in the social and economic worlds. The effectiveness of a policy will be contingent on the type of network upon which it is being enacted." (p. 182). (Ormerod, 2012).

3.5.1 Social networks

Random networks have no predetermined structure and are familiar in epidemiology. Small-world networks have structure which corresponds to the social situation of 'overlapping friends of friends' structures. Scale-free networks have many forms, but are generally recognizable by a few agents acting as 'hubs' with many connections, in contrast to most other agents in the network who are linked to a small number of others (Ormerod, 2012).

Scale-free networks with hubs of agents may be relevant for certain PES markets with a few highly mobile intermediaries with many landowners as clients, but where landowners themselves participate otherwise only in small-world local community networks.

Technological innovation of the internet makes an entirely different model of decision making feasible, in which people take into account directly the choices and opinions of others, with agents living far from (p.197) (Ormerod, 2012). With information about PES instruments available online, and perhaps in future online information available regarding currently participating properties, the potential for scale-free network effects on PES participation seem more likely.

3.5.2 Behavior in social networks

In order to answer whether network effects are relevant explanation for unexpected effects of economic instruments for conservation, we must ask whether a landuser facing the decision to participate in a PES scheme has similar characteristics to situations exhibiting copying behavior in social networks. Copying behavior may lead to unexpected effects relative to what is expected of the rational benefit-cost computing agent. In many situations agents are unable to compute optimal decisions, a position argued by Herbert Simon (1956). These situations involve (i) a large number of product/service choices, (ii) with characteristics that are complex and hard to evaluate, and (iii) where people are aware and concerned about what other people are doing. In these decision situation copying behavior – or social learning – is a common heuristic (Ormerod, 2012).

Copying behavior is a useful heuristic when a person faces choices in which she believes that the group – other people – have better information. A second motive for copying other people is a desire to conform. Copying behavior has also been observed especially for cultural goods (Ormerod, 2012). The magnitude and spatial extent of unexpected effects will depend on the type of network within which copying behavior takes place.

Appendix 3.6: Frameworks for unintended effects of instrument interactions

(by David N. Barton and Irene Ring)

This appendix discusses conceptual frameworks for identifying indirect effects of policy instrument interactions at agent and landscape level and the functional roles of these interactions. Sections 3.6.1-2 and 3.6.4 are extracts from Ring and Barton (forthcoming).

3.6.1 Characteristics of instrument interaction

How would we go about classifying different types of interactions between economic instruments and other instruments in the policy mix? Ring and Barton (forthcoming) compare terminologies of earlier proposed frameworks for policy-mix analysis (Flanagan et al., 2010; Gunningham and Sinclair, 1998, 1999), in a single framework for describing policy instrument interactions (Figure A3.2). Instrument interactions may be described using different 'geometries' which are configurations of interaction, being more or less direct, while functional roles describe the nature or value of the interaction relative to policy objectives. Furthermore, economic instruments have functional roles defined by 'action situations' (Ostrom, 2005). Action situations have 'dimensions' in terms of the (1) abstract 'policy space', (2) actors interacting at different levels of governance, (3) geographical locations in physical landscapes, and (4) occurrence in a particular time period, that can be related to the dimensions of interactions in the terminology of Flanagan et al. (2010). The interaction dimensions and geometries define functional roles of instruments. Instruments can have different functional roles at different stages of the policy cycle.

Empirical studies of interactions look for instrument-to-instrument causality in historical policy analysis, cross-referencing legal texts of different instruments; actors referring to multiple instrument rules that influence their decision-making; or overlapping jurisdictions of instruments on specific administrative areas, land uses and actors that can be located and mapped spatially. This framing of functional roles has commonalities with approaches that discuss factors explaining the 'fit' between institutions and their environments and evaluate horizontal and vertical 'policy interplay' (Urwin and Jordan, 2008; Young, 2002).

Ring and Schröter-Schlaack (2011) provide a number of examples of functional roles of economic instruments where they interact with informational or regulatory instruments. Figure A3.2 illustrates that instrument interactions can have some basic configurations (I-V) which are synergistic, complementary, path dependent, redundant or in conflict. We use 'complementary' in the sense that instruments do not interfere with one another in spatial targeting (or possibly that one unilaterally supports the other); while a 'synergistic' role is found when two instruments mutually reinforce one another. Functional roles are meant to describe the predominant role of an instrument in a particular period and a particular geographical area. An instrument can over time have several 'functional roles' in the zigzag sometimes observed in policy change.

Forms or geometries of interaction are increasingly more difficult to study empirically as we move from (I) single instruments acting directly on actors, to (V) multiple instruments acting indirectly through a socio-ecological system. The roles and basic configurations mentioned in Figure A3.3 are by way of example and restricted to two instruments or rules at a time – in empirical studies this is expected to be a multi-dimensional analysis. An example of complex instrument interactions through socio-ecological system are market price effects; large scale PES participation could lead to a rise in crop prices and pressure for further forest conversion to cropland. The 'scarcity slippage' that has been observed in some case studies (Alix-Garcia et al.,

2012) would be compounded by other credit, transport or labor policies that constrained the response of the crop market.

Furthermore, in Figure A3.3 interactions and roles of instruments are defined in terms of their effects on actors as 'targets' (of biodiversity conservation policy). The functional roles of economic instruments in a policy mix must also be seen in relation to different stakeholders' own goals for landuse.

