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Assessing the role of economic instruments in a policy mix for biodiversity conservation and ecosystem services provision: a review of some methodological challenges

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Abstract

In this paper we review a number of methodological challenges of evaluating and designing economic instruments aimed at biodiversity conservation and ecosystem services provision in the context of an existing policy mix. In the context of the EU 2010 goal of halting biodiversity loss, researchers have been called upon to evaluate the role of economic instruments for cost-effective decision-making, as well as non-market methods to assess their benefits. We argue that cost-effectiveness analysis (CEA) and non-market valuation (NMV) methods are necessary, but not sufficient, approaches to assessing the role of economic instruments in a policy mix. We review the principles of "social-ecological-systems"(SES) (Ostrom et al. 2007) and discuss how SES can complement economic cost and benefit assessment methods, in particular in policy design research. To illustrate our conceptual comparison of assessment methodologies, we look at two examples of economic instruments at different government levels – payments for ecosystem services (PES) at farm level and ecological fiscal transfers to municipal /county government. What conceptual problems are introduced when evaluating policies in an instrument mix? How can the SES framework complement CEA and NMV in policy assessment and design? We draw on experiences from Brazil and Costa Rica to exemplify these questions. We conclude with some research questions.

Keywords: biodiversity, ecosystem services, policy mix, social ecological systems, payments for environmental services, ecological fiscal transfers

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Background

In March 2007 at a meeting in Potsdam, the G8+5 environment ministers expressed the need to explore the economics of the loss of ecosystems and biodiversity. They were inspired by the momentum for early action and policy change, the Stern Review on the Economics of Climate Change had created. The European Commission and German Environment Ministry joined forces to pursue such a global assessment on the economic value of biodiversity and the related cost of policy inaction (COPI) associated with global biodiversity loss. A TEEB interim report on “The Economics of Ecosystems and Biodiversity” was presented at CBD COP 9 (EC and BMU 2008), together with a number of commissioned studies (including Balmford et al. 2008, Braat and ten Brink 2008, Mullan and Kontoleon 2008), that set the stage for further work. To gauge the scale of this cost, the authors sought to unite under one rubric the full range of ecosystem service benefits estimates in the extant literature, discriminated by biome. Fairly liberal use of benefits transfer has permitted that many conditions that are not covered by the literature be at least represented in the valuation profiles, despite what may be often dubious similarity among beneficiaries and providers and their political economic and cultural contexts.

Besides seeking aggregate and biome specific valuations, TEEB is elaborating a series of user-oriented toolkits directed toward decision makers including governments, business, local and regional resource managers and consumers. TEEB has been largely informed by ecological economics perspectives regarding such issues as the choice of a discount rate (low or zero), distributive justice (a rawlsian standpoint that would weight more heavily the generally greater dependence of the poor on ecosystem goods and services for their livelihoods) and critical natural capital limitations as essential for resilience to global change.

Largely due to the global scope adopted by TEEB, the applicability of its results to specific local and regional circumstances may be questionable. Likewise, there has been a tendency to assume a fairly level playing field in terms of the policy context within which such analysis and tools may be brought to bear. But this of course is rarely the case, since policy instruments must respond to the history and experience of policy frameworks, power structure and cultural acceptance of such measures in each sociopolitical context and moment of opportunity. These of course represent challenges faced by any such global effort, but we argue that it implies the need for a different point of departure in framing the questions that must be answered in developing context specific instrument packages in biodiversity conservation policies.

Current economic policy assessment criteria and their limitations in a policy mix

Real-life policymaking is seldom a neutral search for the optimal instrument to maximize welfare (Sterner 2003). In no other environmental policy-making field is this more true than biodiversity conservation and ecosystem services provision. Despite this recognition, recommendations by economists for policy mix design in the environmental field have mostly been written based on the nature of pollution issues, focusing on a limited set of policy design principles that have been amenable to economic modelling (OECD 2007); achieving maximum marginal benefits (**environmental effectiveness** criterion), minimization of marginal costs of achieving a given policy objective (**cost-effectiveness** criterion), equating marginal benefit with marginal costs (cost-benefit or **efficiency** criterion).

Comparisons of the effectiveness of a particular category of conservation policy or project has a number of data and methodological challenges (see e.g. Brooks et al. 2005, Pagdee et al. 2006, Engel, et al. 2008.), as does accounting of any given conservation policy's opportunity, implementation and transaction costs (Birner and Wittmer 2004, Wolf and Primmer 2006, Pagiola 2009). Valuation of the benefits of biodiversity conservation has been challenged both on grounds of theoretical consistency (e.g. Spash 2008), and methodological soundness (Mullan and Kontoleon, 2008). Cost-effectiveness or cost-benefit assessment ranking alternative "instruments" is therefore already a formidable task, assuming that internal methodological limitations and data gaps can be overcome at a particular location.

What is a policy mix?
Vatn (2005) distinguishes between three main categories of policy measures: *Economic instruments*, based on the assumption that agents, be they individuals or institutions, are rationally calculative; *Legal instruments*, assuming both rationally calculative and social or normative reasoning; *Informational instruments*, operating through cognitive and normative processes, and an assumption of agents' bounded rationality. In the context of designing payments for environmental services, Vatn (2008) frames the question of policy design as fundamentally a search for combination of alternative governance structures (hierarchies, markets, and community management) that can be used to handle ecosystem interconnections that cause externalities across socially defined property and use rights boundaries. Here we initially take a policy mix to mean combinations of multiple types of 'instruments' in a particular geographical area. The relevant geographical areas for evaluating the mix of instruments can be defined by the different governance levels (community, market, government level). Further on we will argue that for purposes of empirical analysis and policy design, a policymix should be decomposed into a *portfolio* of institutional characteristics.

To these optimization criteria, economists often emphasise other sub-goals that support welfare maximization, particularly incentive compatibility, dynamic effects and innovation, distributional and equity concerns, revenue generation, compliance cost control or administrative feasibility (Field 1994, OECD 1997, Sterner 2003). The task of evaluating policy "instruments" then moves conceptually from optimization on the margin, or ranking along a single C/E or B/C ratio, to ranking alternatives using multiple criteria assessment (MCA)techniques (Jansen et al. 2003, Munda 2008).

A conceptual problem with CEA, BCA and even MCA approaches arises in their use in evaluating policy mixes; policy combinations may be difficult to define as alternatives, and impacts across assessment criteria are likely to be correlated when existing policies constitute a common governance platform for the proposed alternative instruments.

Reviews of economic instruments in conservation hint at these problems. A recent review of biodiversity policy instruments has found little empirical evidence or reporting of the impacts of alternative policy instruments for forest biodiversity conservation (Mullan and Kontoleon, 2008). According to the review environmental outcomes are rarely assessed, and if so not in relation to costs of implementing a policy relative to a baseline. Bräuer et al. (2006) note that clear recommendations of when and where to use 'market-based instruments' instead of, or complementarily to, more hierarchical command-and-control approaches are hard to formulate because biodiversity is such a heterogeneous good - policies need to be tailored to local site specific needs. Both reviews point to the lack of consistency in evaluating cost-effectiveness of policy instruments, suggesting methodological weaknesses in previous studies, or prohibitive data collection costs given the difficulties in transferring existing results across heterogeneous sites.

Studies of cost-effectiveness and efficiency of economic instruments

A handful of papers suggest that the lacking conclusions regarding market-based instruments may be more than a 'measurement problem'. Analysis of policies for biodiversity conservation needs to deal simultaneously with several sources of market failure (Engel, Pagiola, Wunder, 2008) and therefore the implementation of various instruments needs to be considered simultaneously. Economic instruments are often introduced and applied in contexts in which various command-and-control regulations pre-exist (Wätzold and Schwerdtner 2005). Complementarities between instruments have been noted, for example between payments for environmental services (PES) and regulation (Engel, Pagiola, Wunder 2008; Wunder 2008). In fact, most PES schemes work with a mix of other policy instruments. Other examples are voluntary conservation schemes which has been found to be effective when there is a threat of regulation in the background (Langpap and Wu 2004), and a number of complementarities in combining the "stick" of regulation and the "carrot" of market based instruments have been pointed out (Wunder 2007). Further, as economically optimal policies can often not be defined, a "second-best" solution is searched on the combination of instruments (Landell-Mills and Porras, 2002; Bräuer et al. 2006). These issues complicate cost-effectiveness and efficiency assessments as one-on-one comparisons or rankings of policy instruments in a particular setting are severely limited.

Recent theoretical and case study review work, particularly on PES, has identified a number of feasibility criteria for the success of economic instruments. For example, Ferrao and Simpson (2002) use a model to identify the conditions under which direct conservation payments are more cost-effective than indirect subsidies. These conditions include available data to target land-uses providing ecosystem services (ES), low transaction costs, incentive compatible contracts, and enforced property rights. Several of the assumptions of the model are themselves legal and informational instruments , i.e. institutions for environmental monitoring, enforcement, contract law and property rights. From a conceptual point of view there is only a limited sub-set of characteristics of different economic instruments which are feasible alternatives, since all economic instruments build on some form of institutional context or pre-existing regulation. Based on a detailed review of PES schemes, Engel, Pagiola and Wunder (2008) identify some generic instrument assessment criteria[2], which additionally highlight the complexity of factors that have to be considered when evaluating economic instruments for conservation in their policy context. Wunder discusses where and when conservation payments work on the margin (Wunder 2007)[3], and which are necessary economic, competitive, cultural, institutional and informational preconditions for PES (Wunder 2008). He points out that the introduction of market incentives into a setting where PES conditions on the margin are not met can lead to PES undermining protected areas that are working relatively effectively. Further limitations to the implementation of economic instruments, such as PES, can be site specific such as communities' requirements for trust-building, or the unacceptability of *quid pro quo* agreements in donor crowded areas where conservation funding is easily available (Wunder, 2007). While these are examples of theoretical or *ex post* assessments, authors call for more work on *ex ante* assessment, baseline definition and incorporation of monitoring in the policy design. Before detailed design of PES is undertaken, assessments of the efficiency of existing policy instruments and of pre-existing motivations for environmental service provision should be undertaken (Wunder, 2007).

What then of non-market valuation in support of policy mix assessment in the field of biodiversity conservation? In principle, economic valuation of the benefits of biodiversity conservation has several roles to play, although we argue that these roles are limited in practice. *Awareness raising*: Brown and Moran (1994) call for valuation of biodiversity to raise "common concern". Using benefit transfer global level cost-of-policy-inaction studies have been made possible (Braat, L., and ten Brink, P. 2008). Non-market valuation results transferred and aggregated at this scale, have played an important role in raising the policy visibility of biodiversity conservation relative to other social issues. *Regulatory cost-benefit analysis:* We have argued above that evaluating economic instruments in a policy mix poses a problem of defining meaningful policy alternatives. The answer to the question "is PES a cost-effective biodiversity conservation instrument?" in case study reviews is often that it depends on the ecosystem service in question and the social and institutional setting (Engel et al. 2008). Assessment and design of economic instruments for biodiversity conservation is trying to respond to heterogeneity at landscape and farm level scale, so cost-benefit analysis of any particular economic instrument loses most of its meaning for decision-support at aggregate national level. *Spatial targeting:* determining which land-uses provide the biggest incremental biodiversity benefit compared to land-owners opportunity costs. Not being able to link changes in biodiversity and ecosystem services to changes in land-use is identified by some economists as one of the principle limitations to environmental valuation being used as a conservation policy assessment tool (Mullan and Kontoleon, 2008). To other critics, indicators that capture the spatial heterogeneity of biodiversity and ecosystem services cannot be expressed in a meaningful way to the public, and so cannot be subject to meaningful statements of preference or valuation (Spash 2008). Wunder (2007) points out that the costs of evaluating baselines and effectiveness of policies on ecosystem

[2] compliance/enrollment; targeting and additionality; documented link between land-use and ES; schemes permanence; leakage of effect and perverse incentives

[3] PES have potential in situations of positive but low to intermediate range opportunity costs; when the levels of threat to biodiversity loss are intermediate to low ; when PES can avert the on-set of land-use changes in a community where there are projected threats to biodiversity and where land-owners have credible site-specific claims.

services such as watershed services may easily exceed the aggregate benefits of the service itself in small watersheds. He advises rough farm-level opportunity cost calculations will often suffice to evaluate the feasibility of implementing PES practices. Non-market valuation at the spatial scale useful for targeting different economic instruments adds another layer of information costs to the often substantial costs of biophysical baseline and impact analysis. Benefits transfer seems initially to be a low-cost solution to this problem. However, meta-analysis of biodiversity valuation studies are dominated by variables of study design, with variables describing biodiversity impact, local population and institutional characteristics in minority (see e.g. Nijkamp et al. 2008).