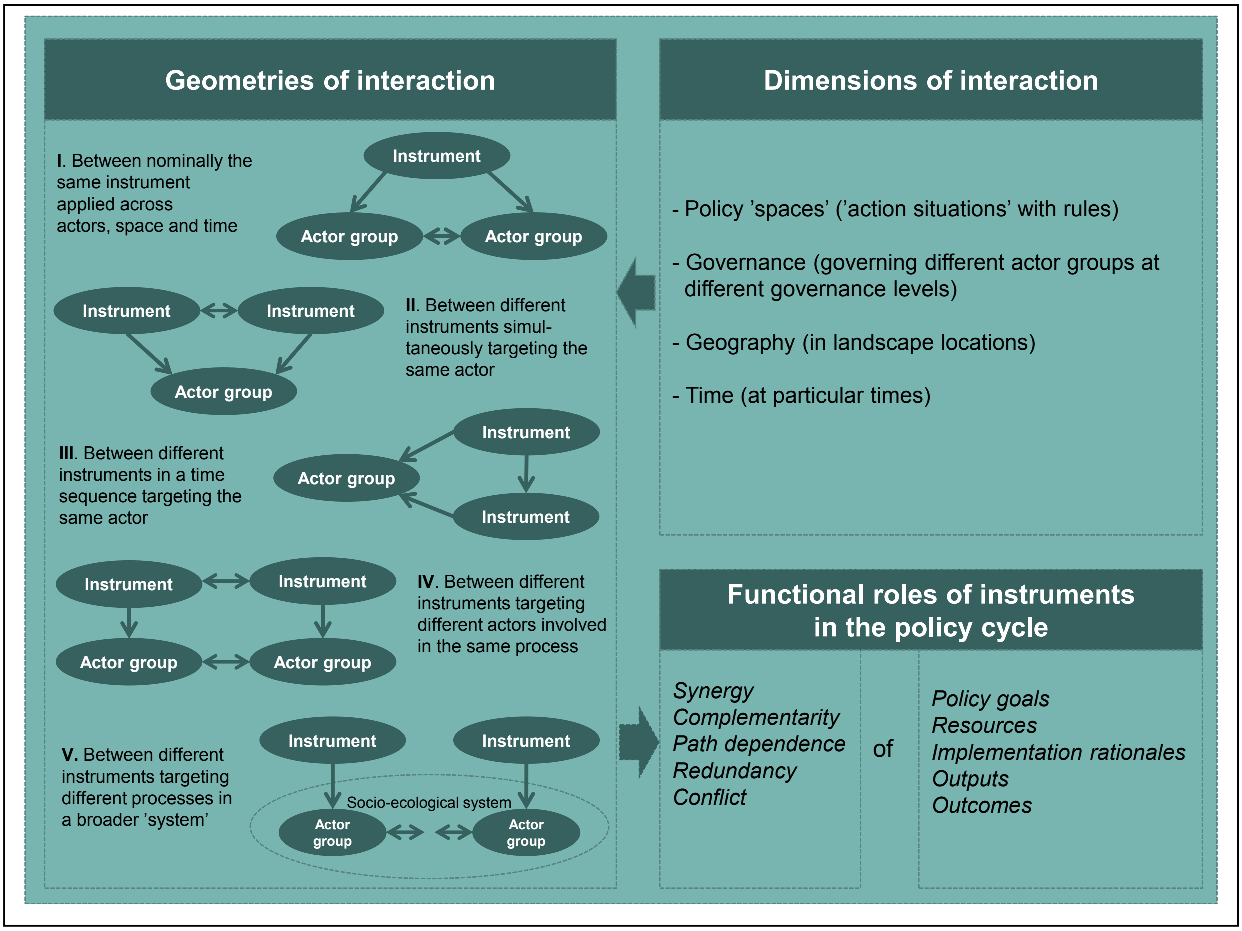

Figure A3.3. A framework for instrument interaction geometries and functional roles

Source: based on Flanagan et al. (2010), Gunningham and Sinclair (1998, 1999) and Ring and Schröter Schlaack (2011b).

3.6.2. Instrument interactions at landscape level

What are the most important characteristics of landscapes that determine the spatial distribution of conservation instruments? A number of studies have found that accessibility/distance and biological land-use capacity significantly explain spatial patterns of rents for land conversion and location of PES and protected areas (Andam et al., 2008; Joppa and Pfaff, 2010; Pfaff and Robalino, 2012; Pfaff et al., 2009; Robalino et al., 2008). For example, national parks are typically found on low productive land, far from markets (Joppa and Pfaff, 2009). A policyscape 'state space' as shown in Figure A3.3 can help researchers communicate with managers and policy makers about situations where multiple policies are implemented simultaneously on the same types of land. This in turn helps to spatially visualize functional overlaps of instruments in the landscape. Policyscape mapping can also help identify potential instrument conflicts at the property level and at landscape level. By evaluating spatial locations of conservation instruments in relation to opportunity costs of alternative land uses, policyscape analysis can also help managers formulate hypotheses about whether instruments can be expected to be effective and additional (because they have opportunity costs).

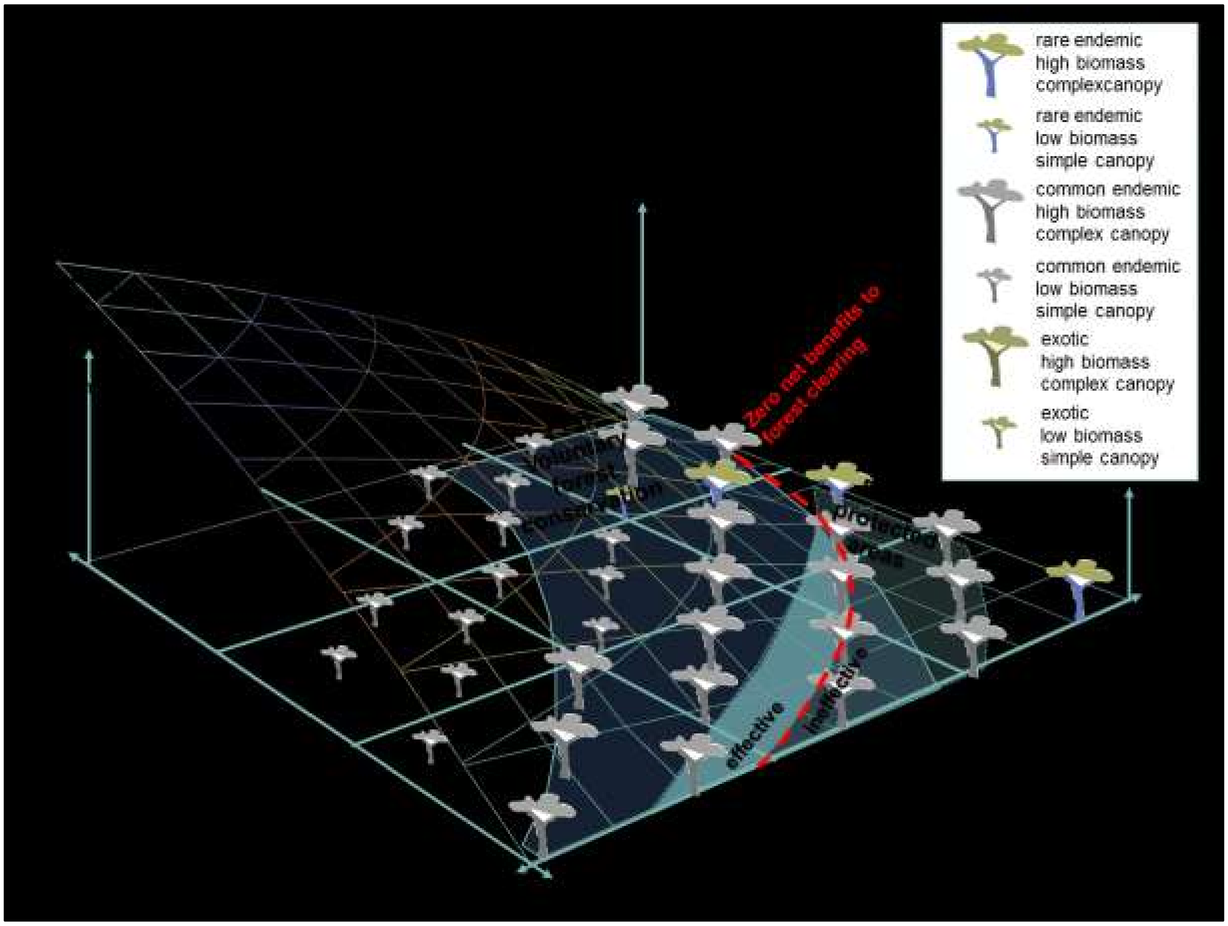

Figure A3.4. Policyscape – the location of policy instruments depends on landowners' perceptions of landscape characteristics and of returns to different land uses.

Source: Barton and Adamowicz (2013b).

The POLICYMIX project has developed an agent-based model for virtual experiments with spatial targeting of different economic instruments in a policyscape[3]. It demonstrates how the targeting of the policy mix determines landuse at landscape level over several forest transition stages. The tool is designed to raise awareness about unexpected landscape effects of using cost-benefit rules at property level for targeting of positive and negative conservation incentives using the PPBF (Figure A3.2) (Barton et al., 2014a; Barton et al., 2014e).

3.6.3. Instrument interactions at property level

[3] http://policymix.nina.no/Policymixtool/Instrumentspublic-privatebenefits.aspx).

What are the mechanisms at property level of interactions between economic instruments and other policy instruments in the policy mix? Pagiola et al. (2005) discuss the PES program and household characteristics that potentially influence eligibility, want and ability to participate in PES.