Non-market valuation and benefit transfer has a very significant research challenge in addressing the concerns of spatial heterogeneity of biodiversity conservation. While this challenge is being met, operational conservation policy needs to find a balance now in using surrogate biophysical indicators for biodiversity conservation effectiveness that are at the same time complex enough to be sensitive to land-use change, and simple enough to be meaningful to land-users and regulators. We turn to this issue in the next section, before discussing a general framework for policy mix assessment in social-ecological systems.

The measurement of biodiversity and ecosystem services for policy assessment

Any assessment of the marginal benefits of conservation measures requires that conservation objectives are operationalised through a set of indicators that are appropriate to quantify gains. The science of systematic conservation planning (SCP) has developed a body of theory and methods to quantify how a set of areas on which conservation policies are implemented represent natural values, as well as which is the marginal contribution to biodiversity representation of each individual area (e.g. Barton et al. 2009). The key metrics of marginal gain is *complementarity*, i.e. the marginal contribution of an area to biodiversity representation (Margules & Sarkar 2007). The calculus of complementarity is based on various quantifiable indicators, typically features of the landscape that correspond with the patterns of distribution of natural systems (Austin & Margules 1986).

The potential to estimate marginal gains of conservation efforts opens opportunities to link policy effectiveness analysis with the body of theory and methods developed by SCP. However, a series of challenges remain to be solved before the methods can be effectively integrated into policy design and evaluation. The first constraint is that conservation objectives often go beyond that of biodiversity representation at national or regional scales. They are generally stated at various governance levels, spanning from those comprised in international agreements (i.e. CBD, Ramsar Convention), through national laws and directives, down to the management plan of a particular unit of land dedicated to conservation activities. Further, conservation values assigned to a particular area are typically multiple and of various kinds such as area size, degree of naturalness, provision of habitats for endangered species, representation of particular bio-geographic regions and protection of water courses (Anonymous, 2005).

More recently, other criteria than representation, and associated with the viability and probability of persistence of biodiversity attributes in the landscape (Polasky et al. 2005), have been incorporated into SCP approaches. These advances widen the scope from biodiversity representation to that of the maintenance of fundamental ecological processes (e.g. dispersal, colonization, meta-population dynamics) associated with the long term persistence of biodiversity (Margules & Sarkar 2007). Surrogate biodiversity indicators often target the provision of particular services or benefits (Fisher et al. 2009). The methodological integration of these criteria within surrogate indicators of biodiversity complementarity/representativeness is incipient (Moffet & Sarkar 2005) and has still not been tested in practical conservation planning (Margules & Sarkar 2007). These methodological advances based on spatial multiple criteria analysis, can constitute new opportunities to simultaneously evaluate the outcome on multiple biodiversity indicators without conducting non-market valuation or benefit-cost analysis. Although there is an increasing amount of evidence establishing a correspondence between functional diversity and ecosystem service provision (Bunker et al. 2005), there may be trade-offs between ecosystem service provision and other biodiversity conservation priorities (e.g. conservation of endangered species) (Fisher et al. 2009). MCA is appropriate for the consideration of different choices or when there exist a number of criteria which conflict to a substantial extent (Belton & Stewart 2002). The approach can therefore provide an analytical framework that enables a combined assessment of the outcome of policy instruments considering objectives of biodiversity conservation, ecosystem service provision and forms of sustainable use of nature which may conflict with each other.

The most challenging constraint to the assessment of the effectiveness of a mix of conservation instruments is the kind and quality of the data available to assess change across multiple spatial scales and conservation objectives. Biodiversity surrogates are often derived from various types of digitized cartography (climate, vegetation, bedrock, land-form maps), available at rather large scales (Austin 1991), which practically restricts the potential of the analysis to large land units at regional scales. Recently, remote sensing data and geo-referenced data bases have considerably improved data availability to describe and quantify natural variation (Margules & Sarkar 2007). However, economic instruments aimed at farm property / forest stand level are implemented at small-spatial scales which requires high resolution/high accuracy data to ensure robustness and transparency in the spatial targeting, monitoring and evaluation.

A conceptual approach to policy mix analysis: social-ecological-systems framework

The policies adopted at different levels might not be designed to match ecological scales (Cummings et al. 2006), and may generate unintended spatial externalities. Effective biodiversity governance therefore has to consider the spatial characteristics of conservation benefits and costs in relation to governmental levels (Perrings and Gadgil 2003; Ring 2008a). Figure 1 illustrates the challenge of carrying out an impact assessment regarding any particular instrument in the context of multiple levels of government and ecological scales.

Figure 1 ABOUT HERE

In rational planning, policy-making at higher spatial scales of governance determines which instruments may be legitimately implemented at lower levels of administration. In the local level context the social-ecological system determines the outcomes of the instruments. However, policies operate as a multi-level governance process, which makes policy implementation more context-relevant, but also creates discontinuities and problems of scale mismatch (Similä et al. 2006; Ring

2008a). Resulting uncertainty in policy implementation is best dealt with using adaptive policy design (Sterner 2003, Renn 2008), where impacts at lower tiers iteratively inform design at higher tiers (Ostrom 2007).

Research on the role of economic instruments in a policy mix therefore requires a conceptual framework that can work at several scales, across cases and be adaptable to the complexity of each case. It also needs empirical tools to implement such a framework that strike a balance between case specificity, generalisation and transferability. Research on governance in social-ecological systems (Ostrom 2007) advocates a tiered approach to evaluation of broad system variables, focusing empirical effort on higher resolution variables only where necessary. It suggests that social-ecological systems variables can provide the basis for evaluating the transferability of policy assessment results across geographically different case study contexts.

The original SES framework (Ostrom 2007) identifies first-tier variables as Resource System (RS), Resources Units, Governance System (GS) and Users (U) and Interactions (I) and Outcomes (O) between these. Table 1 provide an overview over first and second tier variables in Ostrom's SES framework. The variables are listed without reference to any particular type of resource system. In Figure 2 we put the SES framework in the context of evaluating a policy mix at different levels for management of forests and adjoining land-uses[4].

Figure 2 ABOUT HERE

Ostrom's original SES framework implies no causality. The SES framework is a menu of variables for a diagnostic of which resource regimes are sustainable and which are not. The SES framework makes no causality assumptions in the interest of avoiding 'panacea' policy solutions (Ostrom 2007) and is an advantage in putting different disciplines on an even footing. Causality becomes a concern once interest turns from ex post diagnostic to prospective impact assessment[5]. In Figure 2 we make a distinction between ecosystem functions as interactions in the biophysical system and ecosystem services as outcomes. What ecosystem functions are called ecosystem services is the result of use and governance systems[6]. This is an interpretation of the SES framework which implies that

[4] We use this generic SES framework to organise explanatory variables from several studies evaluating policies to address forest cover change. The exercise is meant to generate discussion on the complementarities between policy analysis approaches, as well as on the causality /hierarchy of explanatory variables for forest cover change, and in turn biodiversity conservation and ecosystem services provision. We return to Table 1 several times in the course of the discussion below.

[5] For example, economic analysis of biodiversity and ecosystem services within the context of SES framework requires some further conceptual clarification. Looking at second-tier variables in Table 1, economists would consider "economic value" (RU4) and costs of "human constructed facilities" (RS4) as measures of outcome under efficiency(O1).

[6] Vatn(2005.pp. 283) provides an earlier model of resource regimes, inspired by Ostrom, which makes "agent's choices" dependent on "attributes of the resource", "technology", "institutions - regimes", "patterns of interaction" and "outcomes" (resource use, state of the resource).

ecosystem services are the outcomes or social objectives of interest to policy analysis (Millenium Ecosystem Assessment, 2005; Fisher et al. 2009[7]).

SES suggests a framework for systematic comparisons of policy context. Table 1 illustrates that while second tier variables provide additional detail, lower-tier variables and measurable indicators are needed if SES framework is to be used for cross-case comparisons and describing sustainability characteristics of policy mixes. As an example, Table 1 compares SES second-tier variables with the criteria used by Brooks et al.(2006) in their study of28 Integrated Conservation and Development Projects (ICDPs); with Pagdee et al. (2006) study of 69 cases of Community Forest Management (CFM); with Wunder et al. (2008) study of 14 PES cases; and with Angelsen (2007) model of land rent. In table 1 we sort the significant variables found in these studies under the respective headings of the first-tier variables in the SES framework. Similar comparisons with other quantitative studies on conservation policy success/failure would point out what parts of the SES framework are most easily subject to systematic cross-case assessment[8]. The significance of certain site characteristics for project or management success found by Brooks et al. and Padgee et al. suggest that the task is empirically feasible in practice despite case heterogeneity.

The SES framework falls short of being a methodology for comparing the role of any particular economic instrument in, or the relative sustainability of, alternative policy mixes. An approach to decomposition of policy criteria and conditions in environmental policy design is needed. This is not novel, but comparisons of 'economic instruments' against one another have sometimes blurred the distinction between contextual conditions required for a policy to work, and objectives of the policy (see e.g. Sterner 2003). Also, given the challenges in defining economic instruments used in conservation, some form of decomposition of policy characteristics seems necessary for the purposes of evaluating a policy mix[9]. Brooks et al. (2006) recognised that although specific projects become known for specific strategies, in practice more than one strategy is used in any one project. Accordingly they focused not on the strategies or policies per se, but on the assumptions underlying the strategies currently embraced by conservation organizations, further detailing sub-goals for success/ failure. Despite the differences in conceptual models of numerous studies evaluating forest

[7] Fisher et al. (2009) write, "The concept of ecosystems services has become an important model for linking the functioning of ecosystems to human welfare. Understanding this link is critical for a wide-range of decision-making contexts. We argue that any attempt at classifying ecosystem services should be based on both the characteristics of the ecosystems of interest and a decision context for which the concept of ecosystem services is being mobilized. We discuss several examples of how classification schemes will be a function of both ecosystem and ecosystem service characteristics and the decision-making context".

[8] Brooks et al(2006) cite Bruner et al.(2001), Salafsky et al.(2001) and Struhsaker et al. (2005) as further examples of quantitative comparative evaluations of conservation policy successes and failures.