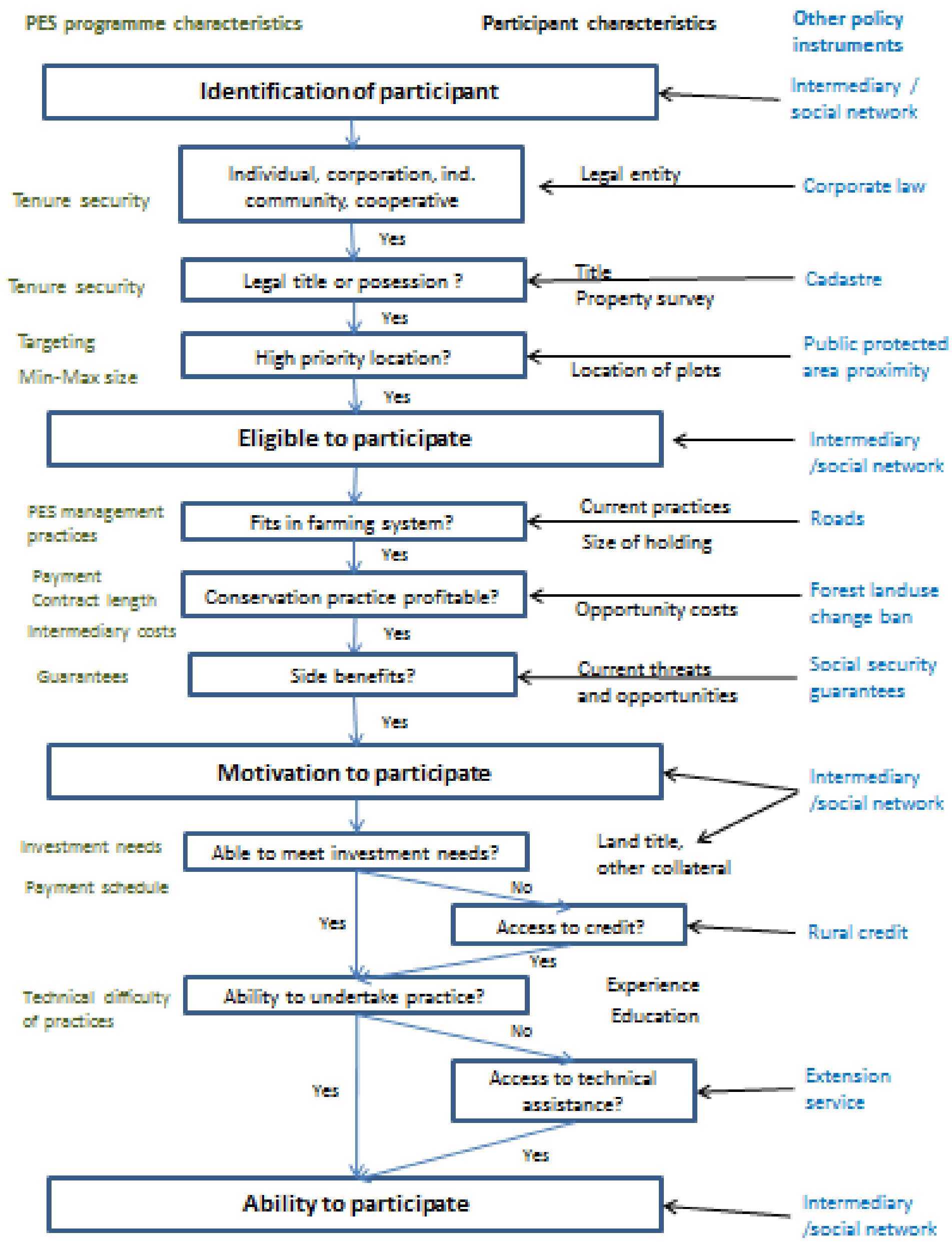

Figure A3.5. PES programme, participant characteristics and other policies potentially influencing participation and non-participation. Source: adapted from Pagiola et al. (2005)

Pagiola and colleagues' framework emphasizes the conditionality and context dependence of PES participation and provides a way of thinking about instrument interactions at property level. We make some modifications to their framework in figure A3.5. We provide examples of where different policy instruments may influence the propensity of landholders to participate (far right column). The list of other policies affecting PES eligibility, motivation and ability to participate is not exhaustive, but provides an illustration of the possible complexity of evaluating instrument interactions. It illustrates the potential for "unexpected" effects of PES, in the sense that policies affecting household characteristics may strengthen or weaken the likelihood of participation through eligibility, motivation or ability. We also rearranged the steps to be more in line with a specific PES programme (Costa Rica, where for example tenure security is also an eligibility criteria).

We reinterpret Pagiola and colleagues framework in terms of 'motivations' in order to tie the discussion to the issues of "motivational crowding". The conceptual model immediately suggests that motivational crowding could be classified into 'motivations for participation', and contrasted with 'motivations for implementation' once the contract has been signed.

We also highlight the linkages to social networks as potential explanations for increased/ decreased participation. The most obvious example for voluntary instruments such as PES is found in the role of the intermediary in identifying and providing information to potential participants, evaluating their eligibility, providing motivation and facilitating their ability to participate through e.g. credit and capacity-building. Intermediary agents also function as nodes in social learning networks. They may complement or substitute other networks for social learning about PES such as farm cooperatives (Bosselmann and Lund, 2013; Matulis, 2012).

3.6.4. Instrument interactions in landuse action situations

What are the 'parts' of an economic instrument such as PES? How can these characteristics be used to explain interactions with other instruments in the policy mix? Barton et al. (2014c) classify voluntary forest conservation and PES in Norway, Finland and Costa Rica using the 'rules-in-use' typology from the Institutional Analysis and Development (IAD) framework developed by Ostrom (2005). They argue that rules-in-use can be used as generic institutional variables to describe both PES and other instruments. Framing PES in terms of a mix of 'rules-in-use' provides the basis for arguing that PES is in itself a policy mix – a number of the rules defining PES are defined by other policy instruments. Furthermore, 'rules-in-use' provide a definition of which characteristics of PES instruments interact with similar characteristics of other instruments in the policy mix, providing a consistent structure for evaluating institutional interplay.

Figure A3.6 illustrates the 'rules-in-use' framework devised by Ostrom (2005). Here we use PES to briefly explain how 'rules-in-use' help define it as a policy mix in its own right (Barton et al., 2014b). We use examples for Costa Rica's PES to illustrate the different rules-in-use.

Scope rules define outcome variables and their ranges, such as the maintenance of forest cover as a proxy for a bundle of ecosystem services.

Choice rules define required, permitted, forbidden and guaranteed actions during the PES contract period. The land-use change ban for forest land imposed by the Costa Rican Forest Law is an example of a 'choice rule' which is a part of the mix of PES instrument rules-in-use.

Payoff rules identify the rewards and sanctions of outcomes of actions. Payoff rules encompass all incentives, rather than a narrow focus on payment conditions. This broadens the scope of costs and benefits of PES participation to include i.a. property tax exemption; the Forest Law's guarantee of public eviction of squatters; contract termination and prison sentences for deforestation with intent.

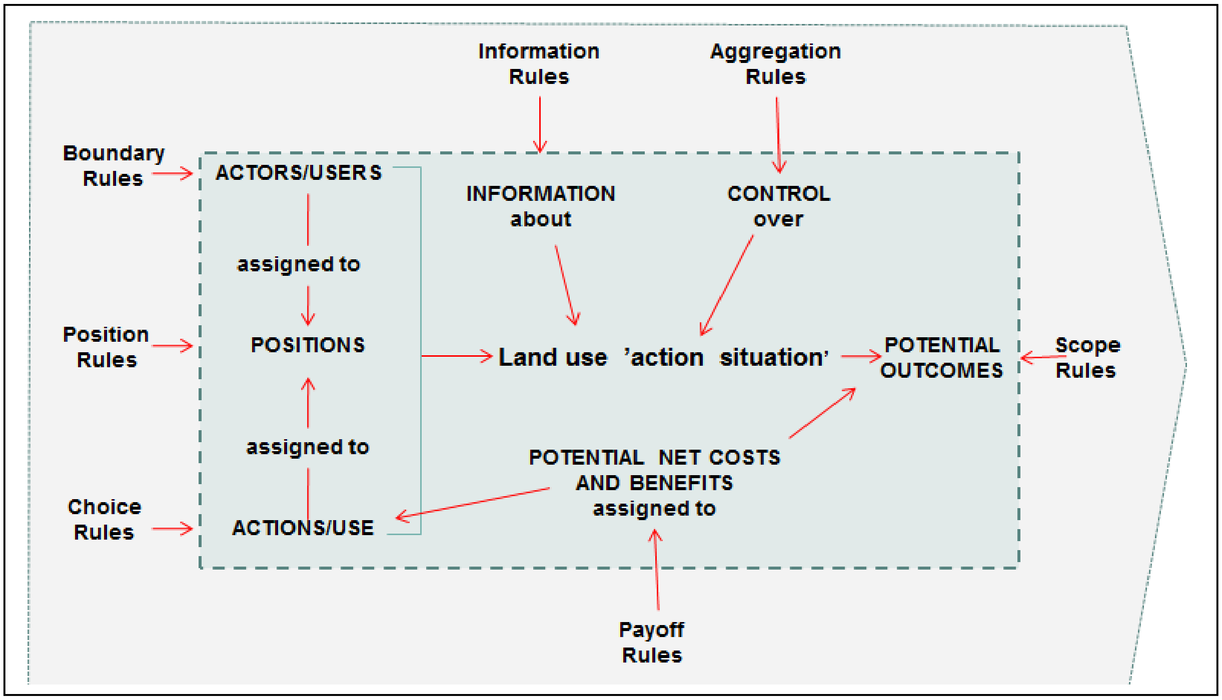

Figure A3.6. Using 'rules-in-use' to characterise payments for environmental services.

Source: Adapted from Ostrom (2005).

Boundary rules govern the entry, succession and exit of PES participation, such as forest characteristics eligible for participation, requirement of tenure rights, contract length and contract renewal criteria. As such, cadastral inconsistencies in property registers are a serious barrier to PES participation and determined by a host of historical land-use policies. Together, boundary, scope, choice and payoff rules capture the key dimensions of PES participant's contracts.

Position rules determine decision-making positions, such as the types and roles of intermediaries in reporting, monitoring and verification of PES contracts. Costa Rica limits transaction costs of intermediaries to 18 % (Porras et al., 2013), but it can be argued that this is possible because the intermediary 'forest regent' carries out almost all transactions including participant identification, recruiting, application, contracting, monitoring, reporting, disbursement and verification. As such position rules are potential proxies for both transaction costs and information asymmetries in PES.

Information rules govern information access and disclosure. In Costa Rica corporations may apply for PES, with owner structure anonymised thanks to a constitutional guarantee of equivalence between physical and legal entities. This constitutional guarantee makes it very hard to evaluate whether PES is targeted to individual small and medium landowners or large conglomerates (Porras et al., 2013).

Aggregation rules refer to collective voting rules and lack of agreement rules. While they have limited relevance for individual contracts, they characterise collective responsibilities in group-based PES contracts – once in place in Costa Rica, and are common in many PES regimes.