[9] For example, Engel, Pagiola, Wunder (2008) adopt a Wunder(2005) definition of Payments for Environmental Services (PES) as (a) a voluntary transaction where (b) a well-defined environmental service (or a land use likely to secure that service) (c) is being 'bought' by a (minimum one) service buyer (d) from a (minimum one) service provider (e) if and only if the service provider secrures service provision (conditionality). The definition adequately encompasses the 14 case studies of PES reviewed, but in doing so is too general to provide much guidance regarding what characteristics are unique to PES and what are common to other proposed economic instruments, or part of the existing mix regulations and economic incentives that affect biodiversity in a particular location.

management success/failure, Padgee et al. (2006) successfully used the SES framework to describe diverse policy settings in some detail. There may therefore be advantages to combining the SES diagnostic approach with environmental policy design criteria of the environmental economics literature [10].

A fleshing out of the governance system characteristics[11] of SES (Ostrom 2007) with observable "design principles" [12] of the self-organising common-property resource management institutions (Ostrom 1990), and criteria for environmental policy selection from environmental economics literature (Endres 1985; Field 1994; OECD 1997; Sterner 2003,) may be a basis for a generic hierarchy of policy 'traits' with observable indicators. A comparison of the institutional design criteria listed in the footnotes immediately show criteria operating at different levels of government/governance and spatial scales. The analysis of the performance of economic instruments therefore needs, at the very least, a criteria hierarchy with nesting of local/implementation level criteria within national policy criteria. Ostrom (2007) refers to 'nested conceptual maps'. At any governance level institutional 'traits' can also be further decomposed into elements of an institutional 'grammar' (Vatn 2005, Crawford and Ostrom 1995) [13].

Decomposition of policies into sub-goals and characteristics has a parallel in agroforestry research where the focus has shifted from comparing the cost-effectiveness of individual species in agro-forestry management, to presenting farmers with portfolios of 'traits' of locally available species, and selecting multiple species based on their capacities to perform specific or multiple functions[14]. Similarly there is a need to adopt a common nomenclature for describing the 'traits and functional

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[10] Criteria and conditions for policy selection: "static efficiency; relative slopes of marginal cost and benefits curves; efficiency with heterogeneity in damage costs; marginal costs flat benefits steep; inflation, dynamic efficiency, complexity, distribution and political issues; asymmetric information and risk, non-point-source pollution; small number of polluters, rent-seeking, general equilibrium; developing economy, global pollution; small, open economy" (Sterner 2003, pp.214-215); administration and compliance costs (OECD 1997); political/administratice feasibility (Endres 1985); fairness and moral considerations as in Field 1994.

[11] Governance system second-tier variables: "Government organizations, Non-Government organizations, Network structure, Property-rights systems, Operational rules, Collective-choice rules, Constitutional rules, Monitoring & sanctioning processes" (Ostrom 2007)

[12] Design criteria used to characterise CPRMs: "clear boundaries & memberships, congruent rules, collective-choice arenas, monitoring, graduated sanctions, conflict resolution mechanisms, recognized rights to organize, nested units" (Ostrom , 1990, p.90).

[13] Known as ADICO (Crawford and Ostrom, 1995) and exemplified in Vatn (2005) with the statement "Alpha's animals must not feed on Beta's cultivated land during growing season or else Alpha will be fined": (A) 'Attributes' are the characteristics of those to whom the institution applies (e.g. attribute is owners of animals), (D) A 'Deodontic' defines what on may (permitted), must (obliged) or must not (forbidden) do. Here the deodontic is 'must not' (I) An 'Aim' describes actions or outcomes to which the deodontic is designated. Here the forbidden action is feding on others' cultivated land. (C) A 'Condition' defines when, where, how or to what extent an Aim is permitted, obligatory or forbidden. In the example the condition is 'during the growing season' .(O) An 'Or Else' defines the sanction for not following the rules – here a fine.

[14] In the past, farmers have been presented with a limited list of tree species, often exotic to the region, that are very good at providing a limited number of services (fodder, fuel, food or soil fertility). A traits-based approach to agro-forestry system design is firmly grounded in recent theoretical advances in leading research in biodiversity and ecosystems functioning (Loreau et al. 2001).

characteristics ' of legal, informational and economic instruments. Such a nomenclature would make it possible to describe a portfolio of successful 'traits' and 'functions' , rather than combinations of instruments per se. The principle of SES diagnostic is to use variable levels of detail depending on policy-design needs and the information available.

Ostrom (2007) advocates nested conceptual maps to deal with the heterogeneity of situation variables affecting the sustainability of governance systems. While the SES framework is a rich conceptual approach, hypothesis testing requires some form of quantitative analysis that can deal with variable data types and sub-sample sizes at different hierarchical levels[15]. The multi-dimensional nature of conservation policy success also complicates model specification, with hypothesis testing facing multi-dependent variables (Pagdee et al. 2006). In the studies we reviewed, the variation across dependent and independent variables needed for hypothesis testing has been generated using cross-case comparison and meta-analysis techniques, assessing one dependent variable at a time (Brooks et al. 2006, Padgee et al. 2006). For multiple dependent variables and hierarchies of diagnostic criteria, Bayesian belief networks and hierarchical Bayesian modelling may be a methodological approach to tackle this, provided a sufficient number of cases are available[16].

Case Studies: Evaluating the role of economic instruments in a policy mix

What are the empirical challenges of evaluating the role of particular economic instruments in a policy mix? How can the SES framework be used conceptually and empirically at different spatial scales (landscape level/governmental levels), in addressing different actors (local governments land users) and policy paths? Ecological fiscal transfers (EFT) implemented in Brazil and Portugal provide an example of challenges in assessing effectiveness of economic instruments at different government

[15] Ostrom (2007) writes "Researchers who prefer case studies sometimes presume that the third- or fourth-tier variables observed in their studies are present in most other broadly similar SESs. When scholars suggest that a particular variable is important, other researchers sometimes respond, ''Not in my case!'' with the implication that the variable would not be important elsewhere. The concept of nested tiers of variables that interactively affect how other variables help or do not help to explain outcomes is a challenge to the way many scholars approach theory and explanation. Scholars who prefer to collect large samples and use multiple regression or similar statistical techniques are initially horrified when a large set of variables is listed, given the cost of obtaining reliable indicators of the same variable across cultural settings. Mistakenly, they presume that all of these variables need to be measured and included in future research. Instead, third-, fourth-, and fifth-tier variables are potentially relevant only when they are subcomponents of a second-tier variable posited to affect interactions and outcomes."

[16] Bayesian belief networks (BBN) (Kjærulff and Madsen 2008) and hierarchical Bayesian regression (Clark 2005) are empirical tools that can handle hierarchies of explanatory variables, variable data types and variable sample sizes across variables and cases. Crucially Bayesian statistics can be used to update prior beliefs generated using meta-analytical cross-section data, with longitudinal monitoring data in a specific site of interests. Baynes et al. (2008) provides an example of how monitoring data from farms participating in 3 years of agro-forestry extension were used in a BBN to model the causal factors and probability of agro-forestry adoption rates. At the level of modelling individuals' behaviour, cross-sectional stated-preference survey data is also easily accommodated as the lowest level of a hierarchy of variables describing the adoption of specific policy characteristics. Joshi et al. (2001) demonstrate how survey data and Rapid Rural appraisal data and farmer survey data are combined in a Bayesian Belief network of management options in rubber-tapping agro-forestry in Indonesia.

levels. Payments for environmental services (PES) as implemented in Costa Rica exemplify the importance of the policy path and instrument complementarity for the effectiveness of PES.

Ecological fiscal transfers

Public spending includes fiscal transfers between different levels of government – an economic instrument that is still often neglected in conservation policies (Ring 2002). Tax revenues are redistributed from national to state and further on to local governments to provide the latter with monies to fulfil their public tasks: building schools and hospitals, or constructing and maintaining roads. Ecological fiscal transfers compensate local and regional governments for the ecological goods and services they provide across local boundaries. Protected areas and local conservation efforts often involve costs at the local level – not least in the form of land-use restrictions – , whereas many of the conservation benefits are provided to higher governmental levels (Perrings and Gadgil 2003, Ring 2008a). Ecological fiscal transfers are a suitable instrument for internalising spatial externalities related to the conservation and sustainable use of biodiversity accruing between public actors at different levels of governments.

Only first steps have been made to introduce ecological fiscal transfers by a few countries worldwide: Several states in Brazil have introduced the ecological ICMS, acknowledging biodiversity conservation for the redistribution of value-added tax from state level to municipalities (Loureiro 1998, Grieg-Gran 2000, May et al. 2002, Ring 2008b). Paraná was the first state in Brazil to introduce the ICMS Ecológico (or ICMS-E, according to May et al. 2002) back in 1992. The ecological ICMS has now been adopted by 12 out of 27 Brazilian states (see Figure 5- Brazil map); others are preparing relevant legislation. The states implemented various ecological indicators for the redistribution of state value-added tax income to municipalities, but Conservation Units (CUs) are the ecological indicator used by all states and relate to the categories of protected areas for biodiversity conservation in Brazil. Portugal has introduced a new national community financing law in 2007 that includes ecological fiscal transfers. The Portuguese fiscal transfer scheme explicitly rewards municipalities for designated Natura 2000 sites and other protected areas within their territories (Prates 2008, Ring 2008b).

Figure 5 ABOUT HERE

Both Brazil and Portugal compensate municipalities for land-use restrictions imposed by protected areas and thus combine a regulatory instrument (designated protected areas) with an economic instrument (fiscal transfers). Furthermore, the ecological ICMS in Brazil has developed as an incentive to value and engage in more conservation activities at the local level, including the designation of new protected areas (May et al. 2002) and improving the quality of existing conservation units (Loureiro 2002). For example, in Paraná, since the introduction of the ICMS-E Conservation Units increased by 165 % in the year 2000, and municipalities with larger shares of protected areas considerably benefited from increased fiscal revenues (May et al. 2002).

However, the Brazilian experience also showed that it is important to introduce ecological fiscal transfers with a good information and communication strategy. In the state of Minas Gerais, the decentralized organization of the State Forest Institute helped to publicize the new instrument, along with its task to monitor all information related to ICMS-E transfers based on conservation units

(Bernardes 1999). Otherwise local governments and citizens may just not know that – at times even a substantial – part of their local public revenues stems from ecological fiscal transfers. Thus, informational instruments clearly contribute to the increased environmental effectiveness of ecological fiscal transfers, although this information does not lead to increased conservation activities by all municipalities that benefit from ecological fiscal transfers (May et al. 2002). But in many cases, this knowledge can reduce local opposition to protected areas and thereby eventually lead to an increase in the quantity of protected areas and – if the indicator for tax redistribution considers the quality of protected areas as it is the case in Paraná – also lead to improving the average quality of protected areas (Loureiro 2002).

In contrast to the rapidly increasing amount of literature on PES schemes, there is relatively little available on evaluating the performance of ecological fiscal transfers. Major empirical research on the ecological ICMS in Brazil has been published by Loureiro (1998, 2002), Bernardes (1999), Grieg-Gran (2000), May et al. (2002), including a more recent review by Ring (2008b). Scientific literature on the Portuguese case is still mostly absent (with an exception of a master thesis by Prates 2008). Further published research on ecological fiscal transfers relates to the design and potential impacts of such instruments that may be newly introduced in contexts where they do not yet exist (Köllner et al. 2002, Ring 2008c).

In this way, ecological fiscal transfers provide an interesting and rather new case for comparative analysis on the effectiveness and efficiency of biodiversity conservation instruments in a policy mix in different SES settings. Whereas in Brazil, economic instruments for compensating the opportunity cost of biodiversity conservation first focused on local governments, other economic instruments directly addressing land-users are more recent. In Europe the situation is different. Here, there is long history of compensating land users for their benefits foregone through agri-environmental payments (Ring 2008b), considered a European variant of PES scheme (Vatn 2008).