3.6.5. Instrument interactions in the policy cycle

A ‘policy cycle’ framework (Brewer and DeLeon, 1983; Kivimaa and Mickwitz, 2006) suggests that PES may interact with other instruments at the level of (i) policy goals (ii) resources (iii) implementation process (iv) outputs and (v) intermediate and final outcomes (Figure A3.7) (Barton et al., 2014d). We therefore use the policy cycle as a framework for characterising when/ where there are policy-interactions, and the functional role framework (above) to characterise the nature of the interaction.

Policy goals. In principle, conservation policy goals are based on identification of conservation objectives and ecosystem service needs. Policy goals for particular instruments such as PES are often more specific and short term such as the size of the budget allocated (resources), the number of hectares targeted for the programme per year (outputs), the types of landuses to be prioritised (intermediate outputs), or more rarely the distribution of costs of provision and benefits from ecosystem services across different stakeholders (final outcomes).

Resources. Concern the sources of financing of PES, whether private, public, domestic, foreign or donor-based. The extent to which available resources cover potential participants applying to the programme has sometimes been an evaluation criteria of PES.

Implementation processes as ‘action situations’. Implementation of PES in an ‘action situation’ takes place in a specific time period and place in the landscape involving a set of actors making a decision about landuse, employing the different types of ‘rules-in-use’. Actors’ perceptions of rules-in-use can be evaluated in terms of process legitimacy. The landuse actions themselves have costs. Learning the rules-in-use, and the number of rules actors must follow, are expected to determine transaction costs. Ostrom (2005) demonstrates how ‘rules-in-use’ categories can be applied at different constitutional levels. Similarly, the policy cycle is a conceptualisation of a more general level of analysis than the ‘implementation action situation’. However, the different ‘rules-in-use’ can also map out to the different steps of a policy instrument in a policy cycle.

Outputs. These are the simplest terms of ‘scope’ or effectiveness criteria for evaluating PES, including, for example, the number of hectares contracted in different contract modalities, amount of spending annually on PES, the number of female participants, or the number of small farms enrolled.

Outcomes. We define intermediate outcomes as different biophysical indicators of biodiversity conservation and ecosystem service provision. These may be explicitly or implicitly defined as goals of PES. The simplest indicators include hectares of different forest types, considered to be proxies for ecosystem services such as carbon storage or hydrological services. Final outcomes can be broadly characterized in terms of the distribution of private and public net benefits of landuse changes that are incentivized by PES(Pannell, 2008). One aim of ecosystem service valuation in the context of PES, is to compare costs of landuse change to willingness-to-pay in order to assess the economic efficiency of PES. Although ecosystem service delivery may be a stated policy goal, PES is seldom evaluated in these terms.

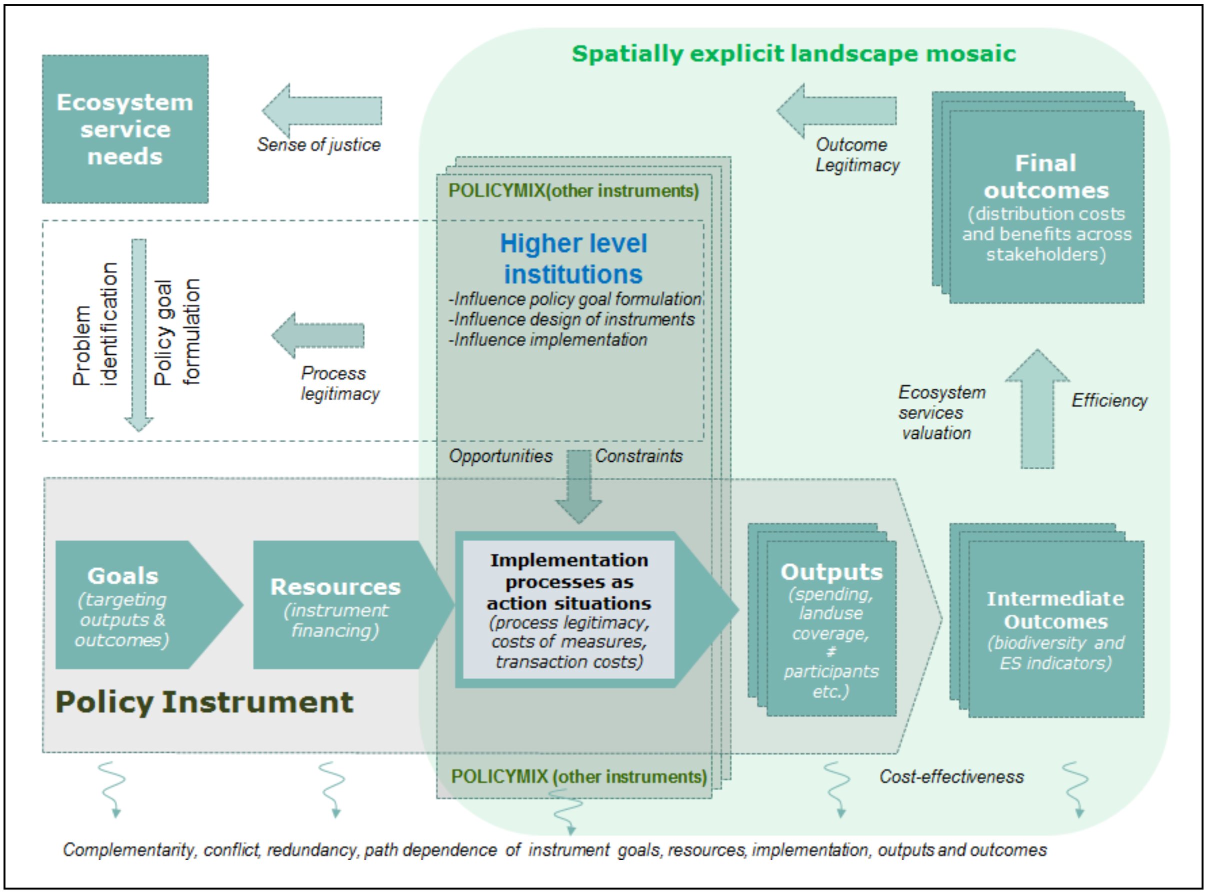

Figure A3.7. A policy cycle framework where a policy instrument is defined by its goals, resources, implementation, outputs and outcomes. Implementation processes are composed of a number of action situations which are governed by rules-in-use, defined by and nested within a policymix of instruments. Rules in use can have different functional roles in relation to one-another within a specific instrument such as PES. A policy instrument may also interact- have different functional roles – relative to other instruments in relation to goals, resources, outputs and outcomes.

Appendix 4: Social impacts of Costa Rica's PES – further documentation[4] (Section 5.2)

(by Ina Porras and David N. Barton)

In practice programme managers have tested a series of measures to promote participation of relatively vulnerable landholders and improve the social outcomes of the programme, rather than as "pro-poor" targeting. Here we describe these features in relation to the different stages of a PES contract planning and evaluation cycle (i) eligibility: size and tenure (ii) targeting PES pre-applications (iii) choice of payment modalities (iv) contract terms (including transaction costs, payment modalities, length) (v) terms of renewal (vi) and impact evaluation (Porras et al., 2013).

4.1 Eligibility criteria for property size and tenure

Only those with land can apply for PES. Tenure must be clear, with property titles or uncontested possession rights, and no cadastral inconsistencies in the property boundaries –which is a problem in some parts of the country where as much as 42% of properties and 70% of eligible properties have boundary inconsistencies (Porras et al. 2013). Property size is also important, requiring a minimum of 1 hectare for reforestation (plantation) and 2 hectares for protection and regeneration projects. Apart from indigenous communities, any private (e.g. not occupants of public lands) landowner can participate e.g. individuals, corporations, cooperatives and NGOs. A further eligibility criteria of relevance for the social impacts of corporations is that all social security obligations to employees must have been satisfied by the time of the pre-application.

4.2 Spatial targeting of pre-application

Costa Rica has since 2011 ranked PES per-applications according to a set of spatially mapped criteria, several of which have an explicit or implicit focus on social impacts. Indigenous territories, properties <50 hectares, farms within public protected areas awaiting expropriation, and areas with a social development index (SDI) below 40% are examples of social impact criteria used for prioritizing pre-applications. The SDI is a composite relative measure of development including health, education, participation and economic indicators calculated at the borough level.

4.3 Contract terms with a social impact profile

In general, intermediaries known as 'regentes forestales' fulfill the administrative requirements of the pre-application, monitoring and transfer of funds to the landowner. Intermediaries fees are generally capped at 18% of PES contract amounts, although abuses do take place. The PES Operations manual establishes that in the case of indigenous reserves the regional conservation area director may certify compliance of PES contracts free of charge, replacing the 'regente forestal' as intermediary and reducing transaction costs. The introduction of the 'agroforestry system' PES in 2002 with a minimum contract requirement of 350 trees has meant that smallholders can access PES independent of property size. Protection PES contract lengths were increased in 2010 from 5 years renewable to 10 years in order to reduce transaction costs, which proportionately is expected to benefit small holders more.

[4] References are found in the overall Reference list placed before Appendices

4.4 What have been the impacts of PES social impact design?

Between 1997 and 2012, FONAFIFO distributed approximately US$340 million as PES. The greatest part – and increasing – of these funds went to legal entities (i.e. corporations 49%), followed by individuals (31%), indigenous groups (13 %) and cooperatives (7%). See Figure A4.1.