In any case, having both ecological fiscal transfers and PES schemes to land-users available poses the important empirical challenge of distinguishing between opportunity cost to local governments and those to the private sector to be able to quantify adequate payments to either municipalities or land-users in the local context. Another important empirical challenge relates to quantifying the spillover benefits – and finally the spatial externalities – of biodiversity conservation. Although it is widely uncontested that biodiversity conservation involves benefits at higher governmental levels up to the global level (Revesz 2000, Perrings and Gadgil 2003, Ring 2008a), there are few studies actually trying to quantify the spillover benefits of conservation (e.g., an exception is Horton et al. 2003). In this context, benefit transfer studies play an important role, because it will be impossible to evaluate the spillover benefits of biodiversity conservation for any possible setting. The SES framework promises to provide guidance on matches between existing valuation studies and social-ecological context,, building on the multiple variables and tiers covered by the framework.

Payments for environmental services

Payments for environmental services (PES) is a category of economic instrument in conservation policy and encompassing a number of variants in country-wide schemes and local projects (Landell-

Mills and Porras 2002; Engel et al. 2008; Porras et al. forthcoming). Given the wide array of adaptations of the generic PES concept (Wunder 2005), the question arises which contextual factors systematically define the constraints and feasibility of PES. Costa Rica's PES is probably the best documented scheme in the tropical world, providing a good case to discuss the empirical challenges of assessing PES role in a policy mix.

The interaction between PES and other policies on enrolment in PES; effectiveness on land enrolled in PES; spill-over effects to the surrounding landscape; and the role of PES in 'forest transition' processes have been mentioned by most authors, but few have assessed linkages systematically[17]. Disentangling the effects of Costa Rica's PES programme from that of other conservation policy measures and economic trends is difficult. Aggregate studies of changes in forest cover are difficult to compare as they apply to different areas, time periods, use different dependent variables and methodologies (Pagiola 2008). Costa Rica's 1996 Forestry Law which legally established PES, is itself a policy package of informational, legal and economic instruments: the law defines forest (in terms of percent canopy and area), bans forest conversion to other land uses, restricts logging within 15-50 m (depending on slope) on either side of rivers, and establishes that payments to landowners for forest conservation and reforestation can be made for carbon, watershed, biodiversity and landscape services (Harvey et al. 2008). Thanks to recent PES targeting, a substantial share of PES contracts are to be found within mixed use conservation units in buffer zones around and biological corridors between national parks and biological reserves. Calvo-Alvarado et al. (2008) argue that the success attributed to PES must be interpreted in the context of conservation policies in place, including protected areas, and the general restrictions on timber extraction and forest clearing on private lands. By 2005, 270 000 hectares were enrolled under PES, 95% of which were for forest conservation activities (Pagiola 2008), while approximately 1 333 000 hectares were in protected areas, with 647 000 hectares in state national parks and biological reserves (Estado de la Nación 2007). To our knowledge the research on the effectiveness of PES has not evaluated the role of location within mixed use conservation units and proximity to strict conservation areas.

Costa Rica's elaborate PES system has been credited with turning on of the world's highest deforestation rates during the 1970s to net reforestation by the early 2000s (Pagiola 2008). Recent research acknowledges that the transition has been built on a series of forest and conservation laws dating back to the early 70s; and set in the context of economic development that contributed to reducing deforestation pressure (Sanchez-Azofeifa et al. 2003; Calvo-Alvarado et al. 2008;

[17] In meta-analytical studies evaluating drivers of deforestation a number of country characteristics have been found to be significant, including income, population growth/density, agricultural prices/returns, agricultural yields, agricultural exports/export share, logging prices/returns/production, roads and road building, scale factors (size of forest stock, land area etc.), institutional factors such as political stability, property rights, rule of law (Arriagada 2008). At lower scales and higher resolutions within Costa Rica one would expect to find that similar drivers are significant. Arriagada 2008 also summarises results from a number of regional studies on causes of deforestation in Costa Rica: price and demand for meat, coffee and bananas; available low interest loans and credit; issuance of forest permits with new forestry laws; location along rivers and roads; soil fertility, temperature, and slope ; population density, unprotected status, and past clearing (endogenous dynamic effect).

Arriagada(2008); Pagiola 2008; Harvey et al. 2008). A brief summary[18] of how Costa Rica's conservation policy mix developed as part of a forest transition (Mather 1990), indicates the challenges in evaluating the cost-effectiveness of PES as a 'standalone' economic instrument. During the period of 1930-1950 economic pressure and population growth in the central highlands pushed landless peasants to lowlands in search of land for subsistence farming. Land colonisation policies and an expansion of road network supported and responded to this development. Land clearing in lowlands such as Guanacaste coincided with a response to international markets. The period (1908-1960) saw large exports of previous timber such as mahogany to the US from areas. From 1950 forest clearing and then land use intensification in the cattle industry took place responding to favourable beef prices, with cattle heards peaking in 1972. By the 1960s agriculture such as cotton and sugar cane was replacing some of the pastureland.

Costa Rica's first forest law of 1969 regulated forest use on public land and established national parks system; it would be followed by new forest laws in 1979, 1986 and 1996. A series of policy and institutional changes would also follow over the next four decades shaping the policy context of PES. In the period 1974-1978 National Parks and Biological reserves would grow in area from 3% to 12% of the national territory. The forest law of 1979 introduced subsidies for reforestation and forest management on private land. 1980 saw the creation of an Environment Department. After 1980 international beef prices went into sustained decline. By the mid 1980s the government was withdrawing support policies for the beef industry. The forest law of 1986 introduced the Forest Credit Certificate extending forest management subsidies also to smaller landholders. The Ministry for Natural Resources, Energy and Mines is created in 1988 (MIRENEM) including responsibility for forest and wildlife management.

Different parts of the country were going though different forest transition stages with both net deforestation and reforestation taking place. For example, Guanacaste province experienced a negative deforestation rate of -0,1% p.a.(forest cover increase), compared to the national average deforestation rate of 0,89% p.a. during 1979-1986; in the period 1986-2005 forest cover increased from 23.6% to 47% against a national average of 40,5% to 47,9% (Calvo-Alvarado et al. 2008).

The pace of conservation policy initiatives had been high and was not without resistance. A 1994 constitutional ruling by Costa Rica's Supreme Court held that the government could seize land from private owners only if the latter were fairly compensated; 15% of national parks and 46% of biological reserves, national monuments and absolute natural reserves had been seized without compensation (Segnini 2000 in Sanchez-Azofeifa et al. 2003). In 1995 the Forest Protection Certificate was introduced. The same year MIRENEM is restructured as the Ministry of Energy and Environment (MINAE) and the three major agencies in charge of the protection of the conservation systems (The Forestry General Direction, the National Parks Direction and the Wildlife General Direction) consolidated into a National System of Conservation Areas (SINAC). SINAC is organised in

[18] Our brief summary cannot do justice to the rich literature on Costa Rica's forest history. For the account contrasting economic and forest transition in Guanacaste with the rest of Costa Rica, we draw on Calvo-Alvarado et al. (2008) and the sources reported therein; and for an overview of policy development that was happening concurrently also on Sanchez-Azofeifa et al. (2003) and Pagiola (2008). Summarising events more or less chronologically does not necessarily mean causality, but helps underline the idea of co-developing policies.

11 Áreas de Conservación (AC) including the conservation categories in their charge and all the national territory. This aimed at a unified land management strategy both inside and outside conservation areas, promoting new core protected areas, buffer areas and biological corridors (GRUAS 1995). A separate Biodiversity Law in 1998 was enacted to provide a legal definition of conservation.

The 1996 forest law mandated PES for carbon, watershed, biodiversity and landscape environmental services. PES payments built largely on pre-existing forest incentives for timber plantations, sustainable forest management, and forest conservation; by 1996, 150 000 hectares had been enrolled under forest incentive schemes due to previous forest laws (Pagiola 2008). Since the 1996 forest law, Costa Rica's PES program has continued its path of trial, error, and reform, with numerous changes in how what land-use practices are targeted, PES administration and financing. In 2000 PES categories are restricted to timber plantations and forest conservation. In 2004 an agroforestry PES contract was introduced. Starting with the 1996 Forest Law, PES was administered by the National Fund for Forest Financing (FONAFIFO) with execution through SINAC from 1997 and until 2003. All responsibilities for PES allocation were passed to FONAFIFO after 2003. Landowners were initially contracted by SINAC and by large NGOs such as FUNDECOR. FONAFIFO took over this task in 2003, establishing eight regional offices to handle applications, sign contracts, and monitor implementation. Regarding financing, multiple environmental services under the 1996 law have lead to a creative and evolving mix of sources of private and public , national and international financing (Pagiola 2008). In watershed services PES has evolved from negotiation of individual contracts with hydropower companies, through Environmental Services Certificates for conservation of an unspecified hectare of forest, to compulsory conservation fees as part of all municipal water tariffs under a 2008 Water Law Financing loosely tied to carbon services has also evolved from a first sale of Certifiable Tradeable Offsets to Norway, a 3.5 % of fossil fuel tax allocated to FONAFIFO for PES, and incipient sales of carbon offsets on the voluntary retail market. Grants from GEF, World Bank and the German Aid Agency had promoted targeting of PES to biodiversity conservation priority areas identified under GRUAS study.

The literature evaluating the effectiveness of PES in Costa Rica is extensive[19], evaluating impact at different scales, including factors affecting farm level adoption (e.g. Pagiola et al.2004; Miranda et al. 2006; Sierra and Russman 2006; Arriagado 2008), factors affecting cost-effectiveness of PES targeting at regional level (e.g. Wunscher et al. 2008; Barton et al. 2009), aggregate forest impacts at regional level (e.g. Calvo-Alvaradoa et al. 2008) and national level (Robalino et al. 2008; Arriagado 2008). A consensus is emerging from these studies that PES impact on avoided deforestation/forest conservation has been negligible, whereas its impact on forest regrowth has been small, but significant. Costa Rica's PES system is evolving from its origins as an untargeted and undifferentiated payments that were seen as a quid pro quo for the costs imposed on land-owners by legal restrictions on forest conversion (Pagiola 2008).

The recognition of a gradual co-development of PES with other conservation policy instruments, and its introduction during a forest transition process, highlights the need to look at governance variables

[19] We admit not doing justice to all the available literature. We reviewed selected studies that had discussed the effectiveness of Costa Rica's PES in geographical and historical context.

and time lags in the study of the cost-effectiveness of PES. Costa Rica's PES was perhaps never a clear policy instrument alternative under consideration by policy-makers, but an evolving package of legal, information and economic instrument characteristics. Complementary approaches to quantitative cost-effectiveness studies are needed to explain the feasibility of different economic instrument characteristics of PES in different settings. Below we discuss these issues further and the potential of the SES framework.

Discussion

Based on the recent studies relating the effectiveness of PES and EFT, some conclusions can be drawn regarding approaches to assessing the role of economic instruments in a policy mix.

Cost-effectiveness studies comparing alternative economic instruments versus legal instruments at an aggregate level are complicated by the fact that they have been introduced as part of a policy package from the start (e.g. Costa Rica's 1996 Forestry Law). A cost-effectiveness analysis comparing legal and economic instruments in the Forestry Law would be technically challenged in trying to disentangle *path-dependency* effects[20]. To give a slightly stylised example, the cost-effectiveness analysis would be in danger of ignoring the fact that PES payments were designed as a quid pro quo for the ban on forest clearing, and that the ban on forest clearing across the country was a reaction to weak forest management already in place in e.g. Forest Reserves.

The same holds for Portugal's ecological fiscal transfer scheme that has been introduced in addition to agri-environmental schemes related to EU Agricultural Policy existing for a much longer time. Furthermore, ecological fiscal transfers by purpose build on protected areas, a classic legal instrument of biodiversity conservation, when they use the size of protected areas as an indicator for redistributing tax monies.

The study of the *effectiveness* of economic instruments (EFT and PES) and legal instruments such as protected areas, suffer similar problems of endogeneity in causal factors, for example allocation to high deforestation risk areas. Matching methods have thus far compared deforestation rates in similar spatial units with and without PES, controlling for biophysical, location and socio-economic characteristics (Robalino et al. 2008; Arriagado 2008). The same techniques should be possible to use in assessing the spatial complementarities between protected areas and PES. However, controlling for the effects of other policy instruments will place great demands on cross-section *sample sizes*, which may not be sufficiently available in any specific region (Arriagada 2008). In most countries with PES schemes in place time series data on PES implementation is very limited (Costa Rica's 12 years of data represent the most optimistic scenario in this sense). Calvo-Alvarado et al. (2008) demonstrate the importance of evaluating the path dependence of different conservation policies, but the lack of time series data place serious constraints on quantifying these dynamics. Although EFT have first been introduced in Paraná in 1992, due to the rather exotic nature of this

[20] CEA practitioners often do their best to ignore the complementarities between the instruments in order to make a clear ranking between alternatives. In modelling of costs and effects of any given instrument economists often invoke the *ceteris paribus* assumption.

instrument in the scientific conservation community, there are fewer data and publications available, and this is even truer for EFT being introduced more recently.

Instruments may also be subject to *other political objectives* than 'narrow' cost-effectiveness. PES in Costa Rica may not have 'additionality' as an objective (Pagiola 2008) when they are seen as compensation for obligatory conservation measures such as under the Costa Rican Forestry Law of 1996. FONAFIFOs approach is to 'recognise' environmental services of whomever is providing them (Pagiola 2008). A proper process of PES negotiation could also be seen as a platform for democratization and improved governance, motivating the interest of donors (Wunder 2008).

The number of relevant alternative instruments to be considered as part of a policy mix naturally increases with *spatial and temporal scale* of the analysis. When research focus is placed on private farm land, PES constitutes the main conservation instrument in Costa Rica. If other types of private land use are considered, such as tourism, other instruments such as private reserves and conservation easements become relevant alternatives. If geographical and time scale is increased further, different land use regulations become relevant options for the policy mix; including different categories of state protected areas (Biological Reserves, National Parks) and privately owned multiple use areas (Forest Reserves and Wildlife Refuges, National Wetlands).

On private lands in multi-purpose protected areas, various instruments are playing different *complementary roles*; a proximity effect of being close to "core" state protected forest areas; compliance with the blanket ban on forest clearing; compliance with land-use regulations of the multi-purpose area proper (e.g. forest management regulations and hunting bans); and finally PES. Due to the spatial configuration of watershed, landscape and biodiversity conservation services (Fisher et al 2009), we would expect the role and design of multiple use protected areas and PES would be completely different in the absence of "core" protected areas such as National Parks and Biological Reserves[21]. This is why we think an analysis of "roles" is more appropriate than an analysis of "cost-effectiveness" of instruments. The toolbox analogy is once again relevant: (i) It is technically possible to evaluate the cost-effectiveness of a hammer versus a saw in the building of a house, but the analysis would be trivial because the tools were purposively designed to have different roles in different areas of house building; (ii) the cost-effectiveness of alternative designs of a given tool in carrying out its designated role is however non-trivial; (iii) there might also be characteristics of different tools which can be recombined to address new requirements of house building. Hence, the ongoing development of e.g. PES with new institutions to address the different externality configurations of ecosystem services might be studied as recombinations of *policy package characteristics*, rather than as new instruments per se. The SES framework provides a *tiered approach* to such an analysis.

The need for an analysis of the policy mix arises mainly at the *landscape scale* or higher. García – Fernandez et al. (2008) assess multiple-use forest management at stand level and landscape level.

[21] In biodiversity conservation 'additionality' is can be interpreted as the number of additional habitat characteristics or species are protected with e.g. a market-based incentive versus characteristics already represented within protected areas . Barton et al. (2009) discuss how the biodiversity complementarity value of any given location varies with the set of locations that are already within this 'protected set' .

Their review of a number of land-use modelling studies find that land-use specialisation at forest stand level, with multiple-use management at landscape level provides superior returns to multiple-use management at stand level. Multiple use at stand level is an option where MFM techniques are low complexity, low cost, typical of extractive uses in hunter-gatherer societies. Richer societies that have passed through a forest transition experienced mixed societal demands on forests which can favour multiple-use across the landscape with land-use specialization at stand level. The set of economic and legal instruments applicable to any particular forest stand, land-use or property rights combination is usually quite restricted, making cost-effectiveness analysis comparing instruments of limited relevance to the owner of a particular forest stand. At landscape scale, overlapping forest characteristics, property rights and administrative levels define a mix of relevant economic instruments. At this landscape level economic instruments play complementary role by addressing different land-use specializations and externalities across governance boundaries. Figure 3 shows an idealised example where economic instruments for biodiversity conservation, often seen as alternatives, have complementary roles in the same landscape. The discussion on 'targeting' found in the PES literature, focuses mainly on biophysical and farm household characteristics (Pagiola et al. 2005). The SES framework provides a rich backdrop of situation variables which can potentially aid in complementary targeting of economic instruments.

At the regional and country scale[22] the *forest transitions* concept (Mather 1990) suggests that appropriate instrument types and combinations depend on the stage along a *development path* (García –Fernandez et al. 2008; Barbier et al. 2009, Angelsen 2007). García–Fernandez et al. (2008) discuss three stages of transition from multiple forest extractive uses (high forest cover), to land use specialization (falling and low forest cover), and sustainable forest management (forest recovery). Angelsen (2007) distinguishes in the last stage between forest mosaics with stabilised cover, and increasing forest cover due to afforestation and reforestation (Figure 4).

The literature on cost-effectiveness of PES from Costa Rica has begun to report limit effectiveness of PES on forest protection, while positive results for forest growth. Researchers have also begun to recognise that PES might be most adapted for the last stage of forest transition (García –Fernandez et al. 2008). More generally, one can ask whether the economic instruments for biodiversity conservation and ecosystem service provision in forests can play the same role in forest transition stage 4 as in stage 1, indeed whether "biodiversity conservation" and "ecosystem services provision" imply different roles for instruments and different policy mixes [23].

Just as there is no policy panacea for common property resource management (Ostrom 2007), there is no single theoretical framework that includes the dimensions needed to evaluate a policy mix for biodiversity conservation and ecosystem service provision. Recently, forest cover change research has emphasised the complementarities of the *temporal* focus forest transition theory with the spatial approach originally proposed by von Thünen where land rents and distances to markets are determinants of land use patterns (e.g. Angelsen 2007), Barbier et al. 2009). Angelsen (2007) argues that the spatially dependent land rents approach provides "a static theory of how land rent

[22] -Across different forest landscape configurations (Mertens and Lambin , 1997)

[23] "Biodiversity conservation and ecosystem service provision" are often said in the same breath, but imply different types of policies if seen in the context of forest transitions.

determines land use and an empirical proposition on how land use is determined by location, while the forest transition approach provides an empirical proposition on how land use changes over time". Partial equilibrium analysis using agricultural and forest land rent models use a number of intermediate factors determining land-owner use decisions. In extension we think the SES framework and CPRM literature may offer empirical propositions on underlying factors driving land rent. Table 1 shows the complementarities between 'underlying factors' in land rent models (Angelsen 2007) and second tier variables in the SES framework. The land rents approach focuses on explaining agricultural rents relative to forest rents as the dominant driver of forest change (Angelsen 2007, Barbier et al. 2009). There is clearly a research agenda in combining the literature on PES in agriculture and forestry to address policy mixes that promote stable forest-agriculture mosaics (stage 3) (see e.g. Kroeger and Casey 2007).

This should also lead to an increasing recognition that in a *landscape mosaic* seen at a large scale of analysis there are several processes of forest transition happening in parallel, requiring differentiation and the flexibility to adapt the composition of policy-mixes in a particular landscape as it goes through a transition. A hypothesis for cross country comparison, is not only that countries with democratic political institutions are more likely to experience positive forest trends (Barbier et al. 2009), but also that federal and devolved forest governance should fare better than centralised systems. The issue of path dependence shows that both qualitative and quantitative research methods will need to be applied. The SES framework and CPRM case literature (Ostrom 2007) offers a rich framework from which to draw potential causal variables. A drawback of the SES framework is that it does not indicate a hierarchy of necessary conditions, nor offer causal models. Land rent and forest transition theory does offer such causal models. The literature on pre-conditions for PES springs to a large degree from the land rents approach (Wunder 2008, Engel et al. 2008). It might help define a hierarchy of 'tiered' conditions for economic instruments within the SES framework. These hypotheses could also be the subject of further testing in matching studies evaluating the effectiveness of instrument characteristics across the landscape(see e.g. Arriagada 2008). Given that SES springs from CPRM theory on sustainable common property management institutions, a further research hypothesis is that underlying drivers of land use change on the forest frontier also determine the types of governance systems that appear in stages 3-4 of forest transition. We might speak of a potential framework for evaluating 'land-use mozaic institutions'[24].

This discussion also begs us to question the degree to which policy mix implications of regional land use gradients may be transferable between cultures and policy contexts, given the importance of *historical moments of opportunity*. PES may stimulate compliance where forest protection would enable proprietors to comply with environmental constraints to which they are legally liable irrespective of initial forest cover (e.g, Brazil's Legal Reserve requirement combined with greater enforcement commitment in the southeast Amazon). REDD is said to be most desirable under conditions of high deforestation pressure and high remaining forest cover, yet may be more feasible to implement where deforestation has not reached a serious level (stage 1 in 'forest transition'), but where forest communities are well organized internally and able to marshal REDD payments to stifle exogenous pressures to open up new frontiers (e.g., the argument of the Bolsa Floresta programme).

[24] The focus in the CPRM literature has been on analysis of sustainable community forest management institutions.

It can be argued that Costa Rica's recent conservation history puts their introduction of PES in 'stage 3' of the forest transition, with protected areas, international terms of trade and economic development having brought the country out of a deforestation and laid the ground for PES to contribute to forest regrowth ('stage 4'). It was also more politically feasible to implement the FONAFIFO model in Costa Rica in a time at which the nation was actively searching for new sources of foreign exchange, and saw the possibilities inherent in the carbon market with considerable prescience.

Conclusions: A research agenda

Cost-effectiveness analysis of economic instruments is complicated by the fact that they are usually part of a path-dependent policy package co-developed over time. Legal, informational and economic instruments are often implemented simultaneously and play complementary roles. Other political objectives than cost-effectiveness , such as poverty and fairness concerns may make economic incentives part of sustainable biodiversity governance, although their strict 'additionality' and cost-effectiveness is difficult to demonstrate. Even though economic instruments are targeted at specific land-uses there may be spatial spill-over effects mutually affecting rates of forest cover change (e.g. between PES and protected areas). Because stand level land-use specialization is often economically efficient, multiple use forest management, including promotion of environmental services, is more likely in a landscape mosaic. The feasibility of economic instruments targeted at stand level, may be greater in the last stages of forest transition when stand level specialization within a landscape mosaic is optimal (i.e. PES in Costa Rica). In the first stage before a deforestation, landscape level incentives with higher scale, lower resolution may be more appropriate (i.e. such as EFT compensation for protected areas in Brazil).