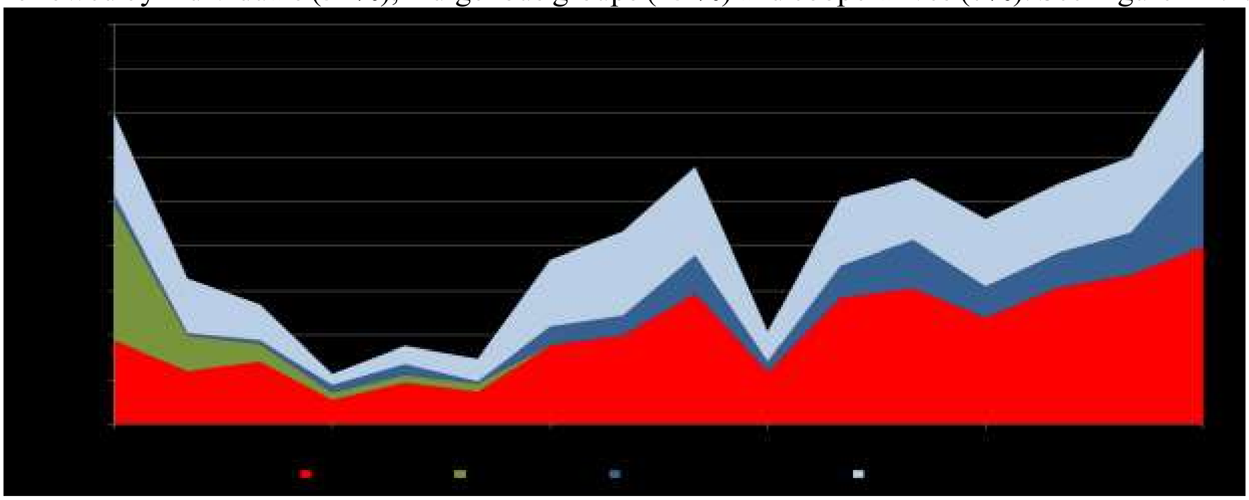

Figure A4.1. Timeline of funds by type of participant, 1997–2012. Source: Porras et al (2013), using data from FONAFIFO

In areas with low SDI receiving social priority[5] 35% of contracts (7% of budget) were with relatively small farms (<30 hectares) – providing an indicator of social benefit. But at the same time 29% of contracts (65% of the budget) went to relatively large farms (>100 hectares, whose owners are not likely to be poor. Porras et al. (2013) conclude that the SDI is spatially too coarse to target social impacts at household level, and recommend using property specific criteria. See Figure A4.2

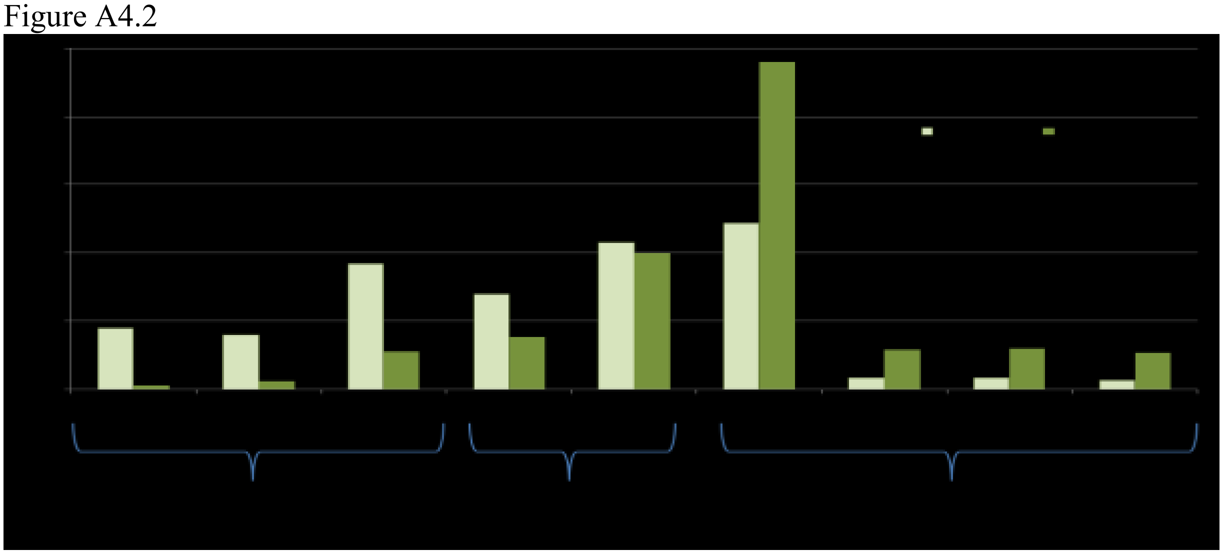

Figure A4.2. Distribution of PES contracts and budget by property size in areas with SDI<40. Source: Porras et al (2013) using data from FONAFIFO. Excludes group contracts and those with indigenous communities.

[5] Excluding group contracts (valid between 1997-2002) and those with indigenous groups.

With respect to priority criteria assigned to small properties, a recent study by Porras et al.(2014) looked at the distribution of PES contracts relative to property value. They found that smaller properties have higher per hectare prices, and relatively pricier areas are more likely to be fragmented into smaller plots. Attributing priority in the application process to properties of less than 50 hectares will not necessarily convey a measure of the landowners' relative vulnerability (see Table A4.1 for examples of heterogeneity of prices in 50 hectare properties).

Table A4.1. Example of prices per hectare at district level

DISTRICT	< 1 HA	1 TO <5 HA	5 TO < 10 HA	> 20 HA	DISTRICT AVERAGE	50 HECTARES PROPERTY VALUE
Santo Domingo, Heredia	-	92,625	86,400	68,200	70,700	3,410,000
Fortuna, Alajuela	112,255	131,614	133,980	39,700	121,287	1,985,000
Naranjito, Puntarenas	271,193	327,608	957,300	34,600	21,633	1,730,000
Sierpe, Puntarenas	10,940	31,437	19,289	11,900	48,079	595,000
Rivas, PZ San José	76,400	18,732	4880	8433	210,057	421,650
La Suiza, Cartago	47,300	23,439	1767	4867	18,408	243,350
Hojancha, Guanacaste	40,925	21,210	13,857	3425	30,432	171,250
Upala, Alajuela	81,020	68,655	6240	2500	34,377	125,000
La Virgen de *Sarapiquí*, Heredia	75,886	43,941	13,392	1414	19,172	70,700

Note: These values represent minimum levels, with outliers taken out to avoid overvaluation. Source: Porras et al (2014), using information from (RCG, 2008)

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