In evaluating a policy mix an eclectic combination of methodologies is needed: the theory of forest transitions to focus on the temporal context of policy, a spatial theory of land rents, and a tiered framework for evaluating governance context and path dependence of sustainable institutions of which economic instruments are a part. We believe that the social-ecological systems may provide a framework for such a synthesis, although this remains to be tested. An approach to decomposing economic instruments into its underlying institutional characteristics would also make it easier to evaluate policy mixes as portofolios of characteristics, rather than instruments per se. This also remains to be tested.

Non-market economic valuation methods are of interest as an awareness raising tool at aggregate level, and perhaps in quantifying spill-over effects at larger administrative levels, such as in the context of quantifying ecological fiscal transfers. However, is hard to see the role of non-market valuation methods in targeting economic instruments within landscapes, or at stand level, given the already considerable methodological challenges in defining (surrogate) indicators for biodiversity and ecosystem services. Defining such surrogate indicators of conservation status is a pre-condition for the relevance of non-market valuation methods in targeting specific types of forest areas for the level of ecosystem services they provide. Until then rough opportunity cost-based approaches will often be sufficient to demonstrate economic feasibility of PES in particular forest stands at farm level (Wunder 2008).

Cross case comparison of the roles of economic instruments in a policy mix needs a common framework for comparison, which we think SES might provide. To the extent non-market valuation estimates are accepted for policy design (targeting or differentiating service payments), SES variables provide a comprehensive check list for evaluating site similarity and a priori validity of benefits transfer. Latin America's experience with economic instruments in conservation policy, exemplified by differential implementation of EFT across a number of Brazilian states, and the –dependent development of PES in Costa Rica, should provide a number of lessons for Europe, as well.

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References

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 D.C.

Anonymous. 2005. Joint management plan for Trillemarka – Rollag Østfjell. Management plan draft 11.03.2005 (in Norwegian).
Austin M.P. (1991). Vegetation theory in relation to cost-efficient surveys. Nature Conservation: Cost effective biological surveys and data analysis. Margules C.R. & Austin M.P., CSIRO, Australia, pp. 17-22.
Austin M.P. & Margules C.R. (1986). Assessing representativeness. . Wildlife Conservation Evaluation. Usher M.B., Chapman & Hall, London, pp. 47-52.

Arriagada, R.A. (2008) Private provision of public goods: applying matching methods to evaluate payments for ecosystem services in Costa Rica. Ph.D. Dissertation. Graduate Faculty of North Carolina State University.

Balmford, A., Rodrigues, A.S.L., Walpole, M., ten Brink, P., Kettunen, M., Braat, L. & de Groot, R. (2008). The Economics of Biodiversity and Ecosystems: Scoping the Science. Cambridge, UK: European Commission (contract: ENV/070307/2007/486089/ETU/B2)

Barbier, E.B., J.C. Burgess, A. Grainger (2009) 'The forest transition: Towards a more comprehensive theoretical framework'. Land Use Policy. doi:10.1016/j.landusepol.2009.02.001

Barton D.N., T. Saloranta, S.J. Moe, H.O. Eggestad, and S. Kuikka, (2008) Bayesian belief networks as a meta-modelling tool in integrated river basin management — Pros and cons in evaluating nutrient abatement decisions under uncertainty in a Norwegian river basin. Ecological Economics 66, 91–104.

Baynes, J., J. Herbohn, I. Russell and C. Smith (2008) Bringing agroforestry technology to farmers in the Philippines: identifying constraints to success using systems modelling. School of Natural and Rural Systems Management, The University of Queensland, Gatton, 4343 Australia.

Belton V. & Stewart T.J. (2002). Multiple criteria decision analysis. An integrated approach. Kluwer Academic Publishers, Dordrecht.

Bernardes, A.T., 1999. Some mechanisms for biodiversity protection in Brazil, with emphasis on their application in the State of Minas Gerais. World Bank, Washington, D.C.

Birner, R., Wittmer, H. (2004): On the 'efficient boundaries of the state': the contribution of transaction-costs economics to the analysis of decentralization and devolution in natural resource management Environment & Planning C 22 (5), 667-685.

Braat, L., ten Brink, P. (eds.) (2008) The Cost of Policy Inaction: The case of not meeting the 2010 biodiversity target. Final report. Wageningen, Brussels, May 2008: European Commission, (contract: ENV.G.1/ETU/2007/0044)

Bräuer, I. R. Müssner, K. Marsden, F. Oosterhuis, M. Rayment, C. Miller, A. Dokova (2006) The Use of Market Incentives to Preserve Biodiversity. Final Report. Ecologic. A project under framework contract for economic analysis. ENV.G.1/FRA/2004/0081

Brooks, J.S. M.A. Franzen, C, M. Holmes, M.N. Grote and M. Borgerhoff Mulder (2006). Testing Hypotheses for the Success of Different Conservation Strategies. Conservation Biology vol. 20, No.5, 1528-1538.

Calvo-Alvarado, J. B. McLennan, A. Sánchez-Azofeifa, T. Garvin (2008) Deforestation and forest restoration in Guanacaste, Costa Rica: putting conservation policies in context. Forest Ecology and Management. Doi:10.1016/j.foreco.2008.10.035.

Clark, J.A. (2005) Why environmental scientists are becoming Bayesians. Ecology Letters, (2005) *8*: 2–14

Crawford, S.E.S. and E. Ostrom (1995). The grammar of institutions. American Political Science Review 89(3): 582-600

Cumming GS, Cumming DHM, Redman CL (2006) Scale Mismatches in Social-Ecological Systems: Causes,Consequences, and Solutions. *Ecology and Society* 11(1): 14.

Endres A (1985) Umwelt- und Ressourcenökonomie. Wissenschaftliche Buchgesellschaft, Darmstadt.

Engel S. S. Pagiola and S. Wunder (2008) Designing payments for environmental services in theory and practice: an overview of the issues. Ecological Economics 65 (2008) , 668-674

EC and BMU (European Commission and German Ministry for the Environment) (2008) TEEB – The Economics of Ecosystems and Biodiversity: An interim report. European Communities, Brussels. Available online at:
http://ec.europa.eu/environment/nature/biodiversity/economics/pdf/teeb_report.pdf

Farmani R., H. J. Henriksen, D. Savic (2009) An evolutionary Bayesian belief network methodology for optimum management of groundwater contamination. Environmental Modelling & Software 24 (2009) 303–310.

Field, B.C. ((1994) Environmental economics. McGraw-Hill, New York.

Fisher, B., R.K. Turner, P. Morling (2009) Defining and classifying ecosystem services for decision-making. Ecological Economics 68 (2009): 643-653.

GanitXL . http://centres.exeter.ac.uk/cws/technology/20-genetic-algorithms/76-ganetxl.

C. García-Fernández, M. Ruíz-Pérez, S. Wunder (2008) 'Is multiple-use forest management widely implementable in the tropics? ' Forest Ecology and Management 256: 1468-1476

Grieg-Gran, M., 2000. Fiscal Incentives for Biodiversity Conservation: The ICMS Ecológico in Brazil. Discussion Paper 00-01, International Institute for Environment and Development, London.

Harvey, C.A., O. Komar, R. Chazdon, B.G. Ferguson, B. Finegan, D.M. Griffith, M. Martínez-Ramos, H. Morales, R. Nigh, L- Soto-Pinto, M. van Breugel and M. Wishnie (2008). Integrating agricultural landscapes with biodiversity conservation in the Mesoamerican Hotspot. Conservation Biology, Vol. 22 (1):8-15.

Horton, B., Colarullo, G., Bateman, I.J., Peres, C.A., 2003. Evaluating non-user willingness to pay for a large-scale conservation programme in Amazonia: a UK/Italian contingent valuation study. Environmental Conservation 30 (2), 139–146.

Hugin Expert software: http://www.hugin.com/

Janssen, R. van Herwjinen and E.Beinat (2003) DEFINITE case studies and user manual. IVM. Vrije Universiteit

Joshi, L., G.Wibawa, F.L. Sinclair (2001) Local ecological knowledge and socio-economic factors influencing farmers' management decisions in jungle rubber agroforestry systems in Jambi, Indonesia. DFID Project R7264 Forestry Research Programme. October 2001

Kjærulff, U. B. and Madsen, A. L. (2008). Bayesian networks and influence diagrams: A Guide to Construction and Analysis. Series: Information Science and Statistics. Springer-Verlag.

Knight, A.T., R.M. Cowling, and B.M. Campbell (2006) An Operational Model for Implementating Connservation Action. Conservation Biology Volume 20, No.2, 408-419.

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

Langpap C., J. Wu (2004). Voluntary conservation of endangered species: when does no regulatory assurance mean no conservation? Journal of Environmental Economics and Management. 47: 435–45.

Loureiro, W., 1998. Incentivos Econômicos para Conservação da Biodiversidade no Brasil: ICMS Ecológico. IAP, Curitiba.

Loureiro, W., 2002. Contribuição do ICMS Ecológico à conservação da biodiversidade no Estado do Paraná. Tese Doutorado, Universidade Federal do Paraná, Curitiba.

Köllner, T., Schelske, O., Seidl, I., 2002. Integrating biodiversity into intergovernmental fiscal transfers based on cantonal benchmarking: a Swiss case study. Basic and Applied Ecology 3, 381–391.

Loreau, M., J. A. Downing, M. Emmerson, A. Gonzalez, J. Hughes, P. Inchausti, J. Joshi, J. Norberg, and O. Sala. 2002. A new look at the relationship between diversity and stability. Pages 294 *in* M. Loreau, S. Naeem, and P. Inchausti, editors. Biodiversity and Ecosystem Functioning: Synthesis and Perspectives. Oxford University Press, Oxford.

Mather, A.S., 1990. Global Forest Resources. Bellhaven Press, London.

May, P.H., Veiga Neto, F., Denardin, V., Loureiro, W., 2002. Using fiscal instruments to encourage conservation: Municipal responses to the 'ecological' value-added tax in Paraná and Minas Gerais, Brazil. In: Pagiola, S., Bishop, J., Landell-Mills, N. (Eds.), Selling Forest Environmental Services: Market-based Mechanisms for Conservation and Development. Earthscan, London, pp. 173–199.

Mertens, B. and E Lambin (1997) Spatial modelling of deforestation in southern Cameroon: spatial disaggregation of diverse deforestation processes. Applied Geography 17(2): 143-162.

MEA (2005) Ecosystems and Human Well-Being: Policy Responses Findings of the Responses Working Group. Millennium Ecosystem Assessment Vol.3, Island Press.

Miranda, M. , C. Diepering, P Glasbergen (2006) Costa Rican environmental services payments: the use of a financial instrument in participatory forest management. Environmental Management 38: 562-571.

Moffett A. & Sarkar S. (2005). Incorporating multiple criteria into the design of conservation area networks: a minireview with recommendations. Diversity and Distributions 12: 125-137.

Mullan, K. and Kontoleon, A. (2008). Benefits and costs of forest biodiversity: Economic theory and case study evidence. Final report, July 2008.

Munda, G. (2008). Social multi-criteria evaluation for a sustainable economy. Springer. pp.210
Ostrom , E. (1990). Governing the Commons. The Evolutin of Institutions for Collective Action. Political Economy of Institutions and Decisions. Cambridge University Press.

OECD (1997) Evaluating economic instruments for environmental policy. OECD, Paris.

OECD (2007) Instrument Mixes for Environmental Policy. OECD, Paris.

Ostrom, E. (2007) A diagnostic approach for going beyond panaceas. PNAS, September 25 2007, vol.104, no.39, pp. 15181-15187.

Padgee A. K. Yeon-Su, P.J. Daugherty (2006) What makes community forest management successful: a meta-study from community forests throughout the World. Society and Natural Resource, 19:33-52.

Pagiola, S. , P. Agostini, J. Gobbi, C. de Haan, M. Ibrahim, E. Murgeitio, E. Ramírez, M. Rosales, J.P. Ruíz(2004) Paying for biodiversity conservation services in agricultural landscapes. Environment Department Paper No. 96. World Bank.

Pagiola, S. (2008) Payments for environmental services in Costa Rica. Ecological Economics 65 (2008): 712-724.

Pagiola, S. and B. Bosquet (2009) Estimating the costs of REDD at the country level. MPRA Paper No. 13726, posted 02. March 2009.

Pagiola, S. A. Arcenas, 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 Vol. 33, No.2: 237-253.

Perrings C, Gadgil M (2003) Conserving Biodiversity: Reconciling Local and Global Public Benefits. In Kaul I, Conceição P, Le Goulven K, Mendoza RU (eds): Providing Global Public Goods: Managing Globalization,Oxford University Press, Oxford, pp. 532-555.

Polasky S., Nelson E., Lonsdorf E., Fackler P. & Starfield A. (2005). Conserving species in a working landscape: land use with biological and economic-objectives (vol 15, pg 1387, 2005). Ecological Applications 15: 2209-2209.

Porras, I, M. Grieg-Gran, N. Neves (forthcoming). All that glitters… A review of payments for watershed services in developing countries. IIED

Prates, J. C. A. (2008). As Finanças Locais e a promoção da Conservação da Natureza, Dissertação apresentada na Faculdade de Ciências e Tecnologia da Universidade Nova de Lisboa para obtenção do grau de Mestre em Engenharia do Ambiente, perfil Gestão e Sistemas Ambientais, Lisbon.

Renn. O. (2008) Risk Governance. Earthscan. Pp 445.

Ring, I. (2002): Ecological public functions and fiscal equalisation at the local level in Germany. Ecological Economics 42, 415-427

Ring, I. (2008a): Biodiversity governance: Adjusting local costs and global benefits. In: Sikor, T. (Ed.): Public and Private in Natural Resource Governance: A False Dichotomy? Earthscan, London, 107-126

Ring, I. (2008b): Integrating local ecological services into intergovernmental fiscal transfers: the case of the ecological ICMS in Brazil. Land use policy 25(4), 485-497

Ring, I. (2008c): Compensating municipalities for protected areas. Fiscal transfers for biodiversity conservation in Saxony, Germany. GAIA 17/S1, 143-151

Robalino J. , A Pfaff, G.A. Sanchez-Azofeifa, F. Alpizar, C. León, C.M. Rodríguez (2008) Deforestation Impacts of Environmental Services Payments. Costa Rica's PSA Program 2000-2005. Environment for Development Discussion Paper Series. August 2008 EfD DP 08-24.

Sanchez-Azofeifa, A. , A. Pfaff, J. Robalino and J. Boomhower (2007) Costa Rican payment for environmental services programme: intention, implementation and impact. Conservation Biology 21(5): 1165-73

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: 131-41

Similä J., Thum R., Varjopuro R. Ring I. (2006) Protected species in conflict with fisheries: the interplay between European and national regulation. Journal of European Environmental Planning and Law 5: 432-455.

Spash, C.(2008) How much is that ecosystem in the window? The one with the bio-diverse trail. Environmental Values 17(2008):259-284.

Sterner, T. (2003) Policy Instruments for Environmental and Natural Resource Management. Resources for the Future.

Vatn , A. (2005) Institutions and the Environment. Edward Elgar.

Vatn, A. (2008) Payments for Environmental Services: An Institutional Analysis. Paper presented at the 10th biennial conference of ISEE 'Applying Ecological Economics for Social and Environmental Sustainability', Nairobi, 7-11 August. 2008.

Wätzold, F., Schwerdtner, K. (2005): Why be wasteful when preserving a valuable resource? A review article on the cost-effectiveness of European biodiversity conservation policy. Biological Conservation 123(3): 327-338

WinBUGS (Bayesian inference Using Gibbs Sampling) http://www.mrc-bsu.cam.ac.uk/bugs/

Wolf, S.A. and E. Primmer (2006) Between incentives and action: A Pilot Study of Biodiversity Conservation Competencies for Multifunctional Forest Management in Finland. Society and Natural Resources. 19(9): 845-861.

Wunder (2003) Oil wealth and the fate of the forest. A comparative study of eight tropical developing countries. Routledge, London.

Wunder , S. (2007) Efficiency of payments for environmental services. Conservation Biology Volume 21, No.1 February 2007.

Wunder, S. (2008) Necessary conditions for ecosystem service payments. Conference Paper. Economics and Conservation in the Tropics: A Strategic Dialogue, January 31- February 1, 2008.

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

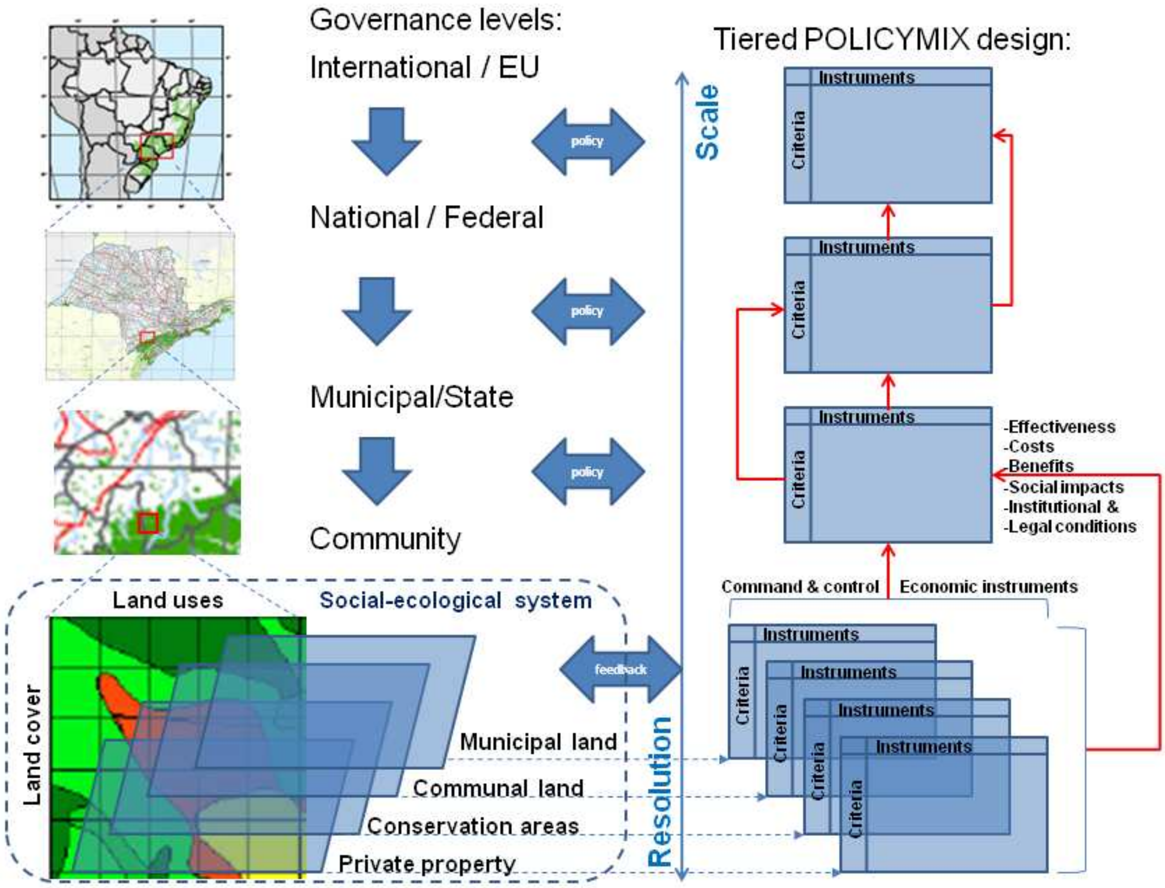

Figure 1 A multi-tiered policy impact assessment framework needs to address interaction across instruments due to common governance structures of apparently alternative instrument; be robust to correlation across assessment criteria due to spatial interactions (externalities) between land-uses that are subject to policy mixes. Case-based, rather than theory-driven assessment assumes that experiences on the ground determine policy design at higher governance levels.

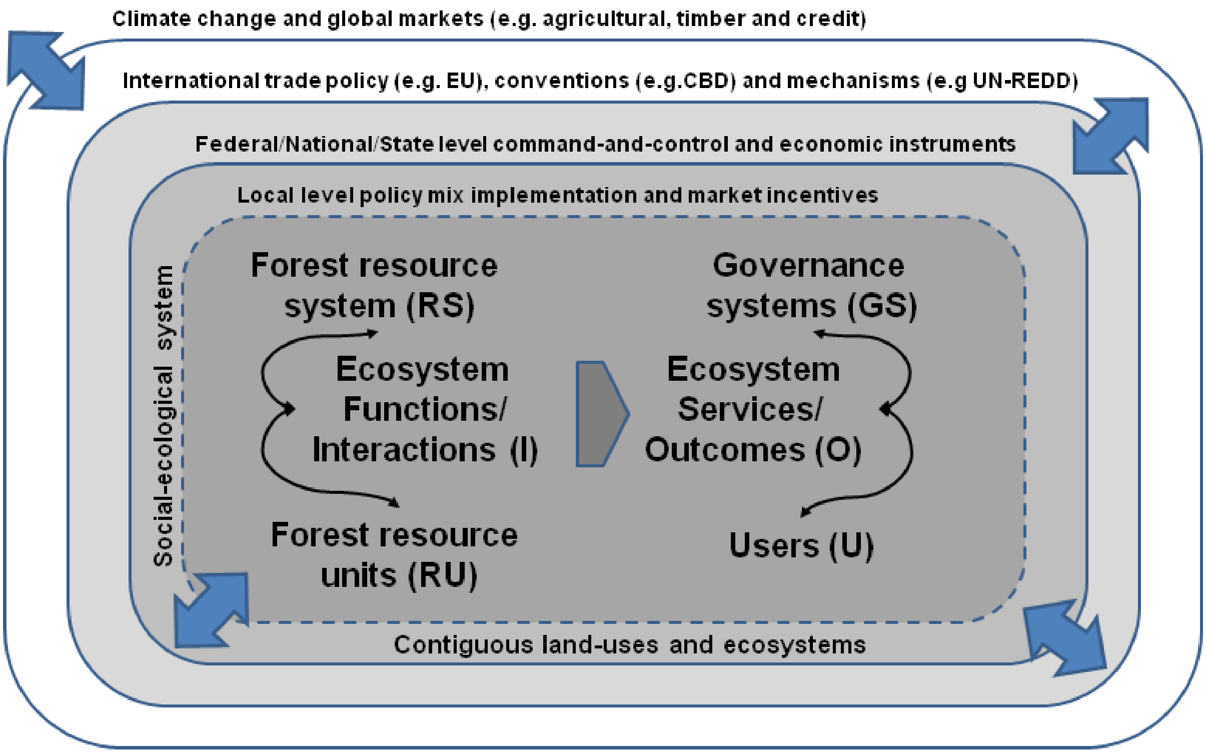

Figure 2 A multi-tier and multi-scale framework for analyzing the impacts of economic instruments in policy mixes on social-ecological systems.

Table 1 A comparison of variables across different empirical approaches to the analysis of forest conservation policies

Otrom(2007)		Brooks et al. 2006	Padgee et al.2006	Wunder, Engel, Pagiola (2008)	Angelsen (2007)
	Social-ecological systems (SES)	Integrated Conservation and Development Projects (ICDPs)	Community Forestry Management (CFM)	Payments for Environmental Services (PES)	Spatial land rent models
Resource System (RS)	RS1- Sector (e.g., forests) RS2- Clarity of system boundaries RS3- Size of resource system RS4- Human-constructed facilities RS5- Productivity of system RS6- Equilibrium properties RS7- Predictability of system dynamics RS8- Storage characteristics			Spatial scale and current size	Yield (y^i) = f^i (l^i,k^i, agroecological conditions: soil quality, rainfall, temperature) Capital k^i = k(technologies applied) v = v(extent and quality of roads, rivers, general infrastructure)
Resource Units (RU)	RU1- Resource unit mobility RU2- Growth or replacement rate RU3- Interaction among resource units RU4- Economic value RU5- Size RU6- Distinctive markings RU7- Spatial & temporal distribution				Yield (y^i) = f^i (agroecological conditions: soil quality, rainfall, temperature
Users (U)	U1- Number of users U2- Socioeconomic attributes of users U3- History of use U4- Location U5- Leadership/entrepreneurship U6- Norms/social capital U7- Knowledge of SES/mental models U8- Dependence on resource	Market integration - market sales - market purchase - wage labor - threat (type of use and severity) Homogeneity of population	Skills and experiences Strong leadership and effective organizing (present/absent)	Who else benefits?	Labour l^i = l(technologies applied) p^i = p(distance to centre, market demand) w = w(distance to centre, off-farm employment opportunities, migration) q= q(distance to centre) c^i = c(distance to centre) v = v(distance to centre)
Governance System (GS)	GS1- Government organizations GS2- Non-government organizations GS3- Network structure GS4- Property-rights systems GS5- Operational rules GS6- Collective-choice rules GS7- Constitutional rules	Utilization/protection - IUCN ranking - use permitted Decentralization -implementation level - local decision-making - targeted beneficiaries	Tenure security Clear ownership Clearly defined boundaries Designated areas Congruence biophysical and social boundaries Rules and regulations	Environmental services targeted Environmental services paid for Who buys? Who sells? Who initiated? Intermediaries External donor support	c^i = c(property regime, law enforcement/corruption, land competition)

Governance System (GS) (cont.)	GS8- Monitoring & sanctioning processes		Effective enforcement Monitoring Sanctions Local responsibility Local autority (present/absent)	Seller selection Monitoring Sanctions Conditionality Payment (mode, amount, timing Differentiation) Contract duration	
Interactions (I)	I1- Harvesting levels of diverse users I2- Information sharing among users I3- Deliberation processes I4- Conflicts among users I5- Investment activities I6- Lobbying activities			Obstacles to implementation	Market prices (p^i) Wage rates (w) Annual costs of capital (q) Cost of enforcing property rights (c) Access costs (v)
Outcomes (O)	O1- Social performance measures (e.g., efficiency, equity, accountability) O2- Ecological performance measures (e.g., overharvested, resilience, diversity) O3- Externalities to other SESs	Ecological Economic Attitudinal Behavioral (failure, limited success, success)	Success/failure		Land rent: $r^i(d) = p^i(d)\, f^i(l^i, k^i) - w(d)l^i - q(d)k^i - c^i(d) - v(d)$
Social, Economic, and Political Settings (S)	S1- Economic development. S2- Demographic trends. S3- Political stability. S4- Government settlement policies. S5- Market incentives. S6- Media organization.			Linked to other policy tools?	p^i = p(taxes, subsidies, export/import regulations, marketing efficiency(competition)) w = w(economic development) q= q(access to credit, interest rates)
	Social-ecological systems (SES)	Integrated Conservation and Development Projects (ICDPs)	Community Forestry Management (CFM)	Payments for Environmental Services (PES)	Spatial land rent models (i= index for land use sector)

Note : for each of the studies we have tried to organise explanatory variables based on the greatest similarities with SES framework second tier variables. The exercise is meant to generate discussion on the complementarities between policy analysis approaches, as well as on the causality /hierarchy of explanatory variables for forest cover change, and in turn biodiversity conservation and ecosystem services provision.

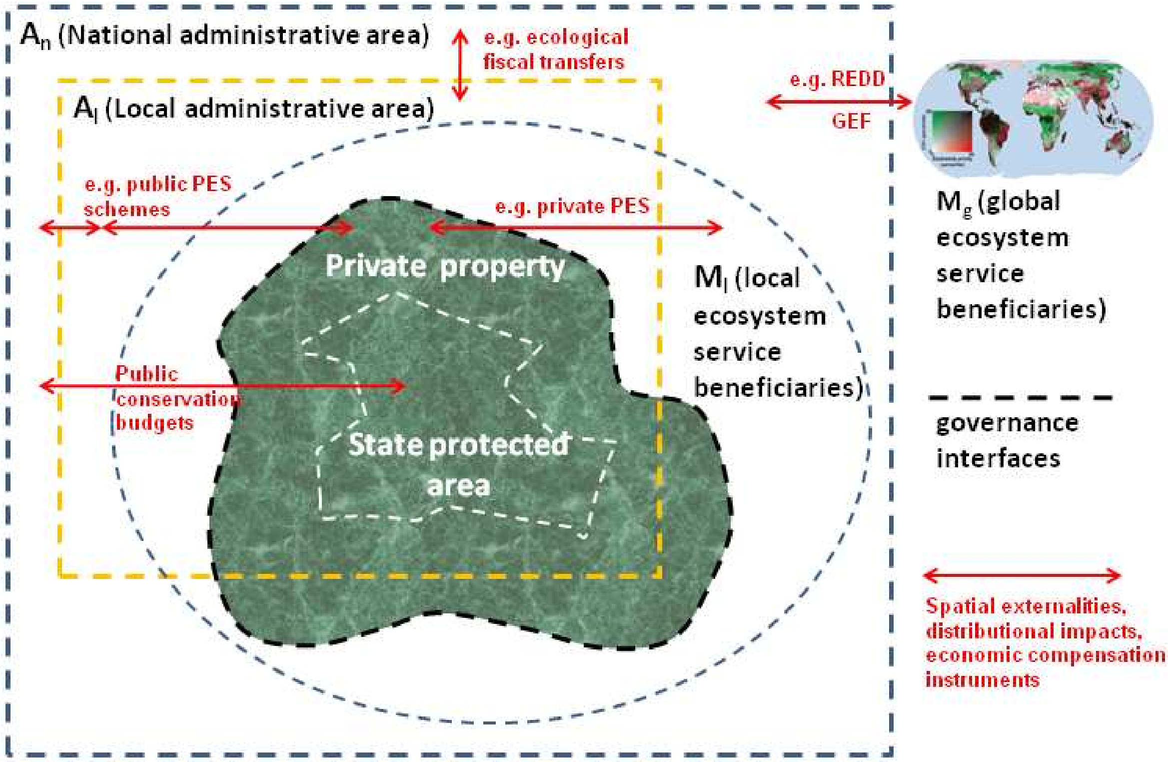

Figure 3 Different economic instruments play different complementary roles across the same landscape, given the configuration of overlapping forest resource, use and governance characteristics. The role of economic valuation is to quantify the externalities (which happen by definition across governance interfaces).

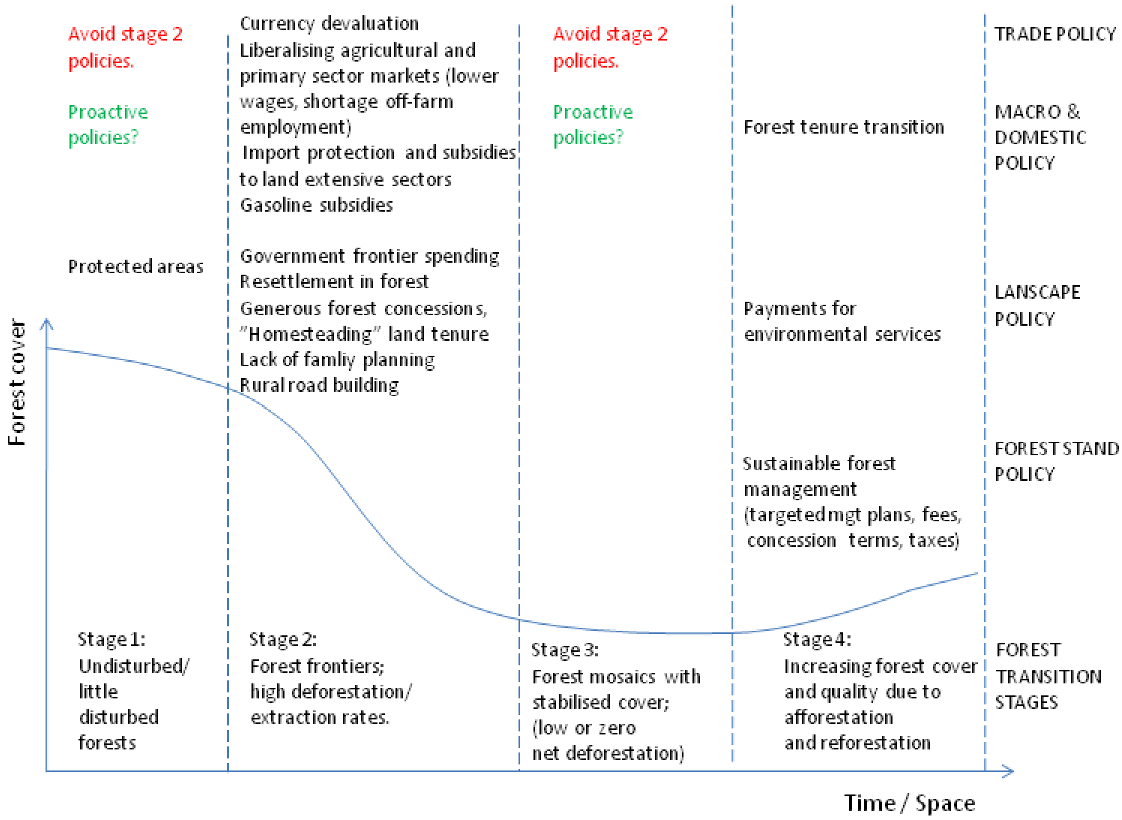

Figure 4 Forest transition stages and policy examples at each stage. Research has tended to focus on policies causing changes in forest cover. Part of the policy mix in stage 1 and stage 3 is avoiding policy and institutional failure of stage 2, but are the proactive policies the same as for stage 4? Adapted from Angelsen(2007), Wunder 2003, García-Fernández et al. (2008) and Barbier et al. (2009)

Figure 5: Twelve states in Brazil have introduced the ecological ICMS that redistributes part of their state value-added tax income back to municipalities based on “Conservation Units”.